12-15-2019

Mojave Desert Ecosystem Recovery: Potency of Biotic and Abiotic Restoration Methods in Low Elevation Plant Communities

Mary Amanda Balogh

MOJAVE DESERT ECOSYSTEM RECOVERY: POTENCY OF BIOTIC AND ABIOTIC RESTORATION METHODS IN LOW ELEVATION PLANT COMMUNITIES

By

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Bachelor of Environmental Science
Loyola Marymount University
2015

A thesis submitted in partial fulfillment
of the requirement for the

Master of Science – Biological Sciences

School of Life Sciences
College of Sciences
The Graduate College

University of Nevada, Las Vegas

December 2019

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Thesis Approval

The Graduate College
The University of Nevada, Las Vegas

November 8, 2019

This thesis prepared by

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entitled

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is approved in partial fulfillment of the requirements for the degree of

Master of Science – Biological Sciences
School of Life Sciences

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Abstract

Mojave Desert ecosystem recovery: potency of biotic and abiotic methods
in low elevation plant communities

By

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The historic and current state of land in the Mojave Desert, including land managed by the National Park Service with fundamental goals of natural resource conservation and preservation, been severely degraded by a variety of anthropogenic disturbances. Due to increasingly sporadic and unpredictable precipitation patterns, land managers struggle to implement restoration projects with high success rates and are resource-limited for post-treatment monitoring. In this study, I examined success rates of biotic (outplanting, seeding) and abiotic (soil manipulation, vertical mulch) restoration treatments on various disturbance types in the creosote-bursage (*Larrea tridentata-Ambrosia dumosa*), blackbrush (*Coleogyne ramosissima*), and Joshua tree woodland (*Yucca brevifolia*) plant communities. Sites were surveyed in springs of 2018 and 2019 for annual and perennial plant species richness, percent cover, and perennial density to determine the effect size between unrestored, treatment, and reference (undisturbed) plot types. Sites were compared to determine what restoration treatment is most successful based on plant community, disturbance type, and the time since restoration or disturbance. Both biotic and abiotic treatments typically exhibited positive rather than negative restoration responses. Biotic treatments tended to have a more positive restoration success response than abiotic treatments. A large number of perennial effects were sensed while annual

effects were often undetected. This study aims to provide evidence-based decision tools for land managers to choose restoration methods in an ecologically and economically effective manner.

Acknowledgements

There could be no greater pleasure than learning from, and working with, the people I have met over the past two years. Each individual's incandescent passion for our planet synergizes into a radiating force that rivals the energy emitted by the Mojave Desert sun. When I reflect on all of the fellow scientists that I wish to acknowledge, I am happy to say our interests as ecologists are so diverse that we seem to represent a complete array of the components in an ecosystem (a reference ecosystem, of course). I want to thank all of you: from the edaphologists, microbiologists, and hydrologists, to the entomologists, botanists, and herpetologists, up to the landscape ecologists and geographers!

I am so grateful to my thesis committee advisor, Dr. Scott Abella. He considered my interests, strengths, and even quirks, and insightfully shaped a thesis project that was quintessentially *Mary*. My other committee members showed me positivity, patience, and encouragement. I used their advice that contained their specific field knowledge to better my work, by adding wholeness to my thesis. I hope I can confer my full appreciation and respect to Dr. Matthew Petrie, Dr. Stan Smith, and Dr. Haroon Steven. I may still very well be botanizing in the middle of the desert if it weren't for the focus, guidance, and steadfastness shown to me. Without Lindsay Chiquoine, my data would never have moved from the datasheet to meaningful figures and discussion. Thank you for maintaining helpful experimental dialogue and your statistical work. That being said, for your sake, I am sorry that I was given the closest possible office to yours. I want to recognize recent past members of the Abella Applied Ecology lab and current members, Vivian Sam and Dominic Gentilcore for their support as we persevered through this undertaking together. To Katie Laushman and Carlee Coleman, thank you for

bearing difficult conditions while still providing thoughtful suggestion and hard work and even harder laughs. I am convinced that field ecologists develop some of the most bizarre and sincere friendships. A special and loving mention is in order to very long-time editor, Steve T. Balogh, Sr.

Finally, I would like to recognize the National Park Service for permission and access to the field sites. Several employees in Lake Mead National Recreation Area, Joshua Tree National park, and the Mojave National Preserve provided invaluable information that made this thesis possible.

Figure 1. If not ecology, then what?
Photo Credit: Stephen T. Balogh, Jr.

Dedication

For Ranger James

Cactus Cop 🌵 Post-Fieldwork Support Staff 🌵 Rattlesnake Whisperer

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Table of Contents

List of Tables

List of Figures

Chapter 1. An Introduction: Humans and the eastern Mojave Desert have a rich history and interconnectedness, but an accurate and holistic understanding of Arid Restoration Ecology is essential to restore the harmonious mutualism that has long been lost.

1.1 Project Introduction

The past and present Mojave Desert suffers severe aberrant degradation from climate change and direct anthropogenic impacts. Because of slow natural recovery rates in arid regions, active restoration methods are often employed to hasten recovery. While a scant portion of literature suggests that some disturbed dryland plant communities have repaired themselves within decades, more literature suggests this process can easily take centuries (Abella 2010; Lovich & Bainbridge 1999). Population growth and socioeconomic opportunity have firmly taken root in a "Go West, Young Man" American philosophy that continues into today, necessitating further infrastructure changes and subsequent inevitable damage. The draw of adventuring across the arid Southwest that began in Conestoga wagons and trains continues in electric cars and airplanes.

Minimal precipitation, high evapotranspiration rates, and scarce vegetation are the barebones principles for defining a desert. Arid vegetation restoration is already a challenge. Current predictions of climate change suggest further hardships on successful restoration. The coupling of basic desert principles with climate change tumults will confound arid vegetation restoration to new levels of complication. Due to increasingly sporadic and unpredictable precipitation events and extreme temperatures correlated with climate change, land managers struggle to implement restoration projects with high success rates (e.g. high plant recruitment and survival into maturity). Plant communities respond differently to restoration treatments, due in part to differences in elevation, precipitation, and ecosystem interactions, but also temporally via precipitation.

While many restoration treatments have been implemented over the past few decades, a lack of short-term and long-term monitoring can hinder understanding treatment efficacy. These recent years have been characterized by a crescendo of calls asking for the inclusion of monitoring and economic findings in published experimental results, but these calls have been answered with painfully slow and quiet compliance. For example, a study by Copeland and others (2017) aimed to assess long-term trends by using nearly 4,000 restoration treatments on Bureau of Land Management land in the Southwest USA. They determined that a mere 9.5% of the projects included post treatment monitoring (Copeland et al. 2017). In addition to the monitoring deficiency that many consider essential, economic uncertainty is causing trepidation to those that control restoration budgets. The inflation-adjusted mean cost for restoration services has risen from slightly over \$8,000 km^{-2} in the 1950's to over \$46,000 km^{-2} in the past decade; the median costs have also almost tripled (Copeland et al. 2017). This serious fiscal upsurge coupled with limited post treatment monitoring data precipitates the need for old restoration site revisitation. Long-term monitoring can provide crucial insight, especially when treatment main effects supersede year main effects that are most nearly directly controlled by inter-annual and intra-annual meteorological events (Beatley 1973). Using this information, treatment effectiveness can be compared to the cost for land managers and hopefully implemented with confidence.

1.2 Purpose and Justification of Research

In addition to garnering new and fundamental ideas about desert ecology, this study aimed to provide evidence-based decision tools for land managers as a means of choosing suitable restoration methods in an ecologically and economically effective manner. It attempted to summarize evidence for or against the necessity of spending time, money, and resources focusing on one or multiple of common restoration techniques. To do this, I assessed whether plant communities across an ecological gradient varied in their recovery rates to popular restoration techniques. Some sections explored how time since restoration (TSR) may influence perceptions of the short-term and long-term value of treatments and whether these perceptions hold true throughout several years.

Climate change may cause deserts to become drier, hotter, and with more sporadic rain events; further desertification of semi-arid and non-arid lands has been identified as an issue (Archer & Predick 2008). Long-term climate aside, weather on a short-term scale can have drastic impacts on restoration success because of the instability that is difficult to adjust for and anticipate. Consequently, the level of accurate and precise comprehension of arid restoration could play a key role in an uncertain future. A better understanding should include conclusions that explain just how different data across different biomes lend themselves to the overall picture of restoration effectiveness.

1.3 Further Notes

In lieu of a chapter devoted to an all-inclusive literature review, I have decided to present my thesis in a manuscript format. Each of the chapters following this basic introductory chapter serves as two independent research articles. The second chapter consists of a synthesis of six restoration field sites on federal lands in the arid Southwest United States. The third chapter describes a case study in long-term succession and restoration response regarding one of the sites: the Fish Hatchery outplanting at Lake Mead National Recreation Area, mentioned in the first chapter in further detail.

Tables and figures can be found immediately following the Literature Cited page for each chapter. All tests of statistical significance for Chapter 2 are listed in Appendix I and for Chapter 3 are in Appendix II.

Chapter 2. A synthesis of six Mojave Desert sites with biotic and abiotic restoration treatments: quantitative restoration success measurements within sites enables qualitative comparisons across sites and create a composite picture of successful arid restoration ecology

2.1 Introduction

Drylands, which represent 41% of terrestrial land are regrettably overlooked and underrepresented in literature by both the public and scientific eye, especially considering its proportional land coverage (Adeel et al. 2005). Unfortunately, the majority of current literature regarding socio-economic values as reasons for restoration focuses primarily on forests, grasses, and shrublands and so simple extrapolation is futile (Wortley et al. 2013). This study focused on the Mojave Desert specifically. Of worldwide drylands, 10-20% of them are considered anthropogenically degraded (Adeel et al. 2005; Hulvey et al. 2017). In addition, this degradation has a disproportionately deleterious effect on common restoration measures because of the slow pace of plant growth and therefore succession and the inherent water-limited and resource-poor conditions of drylands (Hulvey et al. 2017). Restoration has become one of the only options left to maintain this ecosystem. While it is difficult to generalize restoration outcomes from a specific site to a broad spectrum due to differences in aims, challenges, and other factors even within the same biome, adding to the increasing body of arid restoration literature can enrich the collective knowledge and secure arid restoration ecology as a fully necessary science (Wortley et al. 2013). Much of the current experiments are conducted in North America (Ruiz-Jaen & Aide 2005), this study included, but further and more global research leads to more generalities to be posited concerning arid ecological restoration that could then be examined on a more global scale, increasing knowledge and thereafter the success of arid restoration ecological practices.

One particular hindrance of restoration ecology theory transmuting to practice is the lack of consensus regarding how exactly to measure whether an ecosystem is restored and the resource requirements to do so. In 2004, the Society for Ecological Restoration International (SER) introduced a primer of attributes meant to define, outline, and provide parameters describing a successful restoration. It the further suggests variables of measure to a restoration project compared to the parameters of a successful one. Ruiz-Jaen & Aide (2005) and Wortley et al. (2013) agree that the primary outcomes of the SER attributes are to enhance vegetation structure, species diversity and abundance, and ecological processes while also noting the necessity of incorporating socioeconomic factors. These factors are gaining traction, however this paper will not focus on these outside of human recreation and enjoyments

2.2 Society for Ecological Restoration International Primer Attributes

The Society for Ecological Restoration International (SER) lists nine attributes of restored ecosystems to determine when restoration has been accomplished. The definitions and attributes provided in the document are meant to help clarify, standardize, and study restoration ecology. The attributes express what the SER believes are important components of a healthy ecosystem that should be exhibited by a restoration site to some extent in order to be considered "successful." While each of the attributes do not have to be fully realized to be considered efficacious (alternatively, they can simply exist on the proper trajectory), measuring data to provide restoration evidence in relationship to these attributes can substantially decrease restoration costs to land managers in several ways. Ecosystems always require minimal management at the least, whether restored or undisturbed; using data from past projects and some of the nine SER attributes allowed me to speculate on restoration success compared to effort so that land managers can proceed with maintaining each unique site. Typically, a site is monitored for its vegetation structure, diversity, and ecological processes (Ruiz-Jaen & Aide 2005; Wortley et al. 2013). The purpose of this thesis was to assess only vegetation structure and diversity. It is an *a priori* assumption that diverse vegetation structure will likely lead to diverse fauna within these sites, especially because of the sites' propinquities to undisturbed desert areas. The same can be said for ecosystem functions (Ruiz-Jaen & Aide 2005). Therefore, restoring vegetation at the minimum will have a bottom-up ecosystem ripple.

The first attribute defines that a restored ecosystem contains a "characteristic assemblage of species that occur in the reference ecosystem and that provide appropriate community structure" (SER 2004). I measured this directly using percent cover, species richness, and perennial density in my three delineated plot types: reference (undisturbed), treatment (restored, disturbed), and unrestored (no treatment, disturbed). Not only should many of the same species be present in the same percentages, but they also should provide community structure. Proper community structure, whether this be diversity in vegetation, fertile island development, heterogeneous desert pavement patterns, or the like, allows for proper interactions among and across biotic and abiotic components. While percent cover and species richness separately do not describe the picture of biodiversity, qualitatively discussing them together gives rise to hypotheses about the overall species composition or the taxonomic array of species present. This species richness and composition together provide an ample idea of biodiversity comparisons between the plot types. To assume restoration may be successful, I would expect to see the treatment plots of earlier years more dissimilar to reference plots but becoming increasingly similar over time in multi-year monitoring sites. Among inter-site single year analyses, I would expect newer restoration sites to have treatment plots more dissimilar to reference plots than sites that were restored earlier.

The second attribute specified is the presence of native species to the greatest practical extent. Here, the primer distinguishes between invasive species versus non-invasive ruderal and segetal species. Segetal species are weeds within crop fields and thus have long attracted negative attention, but here, ruderal species are of great ecological harm. Ruderal species colonize disturbed lands. I do not categorize the exotic plants as invasive or non-invasive, but make mention of which species tended to be ruderal in each site. For example, exotic grasses in all sites will usually be outcompeting natives. Some exotic plants, especially *Brassicaceae* species, found at sites are sometimes considered introduced and even naturalized rather than an invasive exotic that requires a costly eradication scheme. There is still considerable debate as to whether exotic species are functionally redundant to the natives they displaced or if the magnitude at which they affect ecosystem processes and the subsequent loss of biodiversity is so high that they must be eradicated (Fleishman et al. 2003).

The third attribute describes that all functional groups necessary for continued development and stability are present (SER 2004). To determine functional groups and the relative abundances I would expect to find among the vegetation, I further categorized plants based on their life history group. For example, perennial plants are classified as either forbs, woody perennials or shrubs, or grasses. Shrubs are further characterized based on stature. Small or dense subshrubs do not occupy the same niche and provide the same role as a large shrub like *Larrea tridentata* (creosote bush) or an arborescent *Yucca;* however, both contribute unique element to the ecosystem as a whole. For example, taller species can increase bird recovery and seed dispersal (Ruiz-Jaen & Aide 2015).

The fourth attribute is a sustainable physical environment. That is, species can easily reproduce and continue through time without human intervention at each season or every couple of years outside of minimal ecological management. I measured these indirectly using projects with multiple years of monitoring data available. If a trend developed over time where certain perennial functional groups were dying off and not being replaced, I may conclude that the restoration did not successfully provide the ingredients necessary for the ecosystem's individual success.

The fifth attribute is that the ecosystem functions normally for its stage of development; that signs of dysfunction are largely absent (SER 2004). To measure this would require longer-term monitoring and measuring beyond just flora. While this thesis tried to address age and restoration, in order to determine normal functioning it would be integral to measure other factors: fauna usage, soil nutrients, and hydrology, to name a few.

The sixth attribute requires the ecosystem to be suitably integrated into the larger ecosystem matrix, therefore experiencing exchanges at broader scales. As more development occurs on arid lands throughout the world, this integration attribute is becoming increasingly difficult to attain. For example, many of the sites chosen for this thesis are very proximal to populated or well-used recreation areas within the National Park System. Roads cause habitat fragmentation, possibly disrupting gene flow between biotic populations or juvenile dispersal across a large range (Lovich & Bainbridge 1999).

The seventh attribute ties in closely with the sixth, and among the sites used in this thesis, is only partially achievable. The seventh attribute provides that potential threats to ecosystem health from surrounding areas are eliminated or reduced (SER 2004). Because of the popularity of the federal lands system, attribute seven was never fully achieved among field sites. Sites were either directly roadside, or easily accessed by foot. The one possible exception may have been Morningstar Mine, which aside from its fairly remote location, was fenced off and locked. However, vandalism has been rampant at the site because it is very easy to break into (Figure 1). On several occasions, people have trespassed and stolen metal wires and piping to sell. One such event damaged the water bladder that was irrigating the plants to such a state of disrepair, the idea of irrigating the site's new outplants had to be abandoned (personal correspondence, National Park Service). On another occasion, expensive wires were dug up and removed from the treatment area, creating a deep furrow that uprooted plants. Aside from intentional vandalism, people unknowingly hinder restoration projects. For example, a disturbed but restored area probably does not look as verdant and wild as its reference. Accordingly, this is the area people tend to walk around for pictures or other explorations as the path of least resistance (personal observation).

The eighth attribute stipulates that the ecosystem must be resilient enough to withstand the normal, periodic, stress event (SER 2004). For example, drought years and wet monsoon years vary greatly in the Mojave (Beatley 1973). The same can be said for extreme temperature. Established plant should have enough stability to survive stress events, or if not, spread out seeds ensuring the species' continued success.

The final attribute is that the restored ecosystem is self-sustaining to the same degree as its reference and will continue indefinitely (SER 2004). I qualitatively measured this by comparing the ratios of unrestored, treatment, and reference plots as well as using individual sites over time. If, over the years, treatment plots are moving more quickly to reference levels compared to the unrestored plots, restoration may be successful. Overall, an area will likely restore itself eventually. The question is: how long will this take? Will it be a useful ecosystem functioning unit?

The attributes are vaguely written in a way that they can be applied to all ecosystems and allow land managers to make individualized decisions for success on a site-to-site basis. For example, a fecund and quickly regenerating rainforest has different restoration requirements and goals than a desert, but the main target of improving biodiversity, abundance, and ecosystem structure and services is shared, regardless that the biodiversity and ecosystem services between the two biomes are vastly different. Despite exclusivity, several common restoration techniques emerged when parsing through both old and new literature. The field is currently, and will continue, to expand (Wortley et al. 2013).

The SER Primer states that, "The monitoring data lend themselves to the plotting of trajectories for individual parameters, but their combination into a single trajectory representing the entire ecosystem requires highly complex multivariate analysis of a kind that has yet to be developed. This represents a critical research challenge for the future." We should combine a multitude of similar studies to qualitatively describe possible outcomes, despite lacking the complex multivariate analysis. This combined qualitative assessment can then be applied to a similar restoration still in an inchoate planning stage.

Measuring the SER Primer attributes requires extensive resources and long-term survey data, but restoration success monitoring rarely exceeds five years (Ruiz-Jaen & Aide 2005). I attempted to locate information resources for restoration projects that exceed this five-year limitation.

Data regarding restoration success can be measured differently, and in the SER was classified in three ways:

1. Direct comparison involves selecting specific parameters and comparing with the reference.
2. Analyzing the nine attributes is a semi-quantitative approach that determines which and to what degree goals are being met.
3. In trajectory analysis, trends are established through time to determine whether restoration is leading a site towards reference trajectories (SER 2004).

This thesis included specific measurements, like annual forb percent cover or species-specific densities, of particular study sites. A frequently posited and rarely agreed upon question (how many site variables must closely meet their paired undisturbed parameters be considered successful?) is avoided by shifting to the meta-analysis qualitative framework at this point. This remains a valid question that the SER primer suggests can be answered by proper goal setting within the first stages of planning. After analyzing each site separately, I was able to qualitatively analyze attributes and their circumstances off all sites, which share the main goal of aligning and expediting the disturbed ecosystem trajectory to an undisturbed trajectory. Analyzing trajectories across the broad range of diverse sites was difficult because I was unable to control endless unique situations and variables. I attempted to establish a "restoration effect" or similarity of the trajectories among sites over time to circumvent this. While I cannot say a certain restoration treatment works better on a certain variable in a certain location at any given time, critical broad statements regarding the attributes can help land managers extrapolate useful data to their respective field sites.

2.3 Background, Experimental Theory, & Design

The western, especially the southwestern, part of the United States has long been considered the omphalos for federal lands agencies because of the sheer extent of wildlands and high-visitation parks, preserves, recreation areas, monuments, forests, waterways, and countless other protected land types. For example, there are nearly 100 National Park Service units in the six Southwest states alone (NPS.gov). As of 2015, the five main federal lands agencies (Bureau of Land Management, Department of Defense, Fish and Wildlife Service, Forest Service, and National Park Service) oversaw nearly 80 % of Nevada and 46% of California land (Vincent et al. 2017). This vast land ownership is not without problems. An issue that quite blatantly exists because of this vastness is the Tragedy of the Commons played out in an arid and modern environment: much of this vast land ownership is governed under rather nonspecific rules for landscape-degrading livelihoods like claiming stakes, ranching, and grazing. These activities, which are inherently difficult to control, have effects that are economically condoned, politically legal, but with ecological consequences that are dire. Moreover, there is not enough labor to even remotely patrol every hectare. Much of the anthropogenic damage arises from recreational activities and population growth, as concomitant infrastructure expands. The federal government, through grants and third parties, can manage only some of the inevitable damage. The Mojave Desert in the Southwestern United States is no exception to the influx of anthropogenic stress factors and is a reason why we must strive to better understand arid restoration ecology. The six sites discussed in this chapter fall within National Park Service jurisdiction in the states of California and Nevada (Figure 2).

The Mojave Desert is a winter precipitation desert bordered by the Great Basin and the Sonoran Deserts (Brooks & Pyke 2002) and contains plants from both deserts along with its own distinct flora (Beatley 1973). The most rain falls in cool months (October to April, but more importantly, October to January) and the rest falls during the summer monsoon season from about July to September (Rowlands 1982). The amount of winter rainfall decreases from west to east, with Las Vegas having about 60% of its precipitation fall during the winter months (Rowlands 1982). Precipitation, in both amount and timing, is considered the largest driver of arid plant cover (Kimball et al. 2015).

The Mojave Desert hosts a variety of plant communities that change with elevation, precipitation, aspect, and soil type. As commonly seen in other biomes, precipitation increases and temperature decreases with increased elevations (Beatley 1975). With these changes, different plant communities co-occur, often with broad ecotones. Geographically, precipitation and plant community development patterns change when one moves from the southwest to the northeast. Communities in this study range on a gradient from the creosote-bursage community at the lower limit to the blackbrush and Joshua tree woodland community at the upper limit and remains below the juniper-Pinyon tree line. This study also ranges on a geographical gradient: the eastern Mojave, which is least dependent on winter precipitation, to the south-central Mojave, which has less equitable precipitation patterns (64% winter precipitation) (Walker & Landau 2018).

The largest portion of Mojave Desert vegetation is the creosote-bursage community. This community covers nearly 70% of the Mojave (Lathrop & Rowlands 1983). The community occurs on valley floors in well-draining alluvial flats and gentle slopes below 1500 m (Brooks et al. 2007; Thompson 2004). Woody shrub cover is often low and minimal at around 5-30% (Vasek & Barbour 1995). When precipitation starts to exceed 183 mm, an ecotone begins and the desert scrubland community is gradually replaced with other plant assemblages, especially the blackbrush community (Beatley 1975). The codominant species and namesakes of the community are the creosote bush (*Larrea tridentata)* and white bursage *(Ambrosia dumosa).* Other commonly occurring species include prickly pear cactus (*Opuntia* spp.), goldenheads

(*Acamptopappus sphaerocephalus),* Mormon tea (*Ephedra* spp.), California buckwheat (*Eriogonum fasciculatum*), rhatany (*Krameria* spp.), winterfat (*Krascheninnikovia lanata),* boxthorn (*Lycium* spp.), indigobush (*Psorothamnus* spp.), needle grass (*Achnatherum* spp.), Galleta grass (*Hilaria* spp.), and chollas (*Cylindropuntia* spp.) (Brooks et al. 2007). Common invasive plants include red brome (*Bromus rubens*), cheatgrass (*Bromus tectorum*), and filaree (*Erodium cicutarum)* (Brooks et al. 2007). Because of the community's vast range in both geography and precipitation, floristic species richness greatly exceeds this small listed portion of plants.

A conspicuous feature of the creosote-bursage community is the accompanying old layer of desert pavement and desert varnish. Desert pavements cover around 50% of natural arid lands in North America, although clast size and mosaic tightness vary greatly (Musick 1975; Quade 2001). Often, they are associated with alluvial fans, valley floors, and other ancient aqueous remnant landscapes (Quade 2001). Within the Mojave Desert, they are typically in the low-lying creosote-bursage community and tend not to form in blackbrush areas or above (Quade 2001). The mosaic of shapes affects hydrology and, therefore, the vegetation. Pavement alters the soil morphology, texture, and leaching depths of salts (Wood et al. 2005). It creates a nearly impenetrable surface for water infiltration, or at least very slow infiltration. The overflow water is then directed toward certain areas with the obvious consequence of affecting the flora's ability to root into soil, germinate, and garner ample survival supplies. Desert pavements contribute to a dynamic and spatially heterogeneous landscape with varying shrub distribution (Musick 1975; Wood et al. 2005).

At the upper limits of the creosote-bursage community, the blackbrush community begins with a gradual ecotone, between 5-8 km (Beatley 1975). The blackbrush community is often found where average rainfall reaches 160 mm, with the highest percentage of shrub cover and best development occurring at elevations between 1100 and 2000 m (Beatley 1975: Thompson 2004). The community is dominated by its namesake, blackbrush (*Coleogyne ramosissima),* which can be 90-95% of the total plant cover but decreasing at its upper and lower limits (Brooks et al. 2007). Beatley (1975) found highest shrub cover percentages to be around 37-51%. The literature varies according to percent shrub cover, possibly because of geography and other microsite effects. Other plant associations, especially near ecotones, include creosote (*L. tridentata),* juniper trees *(Juniper* spp.), desert almond (*Prunus fasciculata)*, boxthorn (*Lycium andersonii)*, bladder sage (*Salazaria neomexicana)*, Joshua tree (*Yucca brevifolia)*, needle grass (*Achnatherum* spp.), and Galleta (*Pleuraphis* spp.). Common invasive plants include red brome (*Bromus rubens),* cheatgrass (*Bromus tectorum*), and filaree (*Erodium cicutarum)* (Brooks et al. 2007). The blackbrush community can occur on its own or as an understory layer for other woodland communities.

The Joshua tree woodland community occurs at mid to high elevations below the Pinyon-juniper tree line. Common species include Joshua trees (*Yucca brevifolia*), Mojave yucca (*Yucca schidigera*), and banana yucca (*Yucca baccata*). Because of its intermediate range, it is often found co-occurring with either characteristically blackbrush or creosote communities.

Also occurring within the mosaic of lower-elevation arid plant communities are those based entirely off soil characteristics rather than elevation. Gypsum soils, found frequently in the Mojave, host a unique array of plants, many of which are endemic, and unique edaphic properties that confer a landscape appearance that is almost extraterrestrial. The ecotone between the gypsum community and the adjacent communities often appears abruptly and then shades of reds, pink, rose, and cream meld together and the crystalline gypsum sparkles in the sun.

Relatively high concentrations of gypsum cause the formation a physical abiotic crust on the surface, limiting water infiltration and seed burial (Belnap & Eldridge 2001; Boyadgiev 1974). Even higher on this side of the spectrum, soils with a very high relative gypsum content more efficiently absorb water and sustain biological soil crust communities. While literature results tend to be inconclusive, biological crusts play important roles that should increase the potential of a seed to reach maturity (Chiquoine 2012). Such vast varieties of factors contribute to the unique behavior and appearance of each individual gypsum site. Because gypsiferous soils are very low in organic content and key plant macronutrients, communities are typically sparsely vegetated; at least a third of the density found on alluvial fans (Meyers 1986). While Meyers (1986) found that creosote-bursage still provided the dominant cover even on gypsum soils in the eastern Mojave Desert, there was abundance of gypsophilic and gypsoclinal species. Certain plant species that inhabit the area more regularly than others can withstand the alkaline soil: *Atriplex* spp., *Suaeda nigra, Petalonyx parishii, Lepidium fremontii, Enceliopsis argophylla, Psorothamnus fremontii,* and *Phacelia palmerii* are a few within Lake Mead National Recreation Area. The Las Vegas bearpoppy (*Arctomecon californicus*), an endangered endemic, is found on Gypsiferous soils in the region near the two gypsum study sites. In addition to little vegetation, soil texture and stability, the abundance of biological soil crusts very slowly enhance the microtopography of the soil surface (Belnap & Eldridge 2001).

Anthropogenic disturbances vary greatly in type and severity; however, all degrade, or in many cases, destroy sensitive arid ecosystems. Disturbances negatively affect soil properties and remove perennial plant cover, which hampers wildlife and creates dust and air pollution that persists for many years (Abella 2009; Abella 2010; Prose et al. 1987). While severity by disturbance type is not specifically quantified in this study, it is important to understand how different disturbances affect the lands. Further interesting experiments could quantify the severity of disturbance and subsequent restoration on specific plant life history measurements. The study sites host several different disturbances: mining, recreational disturbances, and construction-related disturbances including pipeline corridors, underground fiber optic lines, and road construction or rerouting.

Recreation disturbances include off-highway vehicle (OHV) trails, old picnic sites and campgrounds, and dirt roads. Off-highway vehicles have been a popular form of recreation for decades. While many trails exist that allow their use, the vehicles often veer from established trails, causing environmental degradation. Negative impacts include destroying soil-stabilizing biocrusts, soil compaction, reduced rates of water infiltration, increased erosion rates, and vegetation destruction (Lovich & Bainbridge 1999). Trails and dirt roads channelize water, degrading natural hydrology with adverse consequences. One example is the channelized water no longer moves as overland flow, bypassing plant clumps that historically would have received a relatively predictable supply of water. These plant clumps are fertile islands that provide habitat for plants and animals and enriched soil conditions. Although OHV trails have become regulated since the 1990s, more miles of OHV trails than paved roads exist in the Mojave (Walker & Landau 2018).With vehicular disturbance, soil compaction occurs swiftly and greatly limits plants' abilities to root into the soil with the unnaturally strong soil strength. Annual cover can be seriously reduced with just one pass by a 2190 kg vehicle on wet soil (Adams et al. 1982). Recreational disturbances like these tend to be thin and superficial, but severe. The seed bank underneath is likely still viable and there is a ready supply of nearby plants to colonize the area, assuming they can actually penetrate the compacted soil. Roads alter the landscape geology and speed up erosion, further perturbing natural hydrology (Walker & Landau 2018). With an increasing number of residents who desire nearby recreational opportunities, and subsequent increase in the number of roads and trails, invasive plants have also proliferated throughout the desert.

Mining damages the Mojave Desert considerably because of its high resource requirements and the oftentimes toxic components associated with extraction of ore. Within California deserts, mining has been important since the 1880’s, and remains so today (Lovich & Bainbridge 1999). In addition, the older the mine, the less likely it was regulated in terms of environmental damage. Toxic components, such as cyanide, can cause animal deaths and leak into waterways. The stripping of a mountain ca n completely destroy the vegetation and surface soil biota and soil profile integrity.

Increasing human populations in the Mojave have necessitated development and infrastructure. The construction of pipeline corridors, cell phone towers, transmission lines, and roadway construction often require the removal of the topsoil surface layer and vegetation. Roadway construction severely compacts soil. Webb and Wilshire (1980) found that after 51 years, long-lived perennials such as creosote colonized uncompacted land at 40%, while compacted areas were only 3% colonized. Construction typically results in complete habitat destruction with secondary effects of habitat fragmentation, reduced gene flow, and access to remote areas for illegal collections (Lovich & Bainbridge 1999). Pipeline corridors, despite more recent projects implementing environmental measures, still greatly affect ecosystems because of the extensive trenching required. Negative impacts include churning soils, disturbing biocrusts and rock surfaces, and concentrating runoff and erosion (Lovich & Bainbridge 1999).

Despite the differences of each disturbance type, they share similar characteristics of negative soil impacts and decreased vegetation. Anthropogenic influences are more controllable than the influence of climate change; without regulating these factors as they occur and actively restoring past disturbances, the Mojave Desert will soon become a drastically altered landscape.

I focused on three common restoration treatments in the above disturbances and plant communities for a total of six sites, detailed in Table 1 and depicted in Figure 3. The restoration treatments are first organized into abiotic or biotic treatments. Biotic treatments include seeding and outplanting. Outplanting includes transplants and nursery-grown stock. Abiotic treatments comprise the second category. These treatments include any manipulation to the soil and vertical mulching, which is propping up dead plant matter buried partially into the ground. While vertical mulching could verily be considered a biotic treatment, I decided that the current biotic treatment category includes methods that can quite quickly alter the environment around them during growth. The biotic category provides nutrients and inputs a continuous source of organic material. Vertical mulch would eventually confer organic material back to the soil, but very slowly and at a loss to the original structure itself. High arid decomposition rates depend on favorable moisture, temperature, and microbial activity (Klemmedson 2009). Microbial activity is a function of recent temperature and precipitation patterns. Partly because the Mojave experiences dry and wet cycles, litter decomposition rates vary. In addition to temporal variation, rates may also change because of spatial variability to water and resources; otherwise, decomposition rates simply vary inter-specifically (Gaxiola & Armesto 2015; Klemmedson 2009). Biotic decomposition may halt entirely if conditions are too dry for microbial breakdown of organic material and depending on the species, may use photodecomposition as the main method for nutrient cycling (Klemmedson 2009). Evergreen species more frequently utilized biotic decomposition: during heavy rains the species completed this process more quickly than its abiotic counterparts, but with les predictability (Gaxiola & Armesto 2015). Regardless, both types of decomposition are so slow that the potential biotic treatment conferred would be close to nil compare to abiotic decomposition. The vertical mulch is also acting like a nurse plant to induce the fertile island effect. While the contribution of organic matter is an important aspect of the effect, so is its role in obstructing harsh conditions that would destroy sensitive or young plants. The physical vertical mulch structure acts as a wind barrier allowing seed and organic matter accumulation at its base; the structure provides shade hence it regulates soil moisture; and the structure protects sensitive plants from possible herbivory or stray hiking feet (Berg & Steinberger 2012). To perform these functions the structure does not have to still be living. Furthermore, Berg and Steinberger (2012) used artificial plastic shrubs as live plant replacements and found that the plastic shrubs were still regulating the annual plant community. The abiotic treatments tended to focus most directly on hydrology and soil properties. Vertical mulch falls into this category more clearly.

After categorizing restoration treatments on biotic or abiotic, I further divide them into their specific treatment and subtreatments included (see Table 1). Biotic components, seeding and outplanting, can have many nuances and synergistic treatments depending on the specific project protocol. Seeding has been a common practice of restoration ecology because of its implementation ease, large area coverage, and relatively low costs. The technique covers a much bigger area than outplanting possibly could. There are a variety of methods including seed balls, hydroseeding, hand-broadcast seeding, and aerial seeding from airplanes or helicopters. Hand-broadcast is efficient in small areas, but for large disturbances, aerial seeding provides a better option (Abella 2010). Seeding success rates vary greatly between studies in arid ecosystems because seeding is easily affected by factors like herbivory, granivory, and infrequent or unpredictable precipitation events (Abella et al. 2012; Bean 2004). Because of precipitation infrequencies, Bean (2004) concluded that seeding works only one out of every ten attempts. Nevertheless, there have also been many successful seeding attempts, including developing better methods of seeding. For example, seed balls are an amalgamation of seeds, clay, and compost. Seed balls decrease herbivory damage. During a rain event, the seeds and nutrients are released because of the favorably wet conditions for seed germination (Walker & Landau 2018). Hydroseeding, or mixing seeds with organic slush that is sprayed out by a gas-powered hosed machine, is gaining serious momentum as a practice. Seeding, unfortunately, has increased in cost nearly 600% from the 1980s and 1990s to the 2000s (Copeland et al. 2017). To recalibrate cost-effectiveness, more information regarding seeding success is needed. For example, there is potential to increase its success by pairing it with favorably forecasted precipitation years (Copeland et al. 2017; Hardegree et al. 2017; Kimball et al. 2015). Alternatively, trying the same technique in multiple years may increase the chances that the seeds are germinated by at least one favorable year. Careful planning helps, but luck seems to play a role in arid climates presently. Depending on which section of the Mojave Desert is being restored, land managers can predict by accounting for expected precipitation patterns. For example, pairing restoration with El Niño years, or outplanting during the wet season in October-April (Hereford et al. 2016).

Outplanting is beneficial in that it can immediately establish long-lived perennials, assuming their survivorship. While a high survivorship is far from guaranteed, the addition of irrigation, shelters, or cages to protect from herbivory are useful (Abella 2009; Abella et al. 2012). Native plants can be grown in nurseries from seed, salvaged from areas that will be disturbed in the near future and transplanted from healthier and proximal areas, or as cuttings. Cuttings often work well for cacti. The disadvantages of outplanting involve a cost-benefit analysis. Outplanting nursery-grown plants incurs extra costs in their maintenance and resource use. Transport and labor costs can be high. Unpredictable weather patterns and herbivory may lead to meagre survival rates. Examples of poor revegetation attempts include a 1983 highway-outplanting project where after two years, almost all of the plants perished; another study by Brum and others (1983) reported a 0.3% and a 26% survival of transplanted seedlings (Lovich & Bainbridge 1999). Romney and others conducted a successful attempt in 1979 with an 80% survival rate (Lovich & Bainbridge 1999). The first two years for most outplants are critical; most mortality of creosote seedlings occurs within this timeframe (Ackerman 1979). Oftentimes, irrigation and herbivory protection are ceased after the initial critical period, although this depends on resources and funding.

Soil conditioning is any alteration to the surface of the soil. Conditioning has many different practices, which in some way or another, help with seed catchment and plant establishment, and positively alter the hydrology or quality of the soil. Soil manipulation for the purpose of seed catchment is especially helpful considering it does not rely heavily on precipitation patterns as biotic treatments do. It does rely partly on the assumption that a native seed bank is present and viable. These treatments are designed capture more seeds that may have otherwise been blown away by intense desert winds and flash flooding.

Included in this study are the techniques of vertical mulch installation, ground-ripping to form furrows or pits, soil replacement, land recontouring, and decompaction. Vertical mulching is planting large, dead limbs in an upright manner to cheaply attempt to reproduce what has been termed the “fertile island effect”, naturally performed by both living and dead shrubs in an undisturbed setting. Fertile islands harness organic matter and increase levels of essential nutrients for plant growth, including nitrogen and phosphorous (Bolling & Walker 2002). Fertile islands provide the water and shade necessary for seed catchment and seedling establishment. A study conducted by Thompson and others (2005) found that the fertile island effect occurs at all elevations, including the creosote and blackbrush communities. While soil moisture and organic matter increased by a factor of 1.5 from low to high elevation sites, the ratio of different shrub species to interspace concentrations did not change; therefore suggesting that the ratios are independent of species type (Thompson et al. 2005). As a part of restoration, it is thought that these islands may represent an ecological nucleation where propagules can spread out from (Hulvey et al. 2017). Vertical mulching also provides secondary effects that contribute to the restoration process. Degraded areas, such as closed roads, are less visible with vertical much treatment (Abella 2012), which helps prevent illegal OHV use and off-trail hiking or other recreational activities.

Ripping the ground, often using heavy machinery, decompacts the soil and allows for better seed catchment. In addition, the decompacted soil will allow small roots to take hold. Because ground ripping required the use of heavy machinery, consideration must be given to the impacts of the land that the heavy machinery must drive through to actually get to the restoration area.

The addition or replacement of surface soil and recontouring of a site positively alters the hydrology of the land by directing rainfall towards the restoration treatment (Abella 2009). Because natural plant growth and establishment is already a slow process, remediating compacted soil through decompaction will benefit plants. Decompaction increases the ability of soil to absorb water and allow plants to establish roots in the softer soil (Lovich & Bainbridge 1999). Decompaction can be accomplished by shoveling soil loose or using heavy machinery. However, the use of heavy machinery further harms the area directly driven over, potentially neutralizing the positive effects.

Because many soil conditioning techniques decrease soil density and restore a more natural hydrology, these procedures may lead to increasingly successful outplanting or seeding treatment. Water can be directef towards new outplants or seeded areas. The looser soil is more conducive to root development and growth. Vegetation and soil conditioning treatments have increased 750% from the 1980s and 1990's to the 2000's (Copeland et al. 2017). Most likely, implementing several restoration techniques will be the surest way to a successful revegetation project. Each technique is inherent with individual weaknesses that can possibly be mitigated by another technique. However, all of these abiotic techniques serve to stabilize substrate and build up soil, which are considered common goals of restoration (Heneghan et al. 2008; Webb & Wilshire 1980).

I selected a series of six studies to assess or reassess past restoration methods. With these sites, I attempted to address the following questions:

1. What are the effects of restoration treatment?
2. Do different types of restoration lead the trajectory of treatment sites closer towards the be?
3. Do biotic treatments (seeding, outplanting, transplanting) enhance the recovery trajectory?
4. Do abiotic treatments (soil manipulation, vertical mulch) enhance the recovery trajectory?
5. Does the slew of variables measured in the past years' data and my current data sufficiently substantiate the health of the site, especially in regards to the SER 2004 Primer attributes?

From these questions, I hope to find similarities among the different sites that may indicate to what extent particular restoration techniques work or fail in the eastern Mojave Desert.

2.4 Methods

Synthesizing sites provided for a unique challenge because of the various factors that influence the environments, murky histories of sites, and unknown consequences of multiple years of monitoring. Despite these unknowns, I attempted to decipher certain trends. The key word is trend, because it indicates directionality and therefore, a trajectory. By measuring different variables against each other and against the static success parameters quantitatively to their reference, as suggested by the SER 2004 Primer, these resulting "effects" can be compared among all eastern Mojave Desert sites, and perhaps beyond.

2.4a Selection Criteria and Field Study Area Description

All six sites were typified by low-elevation Mojave Desert shrubland plant communities, very specifically below the juniper-Pinyon tree woodline, between 375 and 1526 m. Sites characterized by Mojave-Sonoran desert and juniper-Pinyon woodline geographical and elevational ecotones, respectively, were immediately omitted for consideration. Three of the sites are in eastern California and three of the sites are in southern Nevada. Sites occupy the National Park Service units of Lake Mead National Recreation Area (three sites), Mojave National Preserve (two sites), and Joshua Tree National Park (one site).

At each site following a disturbance, various combinations of biotic and abiotic restoration activities (Table 2) were applied to only a portion of the disturbed area. This partial application provided the opportunity to assess restoration effectiveness because treated areas (henceforth, treatment) could then be compared to its disturbed, unrestored (henceforth, unrestored) and a proximal area representing the landscape if the disturbance had not occurred (henceforth, reference). Sites chosen may have multiple treatments. The individual treatments were not considered as a distinct treatment to monitor unless the treatment in question was applied alone in some area of the restoration site. For example, Keys View Road had necessary recontouring related to the road widening, but is not considered a treatment because it occurred in all other treatment types; Morningstar mine had recontouring, seeding, and transplanting treatments, but all are considered under the umbrella of a single treatment because they were applied to the same area and are thus indistinguishable.

In the spring of 2018, I scouted a series of sites to determine suitability to this study. The sites had to have ample background information (e.g. maps, pictures, and data) available to definitively determine where exactly the restoration occurred. Out of around eighteen sites, six were chosen for monitoring (Table 2).

Sites were not based on the time, duration, or type of disturbance in any way, although studying specific disturbance types (fires, linear disturbances) is becoming an increasingly popular concept in the current literature. The applied restoration age had to exceed at least five years in order to be considered long-term. Five years was chosen based on the average time a site is monitored after restoration treatment (Ruiz-Jaen & Aide 2005). In fact, the youngest restoration site was finished about a decade ago. Restoration dates do not exceed two decades in any of the sites.

2.4b Experimental Set Up and Monitoring Protocol

At each site, restoration (treatment), non-treated (unrestored), and undisturbed (reference) plots were relocated using a GPS in cases of previous years' monitoring (four sites) or installed in cases of no preexisting monitoring (two sites) in spring 2018. Within the two new sites, restored and treatment plots were determined visually and by written information based off Department of the Interior maps and reports with certainty. Reference plots were determined to be out of the restoration area by maps. In addition, it is visually apparent in many cases that they did not receive the same disturbance as the other plot types. It is not always possible to determine whether reference plots have been degraded in another time and place, however they represent the potential successional community of the area. For sites that have not been measured before, a standard design was used with six 1 m × 1 m subplots for each 100 m^2 plot to scale up to the whole plot level. Ten plots were established for the unrestored, the reference, and the treatment(s) plots for a total of 30 plots in each site, assuming they are not following a previous monitoring protocol. A subset of four plots per plot type was randomly chosen for monitoring in spring 2019.

Two sites with preexisting monitoring were measured in 2018 because there was enough information at the start of the monitoring season to proceed with certainty. The three plot types per all six sites were then assessed in spring 2019. Site-specific methods, study designs, and results are presented in Appendix 1. Methods for previously monitored sites follow the original monitoring protocol for clarity across years.

In each site, a 1-m^2 or a 0.25-m^2 quadrat was used to create a varying number of subplots in the main plot following a nested design. In each subplot, every species was identified and given a percent cover class (classes change based on project and modified from Peet et al. 1998). Species that were not captured in subplots but were present in the whole plot were recorded and given a cover class based on the whole plot. The average of the cover class was the ultimate percent cover for that particular species. These measurements also allowed for the calculation of species richness. All perennials were counted to record density. Seedling and mature perennials were not distinguished, so counts represent every individual, despite known and varying survivorship among seedlings.

Each individual plot type was compared to its control. A ratio was calculated to determine its similarity to the reference and to the baseline plots. These ratios allowed different sites to be quantitatively, or at least qualitatively, compared with one another; thereby, assessing the restoration treatment itself in terms of "effect size," while trying to limit other confounding factors inherent in choosing different geographical locations. With the inclusion of unrestored plots, both of the "tail end" effect sizes could be measured (e.g. change from unrestored to current, change from current to reference). Other site and plot level characteristics, which may affect the recovery trajectory taken were: aspect, slope, and 0-3 years of climate data pre- and post-restoration. In addition, precipitation and temperature data were compiled from the NOAA database for each monitoring year.

A meta-analysis framework was used to find patterns across the broad scales of all variables. In sub-analyses, other methods were employed depending on the suitability of the data. Patterns were then compared to SER attributes and other characteristics that the sites may have in common.

2.4c Data analyses

For each project site, treatments were compared to both of their respective reference and unrestored sites. Data analyses were designed on a project-to-project basis and can be found in Appendix 1. A multivariate analysis in trends with sub-multivariates was utilized using SAS.

Survey results for each site were quantitatively and qualitatively assessed, and qualitatively compared to each other. Qualitative analyses allowed for possible conclusion to be drawn among sites with differing characteristic that may impact their effectiveness in order to ascertain whether "restoration" itself is effective in arid ecosystems. Data results considered significant ($p \leq 0.05$) or moderately significant ($p \leq 0.10$) across sites are visualized in tables that indicate the direction of restoration effect and in figures that gauge that effect in each site.

2.4d Climate data

Climate data was obtained from the National Oceanic and Atmospheric Administration's National Centers for Environmental Information website (ncdc.noaa.gov). Weather stations closest to the site were averaged for precipitation (mean cm year^{-1}), maximum average temperature per month and year, and average minimum temperature per month and year. The amount of stations averaged depended on the data availability of each station within a reasonable enough distance and elevation of the site. For 2019 climate data, annual averages only span from January to September. All six sites were located in the eastern or east-central Mojave Desert. According to Walker and Landau (2018), five of the six sites were located in the Eastern Mojave ecoregion and the sixth site was located in the South-Central Mojave ecoregion. The further east in the Mojave, the more bimodal the winter precipitation-summer monsoonal cycle becomes, consequently most field sites are expected to most likely have a majority percentage of precipitation fall in the winter but also an important portion during the monsoons.

2.5 Results

2.5a A Note on Figures

This thesis focuses on 16 commonly measured variables. Among these metrics, 25% involved annual plant communities only. Twelve perennial measurements, including density, species richness, and cover of three common life history groups were analyzed. There are three biotic-only sites (simply referred to as ‘biotic’), one abiotic-only site (referred to as ‘abiotic’), and three abiotic-biotic combination sites. From this, out of the six total sites, five of them were treated with at least a biotic element (referred to as ‘total biotic’) and three of the six sites were treated with at least an abiotic element (referred to as ‘total abiotic’).

Variables in which the most sites showed a positive restoration reaction was perennial density, and shrub density, richness, and cover. There seems to be no unambiguous and consistent indication as to whether biotic or abiotic treatments helped increase more than the others did with these measurements.

The three types of figures represented in this section each indicate a small part of restoration effect. Figures 4-8 A and C (disturbed, reference) show how each site scored with the common parameters in the 2019 monitoring season. Each number is a ratio of either the mean percent cover or mean species richness of a certain metric within disturbed plots divided by the reference plot mean. Each ratio consists of the mean of the disturbed, treated plots and the mean of the disturbed, unrestored plots from the mean of its reference at both ends. A solid circle represents treatment plots and a circle outline represents unrestored plots. Each point lies a certain distance away from the reference conditions, which is set at x=1.0 and is denoted by a dotted gray line. Circles that lie on the right side of the grey line indicate higher values and circles on the left side indicate lower values than the reference. The particular type of disturbed plot is very similar to reference conditions if the circle lies near the gray dotted reference line, thereby making their ratios close to equivalent.

Figures 4-8 B and D immediately next to the first figures show the ratios of restored (treatment) to unrestored plot types in the 2019 monitoring year. All data were drawn from 2019 monitoring only. The x-axis is the ratio of the mean of unrestored plots divided by the mean of restored plots. A square that lies close to x=1.0 indicate that there is little to no difference between restored and unrestored plots and thus a change has not occurred. Unlike graphs A and C where the gray dotted like represents reference conditions, the gray line in B and D simply indicates equivalent 2019 means of unrestored and restored plot types. In contrast, high ratio numbers indicate large differences in means between the plot types. Positive, whole numbers indicate that the restored plots outperformed unrestored plots. Lower ratios show that unrestored plots outperformed treatment plots.

The label of the y-axis in both figure sets designates the year that the restoration project was completed. We expect restoration projects to take a little bit of time before they start indicating an altered trajectory as the community shifts in response to the induced treatment effects. Therefore, it is possible that newer restoration sites may show less stability than older sites.

Not only is the ratio number itself important but also important is which type of disturbed plot is more like the reference conditions. Can any indications of age stability be noticed despite differences in treatment? For both figure types, presence on the graph does not necessarily indicate a significant Type III Test. Significance ($p \leq 0.05$) and moderate significance ($p \leq 0.10$) are shown in Table 3. All complete Type III Tests are available for review in Appendix 1.

In Table 3 following the figures, the variables are measured by their treatment effect size throughout all years, where applicable. If effect tests indicated that treatment plots are significantly growing more similar to the reference than unrestored plots, the category received an up arrow (↑) indicating positive restoration effects. If restoration sites were not significantly influencing restoration success in a positive direction, the category received a down arrow (↓). In some instances, especially sites with multiple biotic and abiotic treatments like Keys View Road, different treatments (e.g. the outplanting versus the vertical mulch) could be parsed out. If significant, I was able to observe exactly which treatments are causing the positive, deleterious, or null effects. If a test in the site was not significant by either treatment or treatment × year interactions, its category received a dash (-). If the metric was not measured or the life history group was not present, "n/a" was inputted. Despite a non- significant dash, it is possible that the variable was significant by year only. With year as the only significant factor, it would be challenging to determine if treatment is having any effect at this time. Variables that were significant by year only are listed in Table 3. It should also be noted that for tests involving exotics only, an up arrow would indicate a detrimental result rather than a positive one. With the first two figures and table together, an overall storyline of the restoration sites' beginnings emerge. It confers the capability to see whether restoration had, if any, a positive or negative effect over time when applicable.

2.5b Test Results with Reference Parameter Comparisons

In Figure 4A describing native annual cover, half of the sites had treatment plots that were more different from reference plots than the unrestored plots were in 2019; however, the effects move in both directions and indicate opposite results between sites. In two of the sites, restoration substantially increased native annual cover to levels exceeding reference conditions.

In another two of the sites, it appears that restoration decreased native cover. Both of these sites did have biotic treatments. One of these, Keys View Road, included both abiotic and biotic treatments. In fact, these restoration plots had the worst native cover percentages than any other plot type of any other type, including its own. According to differences of square means tests for all years combined, outplanting seems to pull the means up or vertical mulch is pulling them down.

Three sites had unrestored plots that were more different from the reference than the treatment plots were from the reference. One of these sites (biotic treatment) had higher cover in the unrestored plots than even the reference. Two sites, with biotic treatments at least, had both disturbed plot types at lower native annual cover percentages than the reference (Figure 4A). One of these same sites, a biotic/abiotic, and another abiotic site had greater native cover in the unrestored plots compared to the treatment plots (Figure 4B). The other four showed restored to unrestored ratios greater than x=1.0. These results suggest that restoration helped increase native annual cover, but because of the bidirectionality between sites, does not show a clear relationship that the conditions are returning closer to the reference trajectory especially with only the inclusion of 2019 monitoring data. In tandem with these slightly positive reactions, the magnitude of the ratios started negating these positive results. When considering only 2019 in Figures 4B, the disturbed plot ratios are very close to one, indicating nearly equivalent ratios, or poor effect detection.

According to Table 3, which includes every monitoring year, native annual cover significantly changed (increased) among most sites, but with little semblance regarding understanding exactly why. When omitting the two sites that had only a single monitoring year, two sites increased, one site decreased, and one site exhibited both an increase and decrease.

Native annual cover did not show strong effects relating to the treatment received, either, however an increase trend is vaguely apparent. Total biotic sites increased or were not significant. Total abiotic sites were split evenly between positive, negative, and no effects. The two sites with a single monitoring year, Morningstar Mine and the Fiber Optic Cable, did not exhibit detectable effects from their biotic treatments. The Fish Hatchery, Northshore Road, and Keys View Road had positive effects, but Keys View Road exhibited both an increase and a decrease. Upon parsing out the three separate restoration treatments Keys View Road received, I found that the treatments including any type of abiotic vertical mulch treatment brought positive results down. This does help explain, in part, the negative ratios found in Figure 4A and B. I was unable to determine which treatment type was conferring positive results to Northshore Road. It appears that restoration might have an effect on native annual cover, but it is not possible to conclude with certainty what that effect is among these sites and years. Sites are simply too varied to detect effects; however, with time, cover could conceivably increase. Biotic restoration seems to be more beneficial, especially considering Keys View Road, but the unexplainable boom in cover at Road 108 complicates this.

During 2019 monitoring, native annual species richness of treatment plots was more dissimilar to reference plots than the unrestored plots were to the reference plots in four sites (Figure 4C). Two of these four treatment sites included abiotic treatments and showed greater richness than even the reference plots. The other half had treatment plots that were substantially worse than the unrestored and the reference plots. In three sites among the six, treatment and unrestored plots had not yet matched the richness of annuals found in the reference areas. Straying from this trend, Keys View Road and Road 108 have annual richness values in treatment plots that far surpass the number of species found in reference. Road 108 has annual treatment cover and richness that is nearly six times that of the unrestored plots according to Figure 4B and D, exhibiting the results that I would have expected in Figure 4A and B. Namely, I hypothesized that metrics should be increasing from unrestored to treatment to reference plots. Three of the sites did weakly exhibit difference between the restored and unrestored plots. All three had received biotic treatments. Despite the disturbed Northshore Road plots not yet reaching reference-level native annual richness, all three sites that have had abiotic treatments appear to show more differences between restored and unrestored plots. This is, however, a mixed result among the six field sites. Among all years, the two largest sites, Keys View Road and Northshore Road, increased in native annual richness (Table 3). A surprisingly low number of sites changed across years, especially among those biotically treated.

When considering the final year of monitoring data only, four sites had treatment plots that were more dissimilar in exotic annual cover than unrestored plots to the reference (Figure 5A and B). Three out of these four sites exhibited a very small decrease in total exotic cover after restoration. All three of these sites received a biotic outplanting treatment. Within these sites, Keys View Road, which received vertical mulching and a vertical mulching × outplanting combination pooled to procure the ratios, is included. Road 108, which had abiotic ripping only, was the only site where treatment plots had substantially more exotic cover than the reference; similarly to native annual metric, it maintains outlier-like conditions. Road 108 treatment plots have the only positive disturbed to reference ratio (Figure 5A). In the situations of Morningstar Mine and Northshore Road, the unrestored plots had lower exotic cover than the reference. At the Northshore Road site, especially, the unrestored and restored plots were extremely different (Figure 5B). The treatment sites at Northshore most closely matched the reference sites. Cover ranged extensively between sites: the highest cover-by-plot type averages were found in Fiber

Optic Cable followed by Keys View Road and the lowest was in Road 108. Including all monitoring years, sites vary to an extent that no trend was easily or confidently observed (Table 3). When examining Table 3 and Figure 5B closely, it is possible to speculate that treatments effect total biotic sites with time. This is demonstrated in Table 3 and to a lesser extent because of the single year, in Figure 5B. There seems to be little indication that exotic annual cover changes over time or treatment (Figure 5A and B; Table 3).

Regarding 2019 exotic species richness, treatment plots were less similar to reference plots than unrestored plots by a small amount in four of the six sites (Figure 5C). Of these sites, 75% had greater exotic richness than the reference, and the site with less had unrestored values equaling reference values. One site had both disturbed plot types equally higher than the reference plots. Generally, disturbed plots had ratios close to one, implicating restoration as a minimal impact solution to exotic annual species richness (Figure 5D). This is especially of biotic sites: every single biotic-only site had ratios closest to one, followed by biotic/abiotic sites. One site, Road 108, that had extremely disparate ratios, was negatively impacted by the alteration of its microtopography from ripping treatments. For all plot types in all sites, exotic diversity ranged from 0.28 to 4.1 species in a plot. There was an average of 2.4 species plot$_{-1}$ in treatment plots, 1.9 species in unrestored plots, and 2.0 species in reference plots. Results were typically insignificant when assessed with multiple years (Table 3), suggesting that while exotics can be extremely detrimental, richness is perhaps not the best metric to use when determining exotic invasion. With such a small species pool, results should not be construed as gospel.

In general, the four annual variables measured: native and exotic percent cover and species richness remain similar in their reactions. Regarding native annuals, covers vary substantially and richness may be increased by combination treatments but is typically not very significant in the first place. Exotic annuals tended to have higher covers in reference areas but higher richness in disturbed (unrestored and restored) areas. Exotic measurement means tended to increase near high-traffic areas and across larger disturbances, aligning with species-area curve theory. The maximum number of species on a plot type was around four. Some trends emerge when considering annuals regardless of nativity. Across multiple years exotic tests and native tests displayed similar trends. Cover changed more frequently than richness, which tended to remain relatively constant. It is very difficult to discern treatment effects in this scenario, but the trends that are possibly present are: a. Treatment and reference had more cover over time than unrestored areas and b. Richness was higher in the treatment plots compared to the unrestored and reference areas.

Perennial plants provide not only the solution to a highly disturbed area, but also the metric in which to measure ecosystem health and recovery. Analytical focus on perennials was much more specific than with the annuals community because of their concrete socioeconomic value across years and integral contributions to vegetative structure and function. Perennials were classified as graminoids, forbs, or shrubs (woody perennials). Cactus and yucca species were usually pooled within the shrub category because they provide a similar ecosystem function due to their large stature. In many sites, the perennials planted received some type of irrigation, although the exact details are difficult to interpret with the current available resources. Perennial cover, species richness, and density were analyzed.

Overall, increases in total perennial cover were the most common responses among sites; only Keys View Road actually decreased in cover (Table 3). Of the sites that increased, all had a biotic restoration component. In the three abiotic sites, perennial cover exhibited mixed results:

no effects at Road 108, an increase at Northshore Road, and a decrease at Keys View Road. The interesting contrasting effect involving vertical mulch is seen again at Keys View Road, like it was for annual metrics. When individual treatment effects were parsed out and investigated, vertical mulch tended to decrease cover when applied. This further substantiates that abiotic treatments at Keys View Road are hindering vegetation restoration. For Northshore Road, no effect involving the outplanting was detected, suggesting that the abiotic treatments actually does help in this case because perennial cover still increased among years. Despite the positive results of outplanting at other sites, Northshore Road consistently showed no significant effects because of outplanting regardless of the metric (Appendix 1). According to a majority of the above results, biotic treatments such as outplanting and seeding can increase perennial cover and abiotic treatments have mixed, if any, results.

Total perennial species richness across all years and sites showed statistical significance in three of six sites: biotic/abiotic Keys View Road and biotic Morningstar Mine both increased while abiotically treated Road 108 declined. When examining Keys View Road further, it does not appear that vertical mulch had any deleterious effects on perennial species richness, because outplanting and outplanting × vertical mulch treatment plots do well (compared to other plot types) in all years.

Perennial density was not measured at Keys View Road, so there are just three biotic treatment sites, one biotic and abiotic site, and one abiotic site. Despite some non- significant and negative results for perennial cover and species richness in earlier tests, perennial density increased at every site. At Northshore Road, the increase was only moderately significant.

The Fish Hatchery had no plants characterized as “perennial forbs” in all years and plots. In 2019, native perennial forb cover was typically found at levels higher than the reference

(Figure 6A). This occurred in all except one site: Keys View Road, where both the restored and unrestored values were similar and did not reach reference levels. Focusing on sites with treatment plots that had higher cover percentages than their reference, two of the sites somewhat exhibited hypothesized results: Morningstar Mine and Road 108 had unrestored values that were lower than their treatment values and increased through restoration, although treatment means ended up higher than reference means. Interestingly, unrestored values most closely match the reference percentages. Two sites, Northshore Road and the Fiber Optic Cable, had both types of disturbed plots surpass reference levels. In both cases, unrestored plots had far greater cover than either the reference or treatment plots (Figure 6A). Northshore appears to have a much larger restored to unrestored ratio as well (Figure 6B). However, cover was so low for perennial forbs in this area that the subsequent large ratio does not accurately represent the ecological results. Restored plots had a mean of 1.1%, unrestored plots had a mean of 0.7%, and reference plots had a mean that was minutely lower than unrestored plot percentages. The maximum 2019 mean was 2.5% and the 2019 average across all sites and plot types was just 0.75%. When all years are combined, perennial forb cover is significant and density moderately increased at Morningstar Mine and Northshore Road, both of which have biotic treatments (Table 3). Species considered perennial forbs were seeded at Morningstar Mine, and were also observed during 2019 monitoring.

Perennial forb species richness significantly increased at Keys View Road and Northshore Road over the monitoring years (Table 3). No other sites were significantly different by plot type or year × plot type interactions. Richness was, interestingly, higher in almost every disturbed plot at all sites regardless of restoration treatments (Figure 6C); this is of little consequence because most sites did not have more than two species. Northshore Road unrestored plots were shy of reference levels, but treatment plots exceeded reference levels. Including Northshore Road, three sites had treatment plots with higher perennial forb species richness than unrestored plots (Figure 6D). The sites fall into all treatment categories. Perennial forb density in all years was moderately significant higher at Morningstar Mine, where it was seeded, and Road 108 (Table 3). Overall, perennial forbs did not greatly contribute to the high perennial density numbers observed.

Perennial grasses, because of their minimal abundance, were an inconsequential part of the vegetation diversity, even within the reference areas. There were no perennial grasses at the Fiber Optic Cable and Road 108, and there was one species of perennial grass, *Dasyochloa pulchella,* at the Fish Hatchery. The highest species richness on any plot at any site is less than one per plot type. I observed that while there were some grasses scattered around the larger area of some sites, they were so infrequently encountered that it is plausible they did not appear in any plot. Their relative ratios are illustrated in Figure 7 A-D.

Shrubs, which include woody perennials and tall stature perennials such as *Yucca brevifolia* and *Y. schidigera* constitute the final component of perennial plants. A small handful of shrub species can occupy large swaths of land, thus it is unsurprisingly the largest and most important factor (Vasek 1979). Most sites had positive restoration effects in shrub cover, richness, and density. Shrub cover increased in all sites and all years except the Fish Hatchery and Road 108 (Table 3). It is not surprising that no effects were captured on Road 108; low shrub cover already characterizes the gypsum soil and no biotic—seed or plant-- component was added. The Fish Hatchery, however, had shrubs planted directly on to the site. Interestingly, despite not bein significant in plot type among all years, these two sites showed the most different disturbed to reference ratios of all sites in the 2019 monitoring year (Figure 8A). Keys

View Road again had the peculiar effect where it appears that either vertical mulch decreases cover, or outplanting increases cover, or both. In 2019, most sites had treatment and unrestored plots that had less shrub cover than their respective reference and most had unrestored plots with less shrub cover than restored plots (Figure 8A and B). Only one site had greater shrub cover in unrestored plots instead of restored plots. The site had a mix of both abiotic and biotic treatments.

Shrub species richness increased in all sites except Northshore Road and the Fiber Optic cable (Table 3). The Fish Hatchery showed a moderately significant increase. Typically, there was a greater number of shrub species in restored plots compared to unrestored plots (8D). There were also more species in reference plots than disturbed plots, except the Fish Hatchery and Road 108 (Figure 8C). Shrub density increased in all sites except Northshore Road (Table 3). According to these results, it does appear that restoration techniques help positively amplify shrub characteristics, but to what extent and by what treatment type remains largely unknown. It quite possibly depends more on year and precipitation patterns for juvenile shrub survival.

The age of a restoration site has been implicated in the past for how well it will perform. With these six sites, it does not appear that the age of the restoration project is a strong indicator of how the site will perform when tested for one of the sixteen metrics. The sites are still possibly too new, too varied.

2.6 Discussion

2.6a Measuring Success

Sites ranged in their responses to restoration by treatment and, if applicable, by year. From using quantitative variables like percent cover, density, and species richness, I was led to possibly identify qualitative connections between extremely unique sites. Tested variables involving annuals were heavily influenced by precipitation. Exotic cover tended to increase in more human-populated areas and more shrub-populated areas. Perennials in treatment plots were often more similar to the reference than unrestored plots. The question of whether restoration helps amend the successional trajectory of a highly disturbed and unrestored site to pristine reference conditions is for the most part, yes, but with important exceptions and especially subjective regarding the initial restoration goals.

In many of the sites, it is unclear whether long-term ecosystem processes were restored or altered. This requires more consistent and even longer time scales than included here. In this study, and among many other studies, vegetation structure, diversity, and abundance must be used as proxies to determine ecosystem function. Although empirical studies involving ecological process exist, they generally take more time, resources, and money (Ruiz-Jaen & Aide 2005; Wortley et al. 2013). Because I chose field sites that are older than typically considered when planning restoration surveys (Ruiz-Jaen & Aide 2005), each site could be further measured for ecological processes in addition to vegetation structure and diversity. For example, at Northshore Road, literature exists qualifying and quantifying the microbiological community interactions (Chiquoine 2016). At the Fish Hatchery, the Fiber Optic Cable, and Keys View Road, subplots were divided between interspaces and under shrubs. This can characterize the microcommunity development. Soil samples were collected at the Fish Hatchery to test for the pH level, total P and K, total C and N, and soil texture analyses. These analyses can substantiate claims about biological interactions from restoration and the fertile island effect.

The probability of answering these difficult questions on restoration effectiveness and actual restoration success is correlated to the amount of information published. In a research article studying restoration trends for a nearly eighty-year period by Copeland and others (2018), the authors substantiated that the “myriad questions regarding restoration practices and outcomes can be addressed by synthesizing standardized datasets which cover large spatial scales and long time periods…”. Future restoration ecology should be characterized by the idea that the more information available, the clearer the picture or goals, and the more successful the restoration project.

In sites with multiple restoration treatments, at least one element helped improve the recovery trajectory of the restored area. Whether this is considered “good” to land managers may have something to do with whether it was a cheap treatment, like soil manipulation, or an expensive treatment, like outplanting that failed to improve the area as well as the success rate. Regardless, in each site with multiple treatments, the use of at least one or both of the treatments appears to be an effective strategy, even a synergistic strategy. For example, at Morningstar Mine, perhaps the outplanting of smaller perennial shrubs allowed for the seeds of nurse-plant dependent species like *Y. brevifolia* to germinate and then eventually to moderately vary between plot type in 2019. Two sites included a biotic and an abiotic treatment together and apart. While the biotic treatment tended to be more successful, there is some evidence that certain metrics will increase with the use of both. Another example is that very diverse seed mixes, which one would expect to greatly increase diversity may fail due to competition from invasive grasses; thus, pairing a seeding with an aptly timed weed control effort may create a highly diverse and highly native area (Copeland et al. 2017).

Time since restoration was not as large of a factor as anticipated, or the effects were not felt with the amount of time, the metrics measured, or the year measured (which can affect plant growth via precipitation). I expected sites to stabilize as they aged, becoming more resilient to stress and self-sustaining. Each state variable showed new site variation, none of which appeared to be influenced by the time since restoration.

2.6b Annual Plant Success

Exploring the health of annual communities as a conduit for measuring restoration success has its strong and weak points. When exotic and native annual communities are measured on their own, little information can be garnered regarding restoration "success" especially in a desert ecosystem. In successional theory, which was developed in more mesic ecosystems, annuals are considered as the first pioneers; however, besides some invasive plants, this pattern for well-balanced ecosystems is largely unobserved in the desert (Vasek 1979). When combined with other, longer lasting metrics, a more holistic picture of the changing landscape can be discerned. Annuals provide much of the organic matter for the next year's plants (Brittingham & Walker 2000). Pollinators and other fauna may rely on these as food sources. For example, the endangered desert tortoise *Gopherus agassazii* particularly likes eating the native annual forb *Plantago ovata* (Abella et al. 2015). This plant inhabited all of my study areas. Annuals, like *P. ovata,* are very much subject to the timing and amount of rainfall of the previous year or even several years (Berry et al. 2015). Droughts in the Mojave can last a couple of weeks to several years and vary in severity, or particularly cold or hot spells can, in the rarest and extreme circumstances, kill off large portions of the native plant community within days.

Annual plant species may adapt phenotypic plasticity or hedge betting over time to ensure their seed is carried into the future (Gremer et al. 2016). Caution should be exercised when addressing native annuals as an indicator of health. They may be best used as a metric over a long period of time. A useful tool is assessing the annual community under shrubs and between interspaces, especially when comparing exotic and native cover. Berry et al. (2015) found that the overall recovery of winter annuals was faster in interspaces than under shrubs.

Exotic annual cover can more easily be viewed as an indicator of ecosystem health than native annuals. At a landscape scale, it is easier to see plants that are not supposed to be growing there, compared to plants that are. In addition, exotics are posited to slow the recovery trajectory by sequestering nutrients, using limited water, and slowing the establishment of perennial or annual colonizers (Berry et al. 2015). Quantitative measurements are still necessary because we typically notice that most exotic species can be considered invasive as well: they can outcompete native annuals for water, shade, nutrients, have an abundance of seeds and an earlier germination period. All of these growth factors ensuring their success may directly affect native annual failure. A tip of scales like this would directly violate several SER attributes (1, 2, 3) and even longer-term monitoring could show if this has impacts on other life forms, rendering the ecosystem not very self-sustainable.

The below annual tests of restoration success only account for the 2019 monitoring season. However, the richness, cover, and nativity of annuals communities are good indicators of ecosystem health, and thus, restoration. All of the restoration methods used in the six studies affect the microtopography of the soil. For example, both outplanting and abiotic manipulation to the soil can allow for seed catchment and water permeation (Berg & Steinberger 2012). Major obstacles to full native community development include deeply compacted soil, like Road 108, or highly disturbed soils that may be denuded of viable seeds, like the plots that did not receive topsoil at Northshore Road. In sites that had an outplanting restoration, the act of physically planting the shrub decompacts the soil. The shrub, whether it lives or dies, will contribute organic matter to proximal soil and still provide some shade. Vertical mulching should have a similar impact to planting, but I hypothesized it would not be as effective. A surviving plant will continuously defoliate and house animals each year, both of which add nutrients to soil, while the only sure addition vertical mulch confers to the soil is shade and the typical arid, slow-paced decomposition of organic material. Vertical mulch has been advocated as a natural enhancer of annual plant communities (Berry et al. 2015). Considering all annuals at an individual site level, I expected Northshore Road, Road 108, and the Fish Hatchery to show underwhelming annual communities compared to other sites. The first two sites are Gypsid Aridisols, which are characterized by not only low shrub cover, but also a Gypsid, Calcid, and biocrust layer that hinders water permeation (Belnap & Eldridge 2001; Lato 2006). In the case of the Fish Hatchery, I did question whether the well-developed mosaic of the desert pavement and biocrust layer would make it difficult for annual seeds to anchor and germinate.

Overall, there is no indication that the type of restoration will greatly affect native annual cover. The type of restoration, the age of the restoration, and the extensiveness of the treatments did not have noteworthy impact. For example, the Fiber Optic Cable and the Fish Hatchery, which both had *L. tridentata,* outplanting and share very similar plant communities showed opposite results as far as the ratios of restored to unrestored indicate. Four sites showed improvement in native annual cover with treatment and all four of these sites included a unique combination of the treatments discussed in this thesis (i.e. outplanting, soil manipulation, and seeding). Road 108 showed extreme values concerning annuals. A feasible postulate could be that ripping allows for better water permeability and seed catchment (Webb & Wilshire 1980), and this trend is applicable to gypsum soil. The other site with gypsum soil, Northshore Road, showed no significant effects of any treatment type in Type III tests; therefore, it is another example indicating that the restoration effort itself has substantial impact. Keys View Road had cover that was worse in unrestored plots and even worse in restored plots compared to the reference. This site also had the most extensive restoration effort, with three different treatment combinations. Morningstar Mine--which did have a single seeded annual-- and Northshore Road also underwent extensive effort, and while the cover in restored plots is greater than the unrestored and the reference sites, these values are around 50% and 20% better. In addition, Road 108, which arguably required the least expense and human effort, showed the highest changes. All of the sites are relatively small and in close proximity to undisturbed desert conditions and seed-carrying vectors such as wildlife and weather. There is no reason to think the seedbanks surrounding the sites were inadequate. This does only constitute 2019 results, so further years' of data are needed to corroborate the limited impact on native annual cover conferred by restoration. The age of the restoration project seems an unlikely factor in determining restoration success of native annual cover. A more likely indicator of native annual cover is the amount of precipitation the preceding three years. However, this may not be the case when concerning native annual species richness.

Sites typically lost some species richness between restored and unrestored plots as they aged. A notable exception is Road 108, which had six-fold higher the species richness on its restored plots compared to unrestored plots. Overall, I expected that as the restoration site ages, seed banks are replenished and vectors have more time to spread them throughout the site (DeFalco et al. 2012), and make the number of species between treatments more even. Literature suggests, however, that the seed bank is lacking in persistent and late-successional species seeds (DeFalco et al. 2012). Having a small disturbed-to-reference ratio does not necessarily mean less species, only less difference between their numbers, and so to determine the restoration effects on richness, in depth analyses of the species present are performed. The majority of field sites had restored to unrestored ratios greater than 1.0, suggesting that restoration, regardless of type, can help increase native annual species richness. Restoration efforts are focused on perennial plants and typically not annual communities; rather, perennial establishment will subsequently have a positive effect on the native annual communities. Native annual species richness, like all evaluations involving annuals, is most directly related to the year and its associated precipitation patterns than it is to the biotic or abiotic elements around it (Berg & Steinberger 2012). While the nurse plant effect does have a large, albeit unknown impact, if the restoration project is budget or time-limited, it appears that cheaper abiotic methods are just as effective at increasing native annual cover as the biotic treatments. High native annual species richness is, ostensibly, an indicator of good ecosystem health, but hardly a deciding indicator alone.

2.6c Exotic Measurements

When considering exotic species, an integral facet to understanding their cover and diversity is the very nature of exotics and how they spread. For example, exotic species, which in all of my study sites were annuals only, tend to outcompete native annuals with an early germination time, a higher seed load, and a more generalist approach to living (Brooks & Pyke 2000). Exotic plants are more efficient at catching water and nutrients (Suazo et al. 2012). Exotic grasses and forbs are often spread via animal, vehicle, or human attachment (Brooks & Pyke 2002). Therefore, one could expect sites proximal to roadways and with limited geographical barriers, such as water or steep topography to have higher exotic metrics (Berry et al. 2015). On the other hand, sites that are fenced off, extremely remote, or have precipitous topography may have some decent protection from anthropogenic vectors, which is the top way invasive species spread across landscapes (Brooks & Pyke 2002). It seems unlikely that such barriers could greatly affect natural vectors. Only one of the sites actively managed exotic growth. Morningstar Mine was considered successful in its six-point performance goal of weed cover if no more than 5% of invasive covered the revegetation area annually. The site was manually weeded two years post restoration. Eight non-native species were found within the site area in 2011, however only two species were recorded in 2019. Morningstar mine is a site that is fenced off to the public, although easily accessed with a small amount of effort. Keys View Road, Northshore Road, the Fish Hatchery, and the Fiber Optic Cable are all very proximal to major roadways or heavily frequented walking and bike paths. Road 108 does have a nearby hiking trail; however, few people were observed there, especially since the water level of Lake Mead has receded drastically. There has been evidence of burro and other pack animal droppings that may carry seeds. The most remote sites are within the Mojave National Preserve: Morningstar Mine and the Fiber Optic Cable. While both are located near roads, they are likely relatively unfrequented compared to other sites, which are located in some of the most highly visited sections of their respective National Park Service units. In addition to geographical barriers, one could also expect exotic plants to hijack the better habitat microsites because of the earlier germination period.

It is unsurprising that the majority of the sites had less exotic cover on the disturbed plot types (unrestored, treatment) than the reference plot types: in many—especially more recent---cases, the treatment and unrestored sites are still very denuded of the perennial vegetation that could provide ample water, shade, and nutrients to annual cover regardless of nativity. In most of the perennial categories in Table 3, the direction symbols tended to match those of exotic annual cover or, in other cases, were simply not significant. This is mostly true, except where a perennial outplanting may in fact bring the total perennial density above that of the reference. Arguably, a reference area should have less exotic cover. Such places are now rare in the US, and especially within the southwestern federal lands system. The Southwest has a long history of grazing, mining, and caravanning, a lot of which actively continues today or the relict cost continues today; one example being the wild horses and burros that roam Lake Mead National Recreation Area. It is unclear why Road 108 had much more exotic cover than the reference, despite the reference having more plants. While ripping the soil surface to create some microtopography allows for seed catchment and could thus catch exotic seeds more easily and increase their cover, outplanting can also increase exotic cover by providing nurse plant effects. DeFalco and others (2012) suggest that abiotic soil manipulation can negatively influence seed bank regeneration via invasive species suppressing native species. One possibility is that the disturbed plots are crossed several times by an active hiking trail. It is unknown how often this trail is used, although fresh animal droppings have been observed on every occasion; livestock grazing has long been implicated in altering perennial and annual species compositions (Berry et al. 2015). Another possibility is that reference plots have an intact layer of biocrust and gypsum that is creating a semi-impermeable layer to water and seeds (Belnap & Eldridge 2001). In addition, the unrestored plots are so compacted from historic driving and the Gypsid and biocrust layer that seed catchment would be unlikely. Any of these reasons would remain true for native annual cover and as such, we find that symbols between native and exotic annual covers on Table 3 do tend to match when there is statistical significance. Instead of the six sites showing signs of competition, they tended to align. Perhaps in extreme stress circumstances, annual communities do not undergo as serious of competition for space and resources as in a more temperate environment. Brittingham and Walker (2000) posited that high stress environments tend to favor positive facilitation between plants while in low stress environments tend to compete more frequently. I expected exotic cover to be highest in the reference area and lowest in the treatment or unrestored area because I thought the higher shrub cover in the reference area would facilitate all plant growth, natives, and invasive plants. As previously mentioned, there appears to be some relationship between perennial cover and annual community cover, however the connection is less quantitatively clear.

Based on the ratios of disturbed to reference plots, it is reasonable to assume in most cases that at relatively small sites, it is likely to see a similar array of exotic species among every plot type. In addition, ratios between restored and unrestored plots tended to be very close to one, suggesting minimal species assemblage differences. Exotic annual species richness was mostly non- significant values among all sites and years, however this does not necessarily indicate that the levels are low, but rather do not change. All six sites had at least one species of *Bromus* and *Schismus,* and the majorities have two species of each. Between exotic grasses and forbs, exotic grasses had more cover by far, which has been noticed in other studies (Berry et al. 2015). The most common exotic forb among sites was *Erodium cicutarum.* Other common exotic forb species among sites belonged to the mustard family: most sites had at least one type of *Lepidium, Brassica, Malcomia, Descurainia,* or another mustard family member. The sites that were largest in area, Keys View Road and Northshore Road, exhibited the greatest exotic species richness in terms of the numerical means, which suggests that perhaps exotics align with the theory expressed in the species-area curve along with high visitation. The only significant site according to Table 3, Road 108 that had extreme disparity in its ratios, was highly negatively impacted by the alteration of its microtopography from ripping treatments. For all plot types in all sites, exotic richness ranged from 0.28 to 4.1 species per plot. There was an average of 2.4 species per treatment plots, 1.9 species per unrestored plots, and 2.0 species per reference plots. Because of the relatively small number of exotic species present among all sites, it is unsurprising that there is little noticeable difference between disturbed and reference plots.

Metrics where a decrease was detected tended to be precipitation dependent. For example, both native and exotic annual cover decreased in three sites. Road 108 showed decreases in both nativities; Keys View Road showed a mixed reaction towards native annual cover when the abiotic and biotic treatments were parsed out; and the Fiber Optic Cable showed an overall decrease in exotic cover among treatments for the 2019 monitoring season. While it could be considered a restoration "win" for exotic annual cover to be decreasing in sites, the fact that two of the three sites saw a decrease in both native and exotic suggests a problematic outside factor may be involved. Soil conditions and water availability are often implicated (Brittingham & Walker 2000).

In general, the four annual variables measured: native and exotic percent cover and species richness remain similar in their reactions. Regarding native annuals, covers vary substantially and richness may be increased by combination treatments but is typically not very significant in the first place. Exotic annuals tended to have higher covers in reference areas but higher richness in disturbed (unrestored and restored) areas. Exotic measurement means tended to increase near high-traffic areas and across larger disturbances. The maximum number of plants on a plot type was around four. Some trends emerge when considering general annual variables regardless of nativity, because across multiple years exotic and native tests displayed similar trends. Cover changed more frequently than richness, which tended to remain relatively constant. It is very difficult to discern treatment effects in this scenario, but the trends that are possibly present are 1. Treatment and reference had more cover over time than unrestored areas and 2. Richness was higher in the treatment plots compared to the unrestored and reference areas.

2.6d Perennial Plant Success

Perennial plants are often not only the solution to restoring a disturbed area, but also the metric in which to measure if an area has had a positive restoration effect. Reestablishing perennial cover is often a very tangible measurement of restoration, because the plants are allocated to determine survival each year. While perennials, especially forbs and any seedlings are dependent on favorable weather conditions, as they mature, they become much less so (Miriti et al. 2007) with time. Establishing a decent and diverse cover of perennials, especially shrubs, allows for the genesis of the fertile island effect and thus further plant cover. The fertile island effect is initiated by concentration of organic materials and water underneath the canopy of a large perennial structure that is modifying the microhabitat (Brittingham & Walker 2000). When a plant exists in a particular area, soil and nutrients are naturally harvested and redistributed beneath it (Vasek 1979). Plants within the area receive more necessary components for growth than species growing in interspaces between shrubs. In fact, many perennials, especially members of the Agavaceae and Cactaceae families rely heavily on a nurse plant to facilitate their survival to maturity (Brittingham & Walker 2000). Despite the likelihood of fast paced nucleation apparent in other biomes, fertile islands—even as islands that are not fully integrated into the grater landscape—still provide refugia for certain flora and fauna, provide some ecosystem function as a self-sustaining patch. Some of the most iconic Mojave flora are included in the are naturally more diverse than others are, necessitating comparable reference sites (SER

2004). When comparing disturbed sites to reference sites, some possible trends to consider emerged. For example, in all sites, I expected treatment plot perennial plant average cover and species richness to fall between unrestored and reference sites; I expected sites that were restored more recently might veer from this hypothesis more than older sites. These more recent sites would likely be colonized by the "early invaders," while the reference area maintains biomass and changes very little (Vasek 1979).

Perennial plants in general can be classified into distinct life history groups that each contribute something unique to the overall vegetative structure: perennial forbs, perennial grasses, woody perennials ("shrubs"), and other large stature families including yucca and cacti.

In many sites, the perennials planted received some form of irrigation, although the exact details are difficult to interpret or confidently report using just the currently available resources. Some sites used DriWater®, a slow-release watering gel; some sites were hand-watered, and some sites were given substantial water at planting, and none thereafter save for natural precipitation (Newton 2001). Considering that water is the most limiting growth factor in the desert, it is unfortunate to not know the exact circumstances of irrigation at each site. Plants at the Fiber Optic Cable were also caged to prevent herbivory and the cages remain there currently. I was concerned that the cages may be negatively affecting the outplanting' growth, because it may chase them to grow up rather than sprawl, which could impact the verity of restoration success that are measured with percent cover estimates. Density would not be affected.

The treatment sites received should, logically, directly affect perennial cover, species richness, and density because almost all outplants and seeds used were perennial species. Perennial cover and species richness generally increased among sites, although results are mixed. Among the sites that did increase, all had received a biotic treatment component at least. It appeared that vertical mulch pulled down cover values at Keys View Road, suggesting a hindering effect of abiotic treatments; however, since outplanting was not significant at Northshore Road, and the arrow is still indicating an increase, the abiotic topsoil replacement did help increase perennial cover. Perennial density showed increases in every site. To elucidate whether this is a direct or indirect response of restoration efforts, life history groups were examined and also compared to restoration plan details.

Total perennial measurements were not analyzed with just 2019 data. To summarily describe responses of total perennials through time, treatments, especially with a biotic component, increased cover. Abiotic treatment exhibited peculiar results and rendered it difficult to draw conclusions regarding treatment alone. Richness did not change substantially across time or site. Density by far showed the most successful results across all sites, treatment types, and years (Table 3).

Perennial grasses had no consequential part of any site because of their minimal abundance. In fact, the majority of the sites had either none or just one species. Perennial grasses were noticed in the area, but perhaps there are too few in number to have the seeds necessary to colonize the restoration site.

Perennial forbs were a small portion of the perennial plants, and significantly increase at some sites. Species richness did increase at sites that had a mix of biotic and abiotic treatments, but the statistics are too sensitive to determine which treatment may have affected the change. There were typically less than two species per site. Overall, perennial grasses and forbs constitute what is termed "herbaceous cover". Herbaceous cover dies back every season and regrows anew from live underground parts during favorable climate conditions (Beatley 1973).

They tend to comprise of less than 3% of the total plant cover of undisturbed areas (Beatley 1973).

Shrubs, which include woody perennials and tall stature perennials such as *Yucca brevifolia* and *Y. schidigera*, constitute the final component of perennial plants. A handful of shrub species can easily occupy large swaths of land and it is unsurprising to find that this large component is also the most important in gauging restoration success (Vasek 1979). The category is combined because they offer similar ecosystem services due to their tall stature: shade and shelter, and heterogeneity. This category offers decent functional heterogeneity because of their different photosynthetic systems that will contribute to the SER attribute of a self-sustaining ecosystem with structural complexity (Vasek 1979, SER 2004). Also offered is physical heterogeneity: the plants have different biomasses, heights, widths, et cetera (Vasek 1979). Such complexity has been correlated with bird recovery and enhanced seed dispersal (Ruiz-Jaen & Aide 2015). In addition, the heterogeneous mosaic of different shrub canopies, which could be facilitated by outplanting, can influence seed distribution, germination, and survival (McAuliffe 1988). There is also a direct correlation with some shrubs and *Yucca* species (Brittingham & Walker 2000). Obviously, the model shrub would potentially drop far more leaves and thus nutrients than some of the other perennials in the category, but oftentimes the numbers of these were too low to notice an effect. Despite their small number, the plants are a very important part of the Mojave Desert ecosystem and many of them are considered iconic.

Shrub cover of all years typically increased in all sites. The two exceptions were the Fish Hatchery and Road 108. It is not surprising that no effects were captured on Road 108; low shrub cover already characterizes the gypsum soil and no biotic component was added (Belnap & Eldridge 2001). The Fish Hatchery, however, had shrubs planted directly on to the site. When only 2019 was considered, these two sites have the highest shrub cover of all plot types and sites, including their own reference. Since the Fish Hatchery does not show significant with all years among plot type, but shows major changes when it is just 2019, it seems plausible to posit that shrubs are recruiting and growing on a quicker trajectory. The ratio of restored to unrestored in both sites is around two. This further suggests that restoration is quickening the pace of increasing shrub cover. Interestingly, this gives no indication as to whether the abiotic ripping on Road 108 or the biotic outplanting at the Fish Hatchery is more effective. Logically, it seems that outplanting would be a much stronger factor, however if the outplants do not survive, then the site may not be much better as when restoration had yet begun. To add to the conundrum, Keys View Road, while being the only site that did this, had higher shrub cover in unrestored plots compared to treatment plots. In comparing across years, all three types of treatment plots consistently had the lowest values; in most years even lower than unrestored plots. There does not appear to be much year-to-year difference between treatments; treatments with outplanting are slightly higher than those without, but the effect is too small to detect any significant difference. Another important consideration when using shrub cover as a metric for restoration success is to consider the types of shrubs colonizing the area. In a highly disturbed area, the percent cover may be higher than the reference, but the cover consists of short-lived, colonizing shrubs and not climax community shrubs. Thus, the reestablishment of perennials is variable in not only numbers, but also quality (Webb et al. 1987). This is especially visible in the Fish Hatchery where early colonizers only appeared on unrestored plots or exhibited year and treatment interactions while long-lived shrubs tended to be low in number.

In an attempt to answer the question of whether to use biotic or abiotic treatments, it appears that the restoration effect will depend largely on the time scale and budget devoted to the project. For example, if there is a limited budget, abiotic treatments may be sufficient although the process may take longer. For biotic treatments, the trajectory is greatly sped up by outplanting; however, if the project does not have the budget, time or manpower, to maintain the outplants until maturity, it is mostly likely not worth it. Grantz et al. (1998) concluded that transplanting without intensive irrigation did not facilitate survival and was not worth the high cost. If the shrubs die, they could be considered as vertical mulch that may still enhance recovery (Berry et al. 2015), but at the abiotic trajectory pace and not the biotic trajectory pace. Sites surveyed by Webb et al. (1987) showed partial revegetation in an unrestored old road 46-78 years after abandonment, but only "some" revegetation can hardly be considered success if the goal is high similarity to a reference. Other sites with restoration treatments showed similarly slow-paced revegetation regardless of biotic or abiotic components.

Soil compaction is a major disruptor of natural revegetation and recovery (Bolling & Walker 2000). Thus, even a cheap treatment ameliorating compaction may speed the recovery trajectory, although exotic species could colonize the area and outcompete native annuals and seedling perennials like in Road 108. Bolling and Walker (2000) also noted that decompaction will help the natural recovery trajectory, although they do not mention the possibility of exotic take over, presumably because they only focused on perennial cover. In one site, old dirt roads in Nevada were still too compacted for proper vegetation recovery after 51 years (Webb & Wilshire 1980).

According to a regression conducted by Kimball and others (2015), seeding and planting as a single category most strongly influenced native cover, followed by maintenance. The cost-effectiveness of restoration, measured as an index calculated as percent native cover/cost per hectare, was compared between many restoration treatments (Kimball et al. 2015). Because of the labor and expense involved, the authors determined that the transportation and use of salvaged topsoil was one of the most expensive, but most effective measures; seeding native seed and tamping was the most expensive seeding method at \$4,942 ha^{-1}. The cost incurred will not effectively tip scales in favor of restoration without goal-oriented and proper planning in order to ascertain whether the restoration was successful.

2.6e Weather Effects

The Mojave Desert is largely a winter rainfall desert; however, this becomes more bimodal further east (Abella & Newton 2009). The percentage of precipitation that does not fall in the winter is a part of the summer monsoon season. While this knowledge is incredibly hand, it is not dependable from a restoration ecology standpoint. Upon examining weather patterns in relation to restoration events, two interesting points became abundantly clear. Firstly, inter-annual variability in precipitation tends to affect the magnitude of restoration effects. In other words, the clarity of success or failure changes each year is shrouded by unreliability. The second point has consequences that are direr: poor weather conditions may directly impact the restoration success. There are conduits, however, to somewhat handle the conundrums weather caused.

Arid vegetation, especially the annuals communities, is subject to the varying amounts of precipitation and the timing of these rain events each year (Berry et al. 2015). Perennials and annuals are affected; however, the effect with annuals is much more obvious, especially during the famous and colorful display of Mojave Desert wildflowers. The majority of precipitation in the Mojave Desert occurs during winter; however, summer rains will increase in the central and eastern parts (Brooks & Pyke 2002). One of the SER (2004) attributes is that the ecosystem is resilient; it can withstand periodic stress from heat, drought, frost, et cetera. A large portion of the success of a restoration project appears to be up to luck or very good weather prediction models. In all cases, it is absolutely necessary to understand the past climate and yearly variability of precipitation and extreme weather before implementing a restoration project.

Climate change may affect the precipitation patterns and frequencies of drought and extreme heat in arid ecosystems, negatively influencing even perennial plants (Beatley 1980; Berry et al. 2015; Hereford et al. 2006; Miriti et al. 2007). These long-term weather events slow vegetation growth rates. By slowing growth rates, the rate of recovery after restoration for a given project will verily slow; and thus, the need to consider and report weather more frequently in future published articles will lend to predictions of best restoration practices regarding climate influence (Copeland et al. 2017). The precipitation pattern of the Mojave Desert (e.g. larger percentage during winter; smaller percentage during summer monsoons) may change due to climate change causing increases in large water-pulse events (Suazo et al. 2012). Flora will be forced to respond to these patterns, which could drastically alter Mojave Desert vegetation structure.

Both perennials and annuals are especially vulnerable to unfavorable weather conditions during germination and at the seedling stage; and, in the case of annual plants, the particular species' survival is directly dependent on the fitness of the plant being strong enough to release viable seeds (Gremer et al. 2016). Vasek (1979) concluded that large-scale recruitment did not occur even with favorable weather conditions. He posited that some factors associated with raw soil precluded germination and establishment of long-lived shrubs. However, according to Brittingham & Walker (2000), water availability is the limiting factor, followed by nitrogen concentrations.

Drought and timing of precipitation affects species differently. For example, during a wet period from 1976-1998, a particular stand of *L. tridentata* drastically increased in size and density; during a particularly brutal drought from 1989-1991, a population of chenopods and perennial graminoids suffered 100% mortality (Hereford et al. 2006). Annuals typically rely on winter precipitation, while cacti do well with some summer precipitation (Hereford et al. 2006).

Keys View Road systematically had high levels of rainfall. The site also varies greatly in elevation and is along one of the most popular tourist roads within the park. While monitoring the sites, there was evidence of deep soil erosion and fissures from road runoff during precipitation events. As part of this deep erosion, the vertical mulch pieces situated back in 2006 hardly remain standing today. When monitored, they were often laying on the ground, unburied, or broken. In other plots, it was almost impossible to identify vertical mulch pieces. The data shows vertical mulch (VM) treatments tended to lower the means for favorable metrics caused by the outplants. Because outplanting can reduce erosion, we would expect to see less eroded sites where the outplanting occurred: the outplanting (OP) plots and the vertical mulch and outplanting combination (OPVM) plots. After consistently seeing the VM and OPVM plot means lag and the erosion increase over time, I concluded that areas with high rates of water flow (e.g. near roads, steep slopes, high elevation areas) may work best with an expensive outplanting treatment rather than an abiotic treatment. Seeds or vertical mulch, unless perhaps paired with a third abiotic treatment like jute, would simply wash away from the area.

In addition to directly affecting the success of the restoration treatment itself, abnormal precipitation patterns can alter how successful a project appears to be; or, with limited years of data, may show incorrect results. For example, Morningstar Mine and the Fiber Optic Cable were measured in only 2019. These two sites also do not show significant results for most of the annual state variables.

Several sites showed low annual plant cover during 2018 and low perennial cover during 2019. I speculated that precipitation must have been lower than average in 2018, so that very few annuals germinated and that perennials either died off or did not grow for the 2019 growing season. Years of high rainfall have been positively correlated with an increase in biomass of invasive annual grasses, and it is speculated that in poor rainfall years, some invasive species may lose their dominance at lower elevations (Brooks & Pyke 2002). Road 108 had significantly greater total exotic cover in 2019 compared to the extremely low years of 2009 and 2016. As seen on the weather chart in Appendix 1, the annual precipitation for 2009 was 7.6 cm, in 2016 was 22.3, and to September 2019 was 19.4 mm. The long-term average is about 13.9 cm year^{-1}. It is hard to tell from this little amount of data alone if it made a difference, but the speculation of Brooks & Pyke (2002) is substantiated: with increasing precipitation, there were also increases in exotic cover. The year 2016 is more similar to the low cover of 2009 than it is to 2019, despite being the highest annual precipitation rate. Interestingly, if the precipitation of the three prior years to each monitored year is averaged, 2009 and 2016 have a difference of 0.10 cm year^{-1} and the average for the three years prior of 2019 has over seven cm more precipitation in those years at an average 17.9 cm year^{-1}. Native annual cover follows a similar trend. At Northshore, perennial cover increased from 2016 to 2017, but from 2017 to 2019 had decreased again. The mean monthly averages from 2016 to 2019 were 18.6 mm, 8.5 mm, 16.7 mm, and 21.6 mm. It appears that 2017 actually had less precipitation. The problem with measuring yearly, however, is that the majority of the precipitation falls when the year is changing. Almost all other sites follow the trend that the least amount of precipitation fell in 2017.

Pairing biotic restoration treatments with favorable climatic condition years may be a conduit to increasing the success of these restoration projects (Copeland et al. 2017). Although weather is hard to predict, some guidance can be attained by analyzing past climate patterns and attempting to determine when the next El Niño Southern Oscillation will send wetter winters to the American Southwest and avoiding La Niña conditions (Hereford et al. 2006). Perennial plants using the C3 photosynthetic pathway tend to respond favorably to cool-season precipitation and *L. tridentata* germinate well with warm-season precipitation (Hereford et al. 2006). At Morningstar Mine, transplant survival did not meet the proposed success criteria. Researchers speculated that the plants were simply not relocated, washed away by heavy rainfall, or killed by unusually heavy and repeated snowfall. Seeds, assuming the amount of granivory was not too overwhelming, could survive, and thus both biotic treatments were helpful between surviving transplants and germinated seeds. Perennial forb cover and native species richness followed similar patterns. Aside from precipitation constraints, restoration effectiveness can be increased with proper site preparation, the use of suitable native species, proper implementation procedure, and clear and unified understanding of the project goals by all involved parties (Grantz et al. 1998).

2.6f A Word on Literature

t is unclear how much of the ecoregion needed restoration, but it is certainly greater than one percent;

From 1940 to 2010, the Mojave Basin and Range, which has an area of approximately 127, 690 km^2, was restored by the Bureau of Land Management. It is unclear precisely how much of the ecoregion needed restoration, but it was certainly greater than one percent (Copeland et al. 2017). Faring even worse is the Sonoran Desert at less than 0.1% (Copeland et al. 2017). It is unclear how much of the ecoregion needed restoration, however in comparison with other arid regions, the treatment area to ecoregion area is extremely small. The ecosystem is not afforded equal management considerations during planning and this subsequent historical pattern of effective management dearth pushes it even deeper into a threatened state (Giardina 2011). Without planning, success rates plummet and creating a frustrating feedback loop: With low success rates, land managers may not want to spend money on arid land restoration attempts and without restoration attempts and post-monitoring, arid land restoration attempts will continue to have low success rates. Unsuccessful or too few attempts worsens desert health, and so on.

The paucity of cost- and success-related information in published literature in desert scrub and riparian woodlands has been slowly redressed in the past decade along with a restoration goals rather than resource extraction focus (Copeland et al. 2017). If this increase in reporting both costs and success rates in the short and long term continues, effective restoration methods can be written based on common goals, common landscapes, or common problems, where it can be synthesizes and generalized with more accuracy and purpose.

In the spring of 2018, I had a site list of nearly twenty-five restoration projects. By the time of 2019 monitoring, that list was cut to six oftentimes because there was simply not enough information about the site itself.

2.6g Continued Disturbance

The popularity of federal lands in unsurprising: National Parks are designated because such a large number of people find the land so beautiful, special, and worth preserving. However, this very idea makes visitation among the parks high. Despite signage, it is difficult to keep visitors from accidentally or purposefully destroying delicate features. Sites that were along the road were especially vulnerable to human disturbance. For example, many plots in Keys View Road were directly by a parking lot or an information pullout. On many occasions there were people walking directly in the plots, potentially destroying seedling vegetation and compacting the ground. During this year's government shutdown, Lake Mead National Recreation Area was so overrun with trash, that there were several small fires and litter along the roads and into the undisturbed desert lands (personal correspondence). Continued disturbances make restoration even more difficult. It also makes monitoring them even more difficult because it is likely unknown whether an area was disturbed throughout the rest of the year or not. This can be solved by mixing social sciences, studying visitation and land user knowledge, with ecological practice, using restoration to fix degraded areas.

2.7 Conclusion

A restoration site cannot be considered concretely successful unless there were goals set prior to restoration to actually compare the site. Without these goals, it is unclear what exactly the trajectory should be. For example, should the restoration site be the most similar to reference sites to be considered restored? For example, if treatment conditions surpassed reference conditions and were more different than unrestored conditions were to reference, like seen in many variables (Figure 4-8A-D), is the restoration still a success? Should it be least similar to other destroyed parts? Should it provide certain function or service, and once completed, self-sustaining and is considered restored?

The Morningstar Mine reports provided a great example of goal-oriented restoration. For example, they set goals for a minimum amount of exotic cover and a certain annual survival for the transplants. Concrete initiatives allow the question of whether or not restoration was a success to be answered. While it is still possible to determine if there was a "restoration effect" without clear goals, it is vague as to what purpose the area was restored in the first place.

Typically, all sites experienced a positive restoration effect in some regard. Both abiotic and biotic treatments affected at least a couple variables. The above evidence leads me to conclude that restoring the creosote-bursage community requires a biotic factor to strongly notice restoration effects within a practical amount of time. In addition to positive results, no treatment experienced consistently negative results per metric across sites or years. In determining a decent treatment to use, considering final goals and the cost of resources to meet these goals is essential. Overall, biotic treatments tended to have a larger restoration effect and practically essential role in *L. tridentata* reestablishment. The biotic treatments must remain biotic and not quickly perish, thus becoming abiotic according to the definition established in this thesis. Long-term monitoring provides data to uncover solutions to restoration effectiveness questions; solutions that are indeed translatable more specifically to individual sites, ensuring that the predetermined goals are being met.

Time is an essentially unavoidable factor for the slow-paced recovery of dryland ecosystems; however, restoration methods can reduce recovery time substantially. Increased project implementation (including absolutely all relevant information) and more research will advance restoration ecology as a science, and therefore project success rates, even further. Such advancements amplify confidence of land managers concerning the ataraxic surety in the cost-effectiveness of each project. This creates a positive feedback loop that indirectly achieves the broadest possible goal: successful reversal of anthropogenic damage to disturbed arid ecosystems.

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2.9 Tables and Figures for Chapter 2

Table 2.1 Common Restoration Treatments.

Descriptions of three common restoration treatments and subtreatments. All subtreatments options per treatment are not presented. Only subtreatments used in one of the six sites is presented.

Type	Treatment	Subtreatments in Sites	Notes
Biotic	Seeding	Hand-broadcast, hydro-seeding	Seeds collected in wild & private repositories
	Outplanting	Transplants, nursery-grown stock	Fencing, irrigation common
Abiotic	Vertical mulch	---	Used dead, convenient material
	Soil manipulation	Decompaction, ripping, recontouring, topsoil reapplication	Heavy machinery, hand labor

Table 2.2 Project Site Information.

Asterisks (*) denote treatment combinations.

Site	Name	Treatment	Date Disturbed	Date Restored	Years Assessed
1	Northshore	Recontouring, topsoil application, outplanting (nursery stock)	Until 2008	2008-2010	2016, 2017, 2019
2	Road 108	Ripping with heavy machinery	Until 2002	2002	2009, 2016, 2019
3	Keys View	Recontouring, outplanting (nursery stock); vertical mulch; outplanting × vertical mulch	2007-2008	2008	2009, 2010, 2011, 2012, 2018, 2019
4	Morningstar Mine	Recontouring × seeding × outplanting (transplants)	Operational mine: 1907-1942, 1964-1992. Cyanide leaching until 2002	2008-2009	2019
5	Fiber Optic	Outplanting (nursery stock)	Late 1990's	2000-2001	2019
6	Fish Hatchery	Outplanting (nursery stock)	1968, 1998	1999	2006, 2018, 2019

Table 2.3 Summary of restoration effect on plant life history groups commonly surveyed among projects.

Summary effects include all years surveyed for each site. Treatment types represented per project include abiotic (soil manipulation, vertical mulch), biotic (seeding, outplanting) and biotic/abiotic (site received both treatment types that because of statistical effects, were parsed for individual results. An up arrow indicates a positive effect and a down arrow indicates a negative effect of restoration on the plant life history group variables. An up and down arrow together indicate mixed responses in sites that received multiple treatments. Therefore, individual tests indicate that one of the treatments is helping while the other is hindering positive restoration success. A dash indicates no effect. Variables that were not measured are marked as not applicable. Significance was based on $\alpha \leq 0.05$ and was considered moderate at $\alpha \leq 0.10$, denoted with a superscript a ([a]) next to the corresponding arrow.

		Native plants							
Project	Treatment Type	Perennial Cover	Perennial Species Richness	Perennial Density	Perennial Forb Cover	Perennial Forb Richness	Perennial Forb Density	Perennial Grass Cover	Perennial Grass Richness
Fish Hatchery	biotic	-	-	↑	-	-	-	↓	↓[a]
Fiber Optic Cable	biotic	↑	-	↑	-	-	-	n/a	n/a
Keys View Road	biotic/abiotic	↓	↑	n/a	-	↑	n/a	-	-
Northshore Road	biotic/abiotic	↑	-	↑[a]	↑	↑	-	-	-
Morningstar Mine	biotic	↑	↑	↑	↑	-	↑[a]	-	-
Road 108	abiotic	-	↓	↑	-	-	↑[a]	n/a	n/a

		Native Plants						Exotic plants	
Project	Treatment Type	Perennial Grass Density	Shrub cover	Shrub richness	Shrub Density	Annual cover	Annual richness	Annual cover	Annual richness
Fish Hatchery	biotic	-	-	↑[a]	↑	↑	-	↑	-
Fiber Optic Cable	biotic	-	↑	-	↑	-	-	↓	-
Keys View Road	biotic/abiotic	n/a	↓↑[a]	↑	n/a	↓↑	↑	↑	-
Northshore Road	biotic/abiotic	-	↑	-	-	↑	↑	-	-
Morningstar Mine	biotic	-	↑	↑	↑	-	-	↑	-
Road 108	abiotic	n/a	-	↑	↑	↓	↓	↓	↓

Figure 2.1 Vandalism and theft at the Morningstar mine.
Damage occurred in the treatment area, potentially ripping out small outplants and disrupting soil seed bank. Courtesy of the National Park Service.

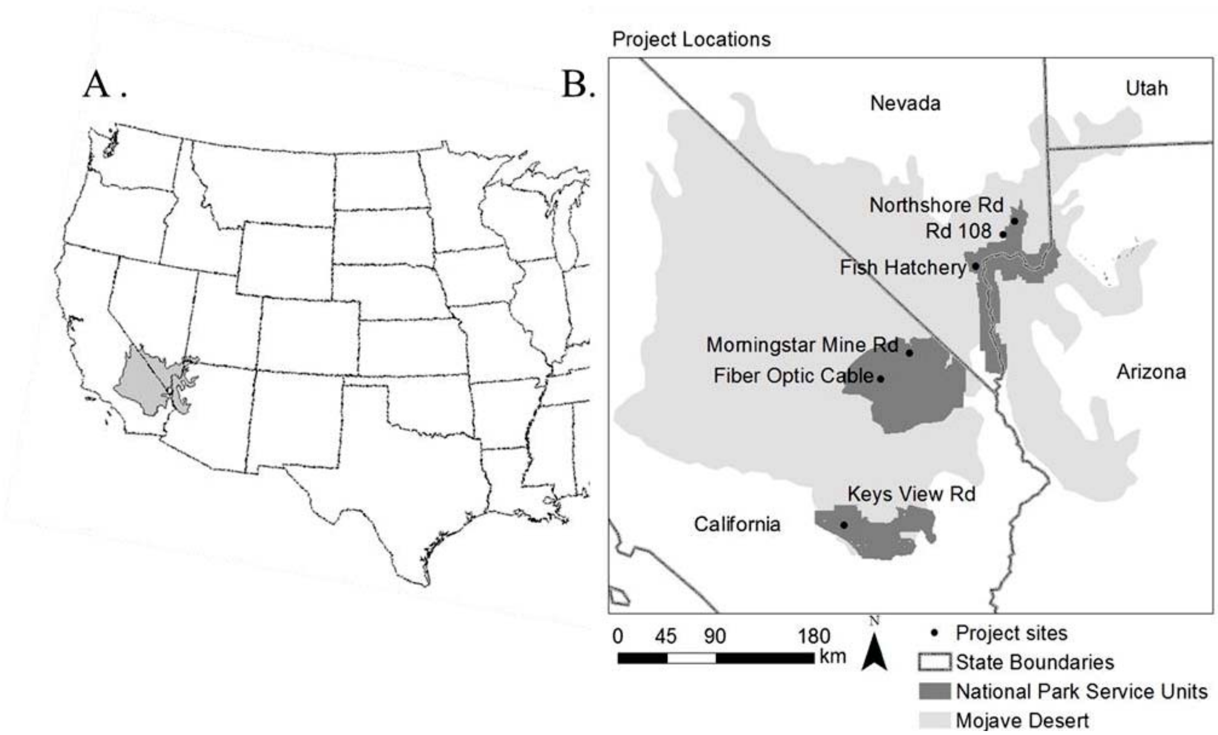

Figure 2.2 General location of six arid long-term monitoring restoration field sites.

A) The Mojave Desert is located in the southwestern United States of America. It is bordered on the north by the Great Basin Desert and to the south by the Sonoran Desert. The Mojave Desert includes the states of California, Nevada, Arizona, and Utah. B) The six sites are located exclusively on federal land managed by the National Park Service. From North to south: three sites lie in Lake Mead National Recreation Area, two sites lie in the Mojave National Preserve, and one site lies in Joshua Tree National Park.

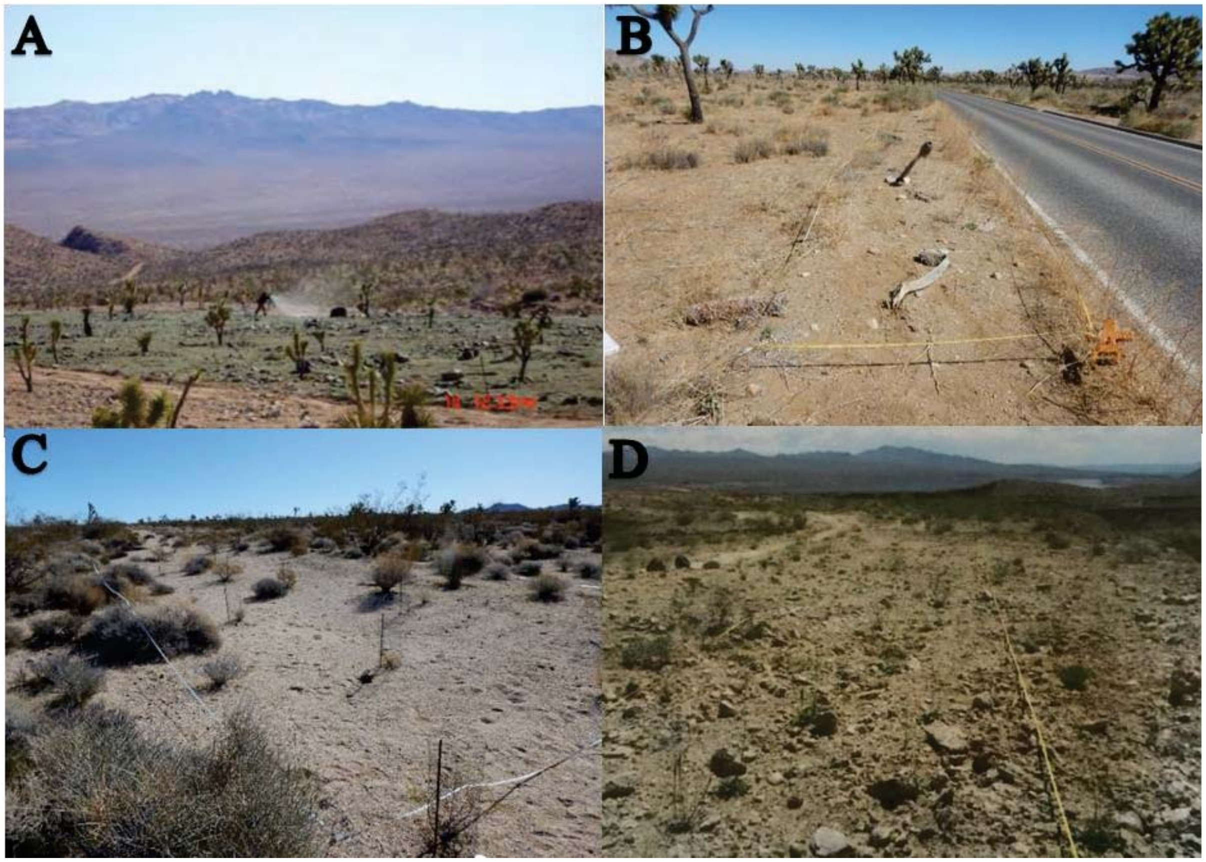

Figure 2.3 Restoration Treatments at Field Sites.
A) Hydroseeding the restoration area after outplanting at Morningstar Mine (National Park Service). B) Joshua tree vertical mulch structures at Keys View Rd. Many had fallen down over time (original content). C) Caged *L. tridentata* outplants at the Fiber Optic Cable (original content). D.) A plot that was ripped by heavy machinery at Road 108 (Chiquoine).

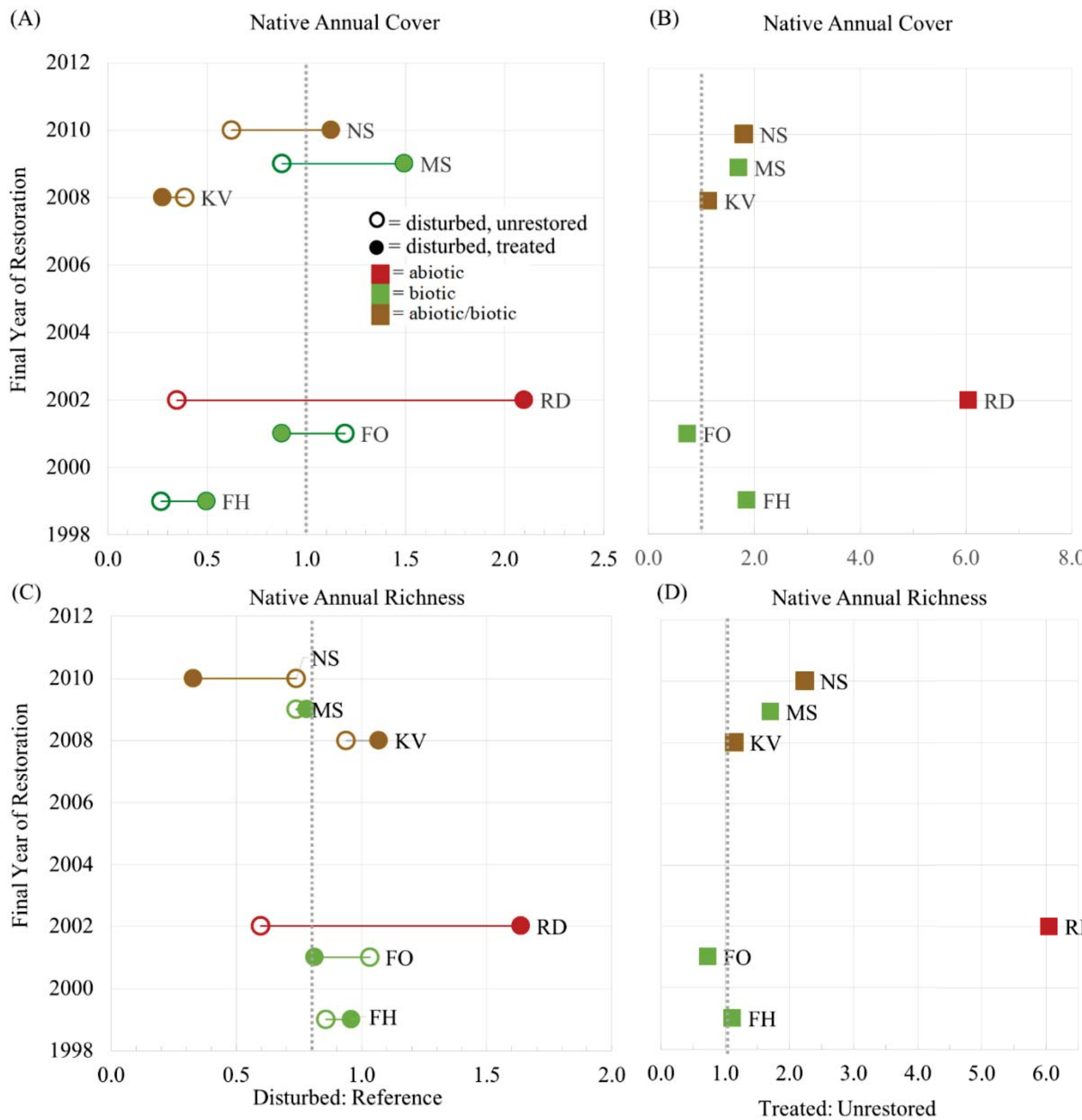

Figure 2.4 2019 results for native annual cover and species richness relationship values between disturbed plot types and reference plots (A and C) and relationship values between treated and unrestored plots (B and D) for six restoration projects in the Mojave Desert, USA.

Sites include Northshore Road (NS), Road 108 (RD), and a fish hatchery (FH) in Lake Mead National Recreation Area; Morningstar Mine (MS) and a fiber optic cable line (FO) in Mojave National Preserve; and Keys View Road (KV) in Joshua Tree National Park. Red indicates abiotic treatment, green indicates biotic treatment, and brown indicated a biotic/abiotic combination treatment. The y-axis presents the year restoration occurred. In A and C, the grey dashed line at x=1.0 indicated the reference value. A solid circle is the treatment: reference ratio value, a circle outline is the unrestored: reference ratio value In B and D, a value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed (treatment: unrestored). The square indicates that the assessed plots are disturbed.

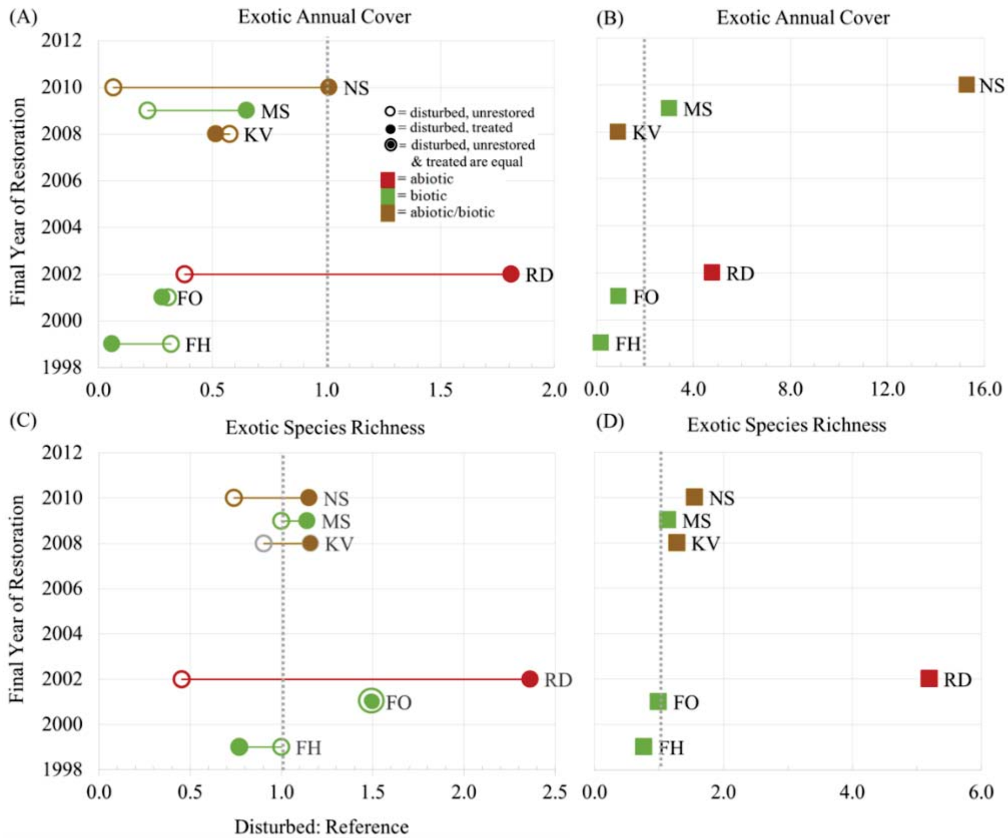

Figure 2.5 2019 results for exotic annual cover and species richness relationship values between disturbed plot types and reference plots (A and C) and relationship values between treated and unrestored plots (B and D) for six restoration projects in the Mojave Desert, USA.

A value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed. Sites include Northshore Road (NS), Road 108 (RD), and a fish hatchery (FH) in Lake Mead National Recreation Area; Morningstar Mine (MS) and a fiber optic cable line (FO) in Mojave National Preserve; and Keys View Road (KV) in Joshua Tree National Park. The y-axis presents the year restoration occurred. In A and C, the grey dashed line at x=1.0 indicated the reference value. A solid circle is the treatment: reference ratio value, a circle outline is the unrestored: reference ratio value In B and D, a value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed (treatment: unrestored). The square indicates that the assessed plots are disturbed.

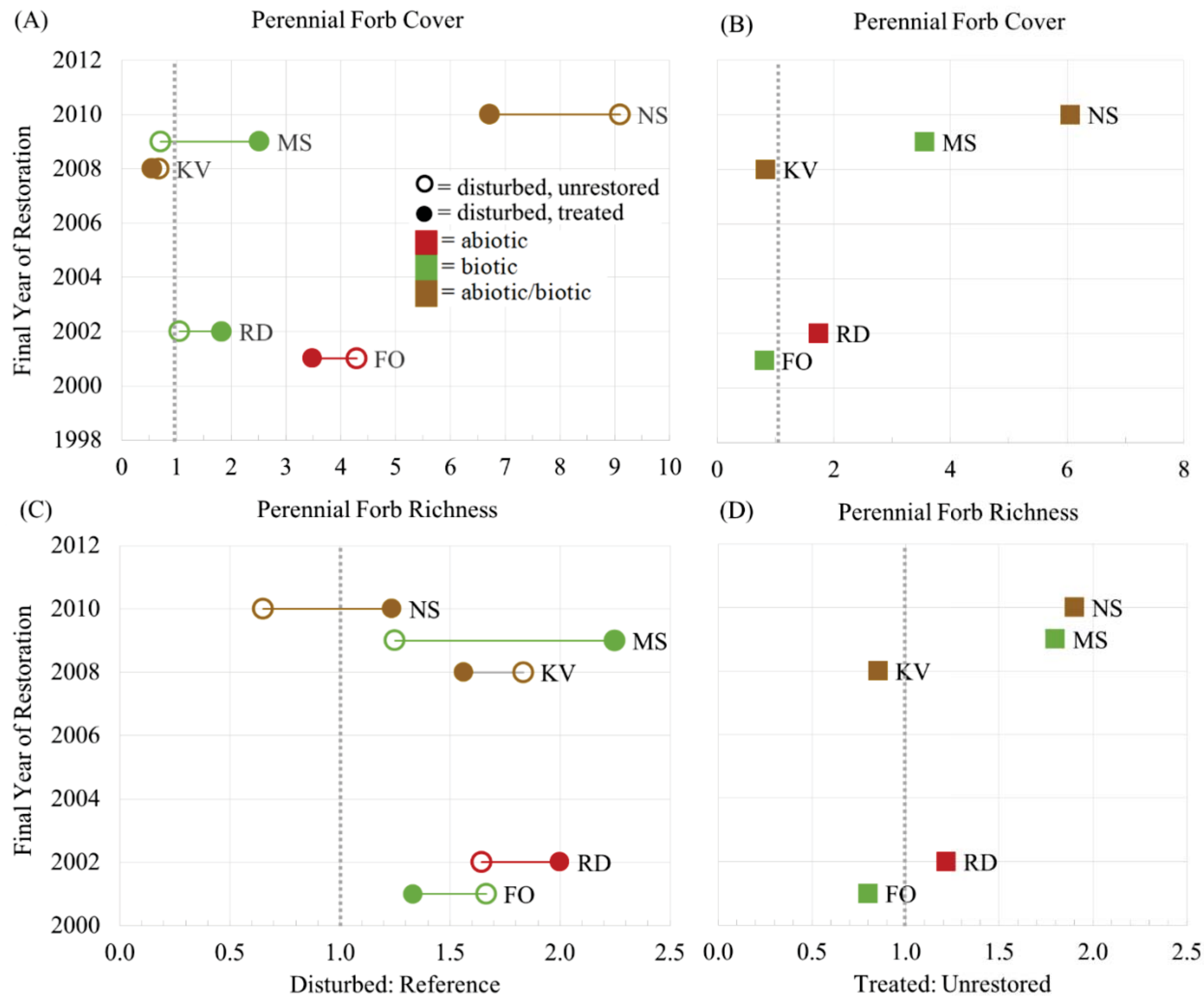

Figure 2.6 2019 results for native perennial forb cover and species richness relationship values between disturbed plot types and reference plots (A and C) and relationship values between treated and unrestored plots (B and D) for six restoration projects in the Mojave Desert, USA.

A value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed. Sites include Northshore Road (NS), Road 108 (RD), and a fish hatchery (FH) in Lake Mead National Recreation Area; Morningstar Mine (MS) and a fiber optic cable line (FO) in Mojave National Preserve; and Keys View Road (KV) in Joshua Tree National Park. The y-axis presents the year restoration occurred. In A and C, the grey dashed line at x=1.0 indicated the reference value. A solid circle is the treatment: reference ratio value, a circle outline is the unrestored: reference ratio value In B and D, a value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed (treatment: unrestored). The square indicates that the assessed plots are disturbed.

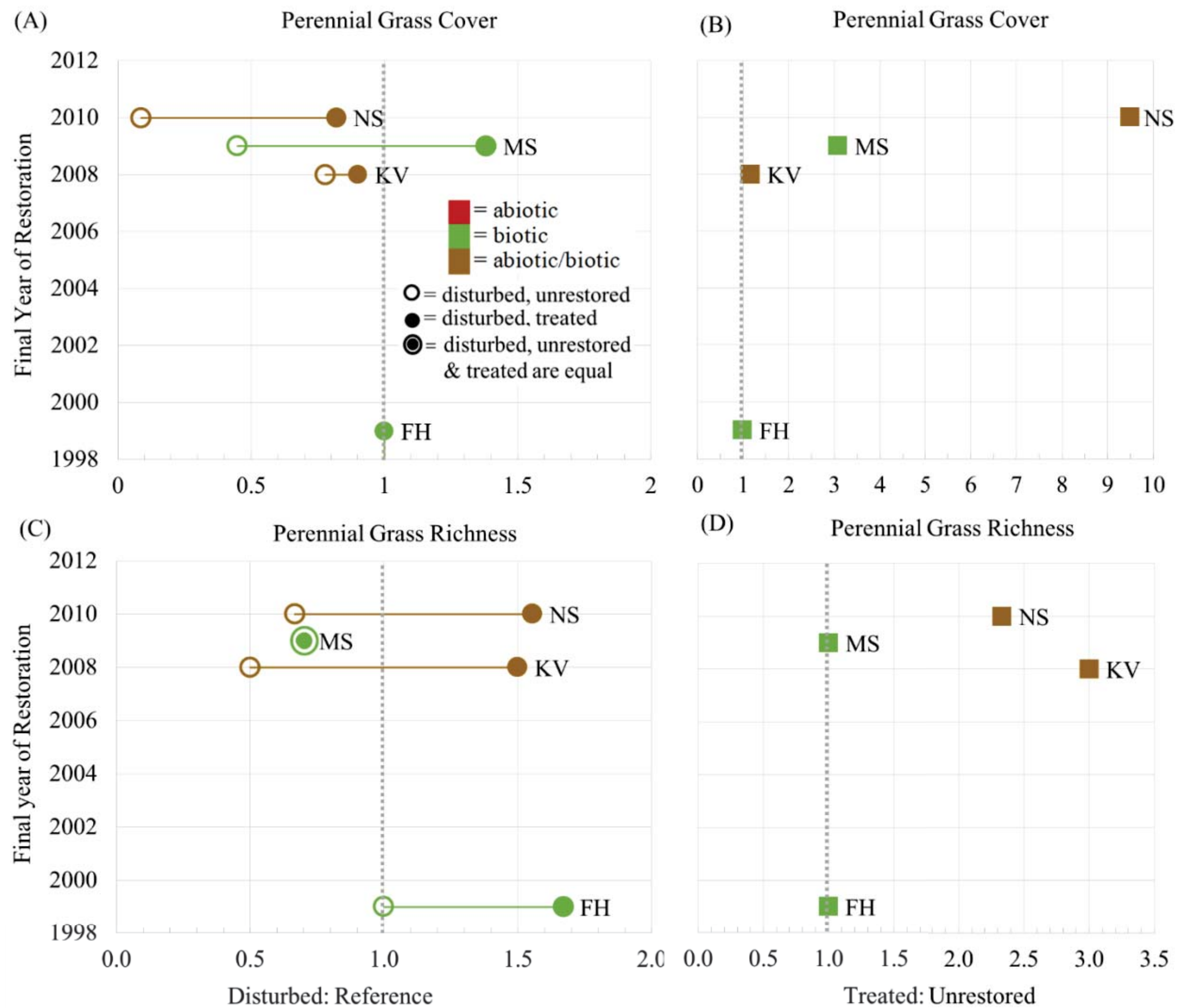

Figure 2.7 2019 results for native perennial grass cover and species richness relationship values between disturbed plot types and reference plots (A and C) and relationship values between treated and unrestored plots (B and D) for six restoration projects in the Mojave Desert, USA.

A value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed. Sites include Northshore Road (NS), Road 108 (RD), and a fish hatchery (FH) in Lake Mead National Recreation Area; Morningstar Mine (MS) and a fiber optic cable line (FO) in Mojave National Preserve; and Keys View Road (KV) in Joshua Tree National Park. The y-axis presents the year restoration occurred. In A and C, the grey dashed line at x=1.0 indicated the reference value. A solid circle is the treatment: reference ratio value, a circle outline is the unrestored: reference ratio value In B and D, a value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed (treatment: unrestored). The square indicates that the assessed plots are disturbed.

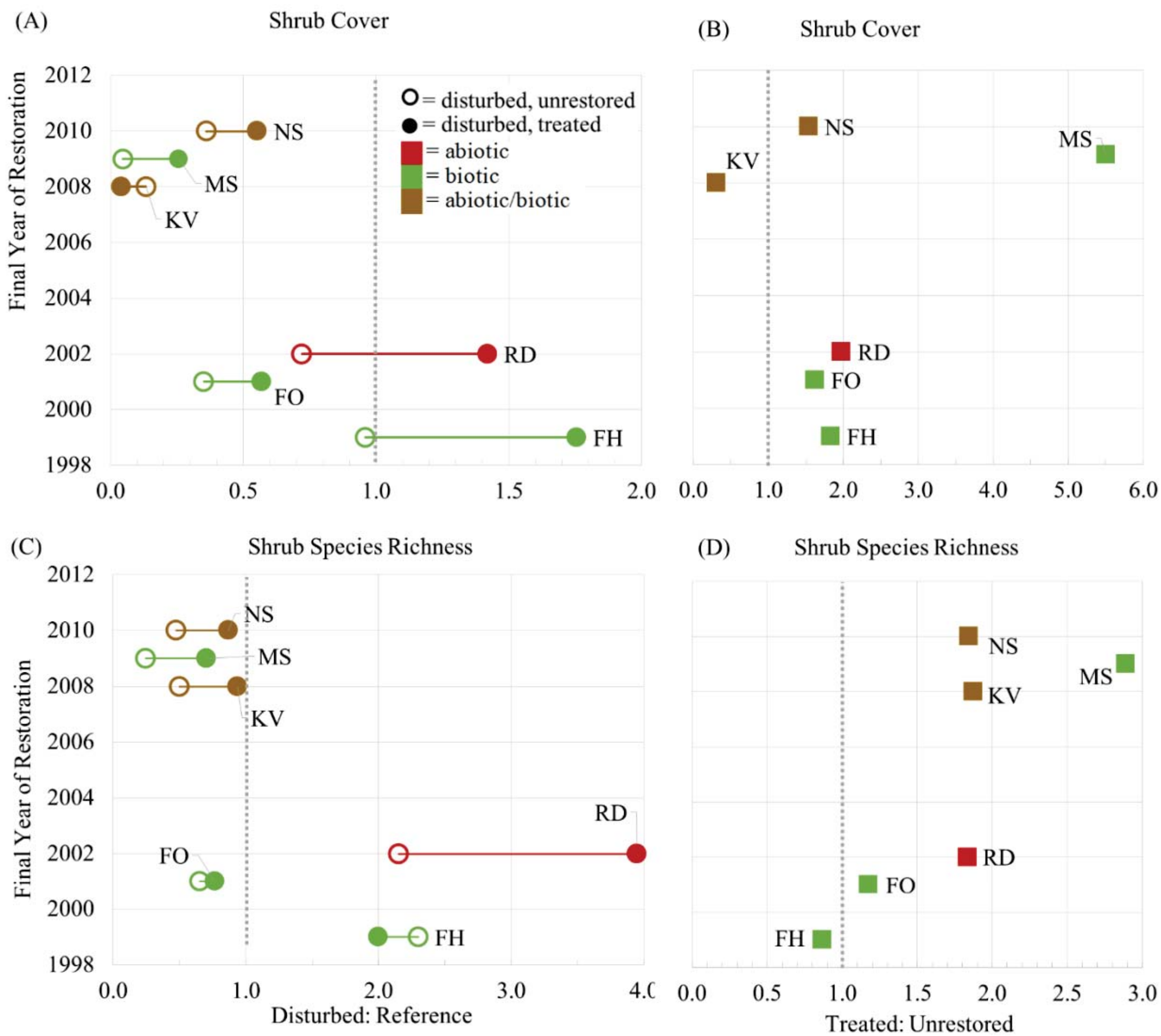

Figure 2.8 2019 results for native shrub cover and species richness relationship values between disturbed plot types and reference plots (A and C) and relationship values between treated and unrestored plots (B and D) for six restoration projects in the Mojave Desert, USA.

A value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed. Sites include Northshore Road (NS), Road 108 (RD), and a fish hatchery (FH) in Lake Mead National Recreation Area; Morningstar Mine (MS) and a fiber optic cable line (FO) in Mojave National Preserve; and Keys View Road (KV) in Joshua Tree National Park. The y-axis presents the year restoration occurred. In A and C, the grey dashed line at x=1.0 indicated the reference value. A solid circle is the treatment: reference ratio value, a circle outline is the unrestored: reference ratio value In B and D, a value of 1.0 (highlighted with the grey dashed line) indicates an equivalent relationship between the two plot categories assessed (treatment: unrestored). The square indicates that the assessed plots are disturbed.

Chapter 3: Outplanting at a low-elevation eastern Mojave Desert pipeline corridor disturbance positively altered the recovery trajectory towards undisturbed conditions, even when compared to 50 years of natural recovery.

3.1 Introduction

The past and present Mojave Desert suffers severe degradation from climate change and direct anthropogenic impacts. While the premise behind establishing a National Park system is both brilliant and beautiful, the stewardship and public protection is not always discerned at the contemporary level. Historically, the southwest United States was a frontier for opportunity via colossal livestock operations and staking mining claims. This socioeconomic opportunity and population booms contributed to massive migrations with a popular "Go West, young man" mantra. Today, the human population of the desert southwest continues its rapid growth, necessitating newer infrastructure to keep up with the pace. Even our protected lands must occasionally be altered to accommodate such an influx. The draw of adventuring across the arid Southwest began in Conestoga wagons and trains continues on I n electric cars and airplanes. Because of the extremely slow timescale of natural recovery in arid regions, active restoration methods are necessary to lessen the effects of these varying disturbances. While some disturbed dryland plant communities have repaired themselves within decades, much of the literature suggests this process can easily take centuries (Abella 2010; Lovich & Bainbridge 1999).

Even before a fuller understanding of climate change impacts, low precipitation compared to potential evapotranspiration rates have made restoration challenging. Due in part to the effects of the El Niño Southern Oscillation, increasingly sporadic and unpredictable precipitation events, the possibility of an increased water-pulse regime, and high temperatures correlated with climate change, land managers struggle to implement restoration projects with high success rates (Hereford et al. 2016, Suazo et al. 2012).

3.1a Introductory site characteristics

The largest portion of Mojave Desert is characterized by two codominant shrubs, *Larrea tridentata* (creosote) and *Ambrosia dumosa* (white bursage). The creosote-bursage community covers nearly 70% of the Mojave (Lathrop & Rowlands 1983). The community occurs on valley floors in well-draining alluvial flats and gentle slopes below 1500 m (Thompson 2004; Brooks et al. 2007). Woody shrub cover is often low and minimal at around 5% to 30% (Vasek & Barbour 1977). When precipitation starts to exceed 183 mm, an ecotone gradually begins and is replaced with other plant communities, especially the blackbrush community (Beatley 1975). The Mojave Desert is a mostly winter precipitation desert with some rain falling as summer monsoons (Brooks & Pyke 2002). However, as one travels from east to west, the rainfall develops a more bimodal pattern, where a larger proportion falls during the monsoon season on July to September (Rowlands 1980).

The codominant species, and namesake of the community, are the creosote bush (*Larrea tridentata)* and bursage *(Ambrosia dumosa).* Other commonly occurring species include: prickly pear cactus (*Opuntia* spp.), goldenheads (*Acamptopappus sphaerocephalus),* Mormon tea (*Ephedra* spp.), California buckwheat (*Eriogonum fasciculatum*), rhatany (*Krameria* spp.), winterfat (*Krascheninnikovia lanata),* boxthorn (*Lycium* spp.), indigobush (*Psorothamnus* spp.), needle grass (*Achnatherum* spp.), Galleta grass (*Hilaria* spp.), and chollas (*Cylindropuntia* spp.) (Brooks et al. 2007). Common invasive plants include red brome (*Bromus rubens*), cheatgrass (*Bromus tectorum*), Mediterranean grass (*Schismus spp.)* and filaree (*Erodium cicutarum)*

(Brooks et al. 2007). Because of the vast range of the community concerning both geography and precipitation, the floristic species richness greatly exceeds this small listed portion of plants.

Desert pavement is a ubiquitous and conspicuous feature of arid lands, covering nearly 50% of natural desert in North America with varying clast sizes and mosaic tightness (Musick 1975; Quade 2001). They are often associated with alluvial fans, valley floors, and other ancient aqueous relict land formations in the Mojave Desert, occurring most frequently in the creosote-bursage community (Quade 2001). Desert pavements are considered a "complex association of landscape and hydrological elements" because the mosaic of clasts on the surface affect where water flows, where vegetation concentrates, and what vegetation can penetrate the pavement layer (Wood et al. 2005). Pavements are easily disrupted by shear forces such as wheels, hooves, and feet and are thus unlikely to survive a major construction disturbance. Biological soil crusts grow on the surface and between rock fragments of desert pavement areas. These assemblages of lichens, fungi, mosses, and other micro-life contribute to soil development, water conservation, nutrient cycling, erosion protection, hydrology, and vegetation distribution (Chiquoine 2016). Their single loss is the loss of an entire integral functional group to an ecosystem; a functional group that provides ecological services already notoriously slow in arid lands.

Artificial desert varnish has been used to recreate the aesthetics of a natural landscape. Desert varnish, which is formed through a very slow and complex process of microorganism (dematiaceous hyphomycetes, mycelial molds, bacteria, fungi) respiration as a hard patina of iron and manganese oxides (Taylor-George et al. 1983). Desert varnish is thought to absorb ultraviolet rays, providing additional shielding. The literature on the effects of artificial desert varnish is nearly nonexistent. Varnish is applied by precipitating iron and manganese compounds onto rock surfaces with a salt and an alkaline component (Elvidge & Moore 1980). Permeon, the specific artificial varnish used at the Fish Hatchery, was developed at Arizona State University. Because it was applied for aesthetic purposes, it is simply included as a subtreatment in the overall Fish Hatchery restoration treatment. Interesting further research could be measuring ambient air temperatures around the areas to see if albedo and other edaphic characteristics are affected.

3.1b Disturbance and Restoration

Increasing human populations in the Mojave necessitates development and infrastructure. Construction of pipeline corridors, cell phone towers, transmission lines, and new commercial and recreational roadways typically require the removal of the topsoil surface layer and vegetation. Roadway construction and the subsequent closing of defunct roads severely compacts soil. Webb and Wilshire (1980) found that after 51 years, long-lived perennials such as creosote colonized uncompacted land at 40%, while compacted areas remained at 3% colonized. Construction results in complete vegetation destruction except perhaps a few long-lived and propitiously rooted shrubs (Vasek 1979). In addition to destroyed vegetation and destroyed biological soil crusts, secondary effects of habitat fragmentation, reduced gene flow, and access to remote areas for illegal collections can slow the recovery trajectory of the disturbed ecosystem (Lovich & Bainbridge 1999). Pipeline corridors, despite recent efforts to implement more environmental measures, continue to affect ecosystems because of the extensive trenching and topsoil removal usually required. Negative abiotic impacts include churning soils, disturbing rock surfaces, and concentrating runoff and erosion (Lovich & Bainbridge 1999). The quantity of restoration projects following construction is increasing, however little is known about the effectiveness of the treatments and worse, the cost of such treatments is often not reported in literature (Kimball et al. 2015).

Outplanting is a common restoration technique among many global ecosystems. The planting of native plants, either from nursery stock or as transplants from undisturbed conditions, is beneficial in that it can immediately establish long-lived perennials and a seed bank, assuming survivorship. While a high survivorship is far from guaranteed, the addition of irrigation, shelters, or cages to protect from the harsh arid climate and herbivory are beneficial (Abella 2009; Abella et al. 2012). Native plants can grow in nurseries from seed, salvaged from areas that will be disturbed in the near future and transplanted from healthier and proximal areas, or as cuttings. Cuttings often work well for cacti. The disadvantages of outplanting involve a cost-benefit analysis. Outplanting nursery-grown plants incurs extra costs because of their maintenance and resource use. In addition, transport and labor costs can be high. Unpredictable weather patterns and herbivory may lead to very low survivorship rates. Examples of poor revegetation attempts include a 1983 highway outplanting project where after two years, almost all of the plants perished; another study by Brum and others (1983) reported a 0.3% and a 26% survival of transplanted seedlings (Lovich and Bainbridge 1999). Romney and others completed a successful attempt in 1979 with an 80% survival rate (Lovich & Bainbridge 1999). The first two years for most outplants are critical; most mortality of creosote seedlings occurs within this timeframe (Ackerman 1979). Irrigation and herbivory protection are usually ceased and removed after the initial critical period, although this depends on resources, funding, and probably, interest.

3.2 Methods

3.2a Study Area

This study was conducted within Lake Mead National Recreation Area, part of the Colorado River system in Clark County, Nevada (36° 3'52.94"N and 114°49'8.27"W), approximately 30 km nearly due east of Las Vegas, Nevada (Figure 1). Elevation of the plots range from 391-412 m. The 0.21-ha area is divided by the River Mountain Loop Trail and Lakeshore Dr. The site is located on an alluvial fan with one soil association consisting of Typic Torriorthents with sandy loam texture (Lato 1996). The plant community of the site is typical of low elevations in the Mojave Desert, *Larrea tridentata-Ambrosia dumosa* scrublands with desert pavement. Dominating features of undisturbed areas besides the seemingly monotonous vegetation include extensive swaths of desert pavement and small, vertically growing piles of biological soil crusts spread across the pavement mosaic.

The majority of rainfall in the region occurs during the cool months of fall and winter (October to April), however because of how far east the site is, a substantial portion falls during the summer monsoon months (Hereford et al. 2016). Las Vegas, for example, has 60% of its precipitation falling in winter (Abella & Newton 2009).

Three weather stations were located close to the Fish Hatchery: 1) Alan Bible Visitor Center (elevation 500.2 m, 6.3 km away); 2) Boulder City, NV US (elevation 762.0 m, 9.5 km away); 3) Willow Beach, AZ US (elevation 225.6 m, 25.3 km away). Weather stations are all close to the site, but the three stations did not provide climate data from June 2010 to April 2013. The next nearest weather stations in Henderson, NV USA were missing the same data.

Precipitation from 1996-1999 averaged 12.9 cm year^{-1} (Figure 2). The average high temperature was approximately 28.6° C and the average low was 15.6° C. Precipitation in 1999, the year of restoration, was 10.42 cm year^{-1}, followed by an even drier year of 5.64 cm year^{-1} in

2000 and 9.42 cm year^{-1} in 2001, which was the year the initial tally of outplant survivorship occurred. The high and low temperatures were also slightly elevated in these two years. Precipitation for the year before and the first year of monitoring (2005-2006) averaged 13.89 cm year^{-1}. Precipitation between 2018 and 2019 which includes the two final years of monitoring, was an annual average of 13.72 cm year^{-1}. According to Abella et al. 2007, the long-term (32 year) average was 14 cm year^{-1} using only the Willow Beach, AZ USA station. When I averaged the three closest stations, I found that the 23 year average (1996-2019) was slightly less at 11.01 cm year^{-1} with an average high of 28.6°C and an average low of 15.6°C (Figure 12).

In 1968, a water pipeline was constructed beneath the desert soils to provide water from Lake Mead to the city of Las Vegas and surrounding areas. The land above this pipeline had no restoration treatment post-construction. In 1998, a second water pipeline was constructed by the Nevada Water Authority.

In January and February 1999, the National Park Service implemented a biotic and abiotic restoration treatment to part of the 1998 disturbed pipeline corridor. The National Park Service then bladed both the treated and non-treated sections of the 1998 corridor and reapplied the upper 20 cm of topsoil after construction. Therefore, both unrestored and restored treatment plots do have an initial treatment of topsoil replacement, which has been implicated in helping native perennials establish (Kimball et al. 2015). The National Park Service spread out displaced rock, and hand-raked the soil surface to ensure an evenness to the topsoil layer to the treated 1998 corridor. They applied an artificial layer of desert varnish (Permeon™, Soil-Tech Inc., Las Vegas, NV) to rocks and the soil surface for natural color restoration and aesthetic appeal. The 1998 corridor was then outplanted with 96 *Larrea tridentata* seedlings, 12 *Ambrosia dumosa* seedlings, 9 *Opuntia basilaris* seedlings, and 2 *Senegalia greggii* seedlings. The planting treatment is detailed in Newton (2001). Four years later (2001), no planted *S. greggii* or *A. dumosa* survived, however *L. tridentata* (92%) and *O. basilaris* (100%) showed strong survival rates (Abella et al. 2007). 2006 marks the beginning of the monitoring period used in this study.

Four adjacent locations constituted four different plot designs: a 1968 disturbed, unrestored corridor; a 1998 disturbed, unrestored corridor; a 1998 disturbed, restored corridor, and an area undisturbed by construction activities (reference). The 1968 disturbed pipeline corridor provides a unique opportunity to measure natural recovery over a longer period of time. Most restoration projects only focus on short term monitoring, which gives little information as to how successful the treatment was and how quickly disturbed desert lands recover. In order to better understand whether treatment, year, or an interaction between the two is having a larger effect, Section I includes statistics without the 1968 pipeline plots. Perennial measurements for the 1998 unrestored, treatment, and reference plots include three monitoring years (2006, 2007, and 2008) and annual measurement include 2018 and 2019. Section II consist of an analysis between all four plot types (e. g. including unrestored 1968 corridor) to help determine whether succession occurred more quickly because of the treatment, or if the area naturally recovered after 50 years. Figure 3 visually describes the experimental design of Section I (Figure 3, top) and Section II (Figure 3, bottom).

3.2b Field and Laboratory Sampling

Initial field sampling data was gathered between 31 August and 25 October 2006. Field sampling in 2018 was measured on 3 April 2018 and field sampling in 2019 was measured on 29 March 2019. For temporal continuity and comparison, monitoring protocol followed that of Abella et al. 2007. The 2018 and 2019 monitoring seasons were designed to capture spring annuals, while the monitoring in 2006 captured some autumn annuals and identifiable dead annuals were recorded. These measurements represent two very different annual communities and annual community measurements in 2018 and 2019 could not be directly compared to 2006. Annual metrics do contribute to overall ecosystem health and were monitored and discussed when possible. Differences include the time of monitoring, which in 2018 and 2019 was designed to capture live spring annuals. For all annual metrics, only 2018 and 2019 results were analyzed and for perennials, all three years (2006, 2018, and 2019) were analyzed.

In 2006, a 30 × 70 m sections were delineated in the centers of each of the four areas. Within each of these four sections, seven 10 × 10 m plots were established. Using simple random sampling, three plots were selected in each area for sampling. Within each plot, six 1 × 1 m subplots were located at the plot corners and at the midpoints (5 m) of the southern and northern plot lines. Areal percent cover of each plant species was visually estimated and assigned a cover class (modified from Peet et al. 1998). Plots were then surveyed for areal percent cover for species that did not occur in the subplots to determine total species richness. The count of each perennial species was recorded to determine density. Nomenclature follows United States Department of Agriculture guidelines. To compare soils among the controls and various treatment plots, the top ten centimeters of soil was collected in an interspace between shrubs (>1m away from a shrub) and underneath a live perennial shrub. Soil was collected in four regions underneath the shrub, halfway between the main stem and canopy edge and combined. In the absence of a long-lived perennial in the plot such as *L. tridentata,* an alternative, often shorter-lived "pioneer perennial" such as *A. dumosa* or *Encelia farinosa* was used (Vasek 1979). If alternative plants were used, they constituted a large proportion of species diversity within the plot.

Between the monitoring seasons of 2018 and 2019, one of the plots within the treatment area was destroyed by further construction activities. For 2019, a new plot was established and monitored in the area directly adjacent (within a couple of meters) of the destroyed plot. Because the new plot was part of the restored area, I determined it was important to have as much data on the area as possible. In addition, there is only a sample size of three plots per treatment, so its omission would bring the sample size down to two plots and only 200 m^2. The unrestored area occupies about 0.13 hectares of land and the treatment area covers about 0.11 hectares of land. With three plots in each disturbance type, between 25-30% of the total area was monitored. The large sample proportion is another reason why a new plot was created. The newly created plot was within the variation of all existing plots over time.

Air dried <2 mm fractions of soil were analyzed for pH, total P and K, total C and N, and texture in 2007. Soil was tested for pH, total C and N.

3.2c Data Analysis

To specifically compare planting treatment effects and year of assessment on native and exotic cover and species richness of life history groups and perennial plant density within the 1998-disturbance, data from the 1998-disturbed/unrestored and 1998-restored plots and undisturbed references plots were compared using generalized mixed models (PROC GLIMMIX, SAS version 9.4 2013), with plot type (disturbed/unrestored, disturbed/restored, and undisturbed/reference) and year (2006, 2018, and 2019) set as fixed effects and plot as a random effect. A second analysis was conducted which included the 1968-unrestored disturbance. Native and exotic cover and species richness of life history groups and perennial plant density were compared among plot types and year of assessment using a similar model described above, with plot type (1968-disturbed/unrestored plots, 1998-disturbed/unrestored plots, 1998-

disturbed/restored plots, and undisturbed/reference plots) and year (2006, 2018, and 2019) set as fixed effects and plot as a random effect.

For each model above, a sensitivity analysis was performed to assess the effect of replacing the missing disturbed/unrestored plot during the 2019 assessment. Results from the models with the replacement plot and with the plot set as missing values were compared. Conclusions did not qualitatively differ; results using the models with the replacement plot are reported. For all models, if necessary, variables were either transformed to improve normality (cover, $\log_{10}+1$ or arcsine-square root transformed; species richness, square-root transformed), or, if normality was not improved by transformations, distributions were assessed and applied to the model (lognormal distribution for cover; Poisson distribution for species richness) and goodness-of-fit tests were examined. Post-hoc tests with Tukey-Kramer adjustments for multiple comparisons were used to further assess significant effects or interactions ($\alpha \leq 0.05$).

3.3 Results

3.3a Section I: Perennial and annual results excluding the 1968 pipeline corridor

To ascertain whether treatment, year, or treatment × year interaction effects are affecting results of the restoration treatment, the 1968 was excluded from this portion of analyses. There are three plot types: three 1998-pipeline unrestored, three 1998-pipeline restored (treatment), and three reference plots. Perennial metrics include measurements from 2006, 2018, and 2019. Annual metrics include measurements from 2018 and 2019, only.

Perennial cover significantly differed by year (Figure 3).The year 2006 had less cover than either 2018 or 2019. Cover percentages increased between each monitoring year. During the twelve-year monitoring hiatus of 2006 and 2018, cover increased approximately sevenfold. This averages to an increase of only 0.56 fold each year, if divided equally across years. During the single year hiatus between 2018 and 2019, perennial cover increased approximately 1.7 fold. Perennial species richness remained homogenous in both time and space. In fact, no effects were detected for almost all of the life history groups. Upon further investigation of the exact perennials present within each plot type per year, the only species that was detected only once was *Tiquila latior*. This suggests that between years and treatments, the species richness was relatively consistent and representative of the creosote-bursage plant community.

To determine what was providing significant differences in perennial cover, perennials were parsed into more specific life history groups: forbs, grasses, woody perennials (shrubs), and cacti. I found that shrubs were the most substantial component affecting perennial cover because other perennial life history groups were minimal.

The only perennial grass present at the site was *Dasyochloa pulchella*, which did not occur very frequently. *D. pulchella* was only found in unrestored plots and had less than a 0.05% cover. The sole perennial forb, *Cuscuta californica*, is a holoparasitic vine that inhabited one undisturbed plot in 2019. No significant effects for cactus cover were detected; the only cactus species was *Opuntia basilaris.*

Following from the previous results, it is unsurprising that the "shrubs" category, which here includes plants classified as shrubs, subshrubs, and subshrub-shrubs, varied significantly by year but not by treatment as well (Figure 4). Shrub cover followed a remarkably similar pattern to total perennial cover. Cover increased over the years, with 2006 being distinctly lower than the high percentages found in 2018 and 2019. Much of this increase occurred between 2018 and 2019, although the two groups did not significantly differ from one another.

Shrubs were further divided based on their stature. In order of increasing size, the shrub types are forb-subshrubs, subshrubs, subshrub-shrubs, and shrubs. I found that typically larger shrubs types were undergoing significant changes within the monitoring period. However, the other shrub groups consisted of only a few species each. The only forb-subshrub present was *Tiquilia latior*, which was only detected in 2006 unrestored plots. Subshrubs, which included *Stephanomeria pauciflora*, did not vary within the 1998 pipeline sites. *S. pauciflora* is often implicated as a short-lived and early-colonizing species found in disturbed areas (Vasek 1979). I expected it to potentially appear in either treatment or unrestored sites, depending on how quickly the recovery trajectory is moving. Subshrub-shrubs included *Bebbia juncea* and *Encelia farinosa.* Subshrub-shrub cover was not significant; however, the unrestored plots typically had slightly higher values. Subshrub-shrub richness was significant. The two species were only found in disturbed plots (e.g. reference and unrestored). This does suggest that the early-colonizers are only appearing in the disturbed area without restoration. Long-lived shrub cover, which included the two codominant species and a third *Ambrosia* species, significantly differed by year only (Figure 4). The year 2019 had more cover of the three species than either 2006 or 2018. In order to determine whether this statistical significance was an artifact of the 2018-2019 plot replacement, the three species were counted in each plot; I determined that the statistic is indeed significant regardless of plot replacement.

Perennial cover and all of its components were affected only by year and not by treatment. Long-lived shrubs increased in cover between the longer period of time between surveying (2006-2018), while smaller shrub species increased between 2018 and 2019.

The density of perennials that are considered as early colonizers, regardless of their life spans, exhibited the most significant changes, especially between year × treatment interactions. Of the six perennial species occurring at the site, five of them are considered to be early colonizers (Vasek 1979), although only three significantly changed.

Larrea tridentata is not only the codominant and longest-living species in this desert but is also posited to have the unique role as both a colonizer and climax species (Vasek 1979). *L. tridentata* density increased by plot type as hypothesized (Figure 5). Restored plots had intermediate values between the unrestored and reference plots, although the restored plots are more similar to the unrestored plots. Reference plots had about twice the density per hectare as restored plots and about three times the density compared to unrestored plots. *L. tridentata* density moderately increased over the years as well.

Ambrosia dumosa, the other codominant of the plant community and its relative *Ambrosia salsola* showed moderate effects concerning year (Figure 5). Both have conflicting literature suggesting whether they are short- or long-lived (Vasek 1979; USDA Plants) but share roles as early colonizers. *A. salsola* is much more likely to be identified as a colonizer in the creosote-bursage community than its relative.

Figure 5 includes the year × treatment interactions for the two codominant species. It should be noted that year is moderately significant for *A. dumosa* and for *L. tridentata* and plot type is significant for *L. tridentata* only. The interactions were not significant.

The short-lived *Stephanomeria pauciflora*, a prevalent early colonizer, and *Encelia farinosa*, another common colonizer, both significantly changed. *Stephanomeria pauciflora* occurred in unrestored, disturbed plots only. *E. farinosa* exhibited year and treatment interactions (Figure 6). In 2006, eight years after restoration, low density characterized all plot types. By 2018 and 2019, disturbance plot density increased. Restored plots in the later years were not significantly different from restored 2006 results. In unrestored plots, however, a major increase occurred between 2006 and 2018, followed by a slight and non- significant decline. Reference plots never had any individuals of these species among years. Between 2006 and 2018 monitoring, *E. farinosa* increased from an estimated 33 individuals hectares to over 5500 individuals ha^{-1} and occurred exclusively in the unrestored plots (Figure 6). While this appears to be an absurdly and unbelievable change, it is possible: seedlings and mature shrubs were not separated during data collection.

Shrubs of all statures and longevities exhibited significant year × treatment interactions (Figure 7a). The density of shrubs in the three plot types did not differ in 2006, however it increased in 2018 and 2019. There was a significant increase in shrub density among unrestored plots specifically. Between 2018 and 2019, a slight decrease among unrestored plots suggests some plant die-off between the two years. Shrubs that are considered early-colonizers increased (Figure 7b). There was a significant year × plot type interaction, which included all of the aforementioned species except *A. dumosa*. During the first surveys, early colonizing shrub density did not significantly differ among plot types. However, in 2018 and 2019, this shrub species group had significantly more individuals in the unrestored plots compared to treatment or reference plots. Early colonizing shrubs densities remained at none to very low in reference plots.

Annual plant communities are subject to precipitation patterns, including both the amount of rain and the timing of the rain event. Annuals were examined on an exotic versus native basis. All non-native plants are exotic annual forbs and grasses. Interestingly, when comparing the three plot types of the 1998-disturbed corridor, there appear to be more effects due to treatment than due to year (Table 2), although only the 2018-2019 monitoring years were included. Precipitation or some edaphic factor may have caused these results.

Exotic cover was highest in undisturbed plots (Figure 7a). Restored plots had the least percent cover among all plot types and all plot types are significantly distinct (Figure 7a). About seven typical exotic species occurred across the sites and years. When parsed between grasses and forbs, I found that exotic grass cover was driving the exotic cover effect since annual forbs exhibited no statistical significance. There were no exotic forb effects detected in 2019 and exotic forbs were not significant by plot type. Exotic grass cover exhibited the exact increasing pattern as total exotic cover, although unrestored and restored plots did not significantly differ from one another (Figure 7b).

All native annuals were forbs. While there are forb subgroups with differing longevities, I pooled them into the same group because the harsh climate of this particular area tends to kill them within the same year as germination. Most of the species at low elevations like the Fish Hatchery have a biennial habit only because they are perennials at much higher elevations and conditions happened to be favorable (Beatley 1973). The only detected species the United States Department of Agriculture (USDA) considers as an annual-biennial was *Lepidium densiflorum.*

The two species classified as annual-perennial were *Eriogonum inflatum* and *Chamaesyce polycarpa.* Native annuals of any longevity, which includes true annual (<1 yr.), annual-biennial, and annual-perennial groups, significantly differed by treatment. Restored plots were intermediate between unrestored and reference plots with reference plots having the highest cover. This could be considered logically as an "increasing health" trend positively related to the restoration of plot types.

3.3b Section II. Perennial and annual results including the disturbed and unrestored 1968 pipeline corridor.

The addition of the 50-year old pipeline to the analyses presents a unique opportunity to compare trajectories of both an older and a more recently disturbed and unrestored area to each other, to the trajectory of an outplanting restoration treatment, and to an undisturbed reference (Figure 13). The species composition of plots with and without the 1968 pipeline, especially the richness of early colonizing shrubs or exotic annuals, indicates whether the plant community is still in a disturbed state or is closer to a reference (undisturbed) state.

The unrestored 1998 pipeline is referred to as "unrestored" for the rest of this chapter. The restored 1998 pipeline is referred to as "restored" or "treatment." Undisturbed plots are referred to as the "reference" plots. The unrestored 1968 corridor is interchangeably referred to as "pipeline" or "1968 pipeline."

Plot type had many significant effects on perennial species metrics (Table 3). More species were detected in both unrestored plot types (e.g. 1968 and 1998), with detection highest in pipeline plots (Figure 8). There were no significant changes in all plot types between 2006 and 2018, although perennial percent cover increased in all plot types except the pipeline (Figure 8). 1998-disturbed plot types did not significantly differ from the reference, but unrestored plots were most similar to the pipeline plots. Within years, reference plots were the next most similar to pipeline plots. In 2006, the restored area had very little cover like the other types, but by 2019 had significantly increased by 7%. The biggest increase occurred between 2018 and 2019. The reference area remained relatively constant over time with just slight cover increases, suggesting stability and long-term perennial growth. Perennial species richness exhibited a similar trend to cover in that the 1968 pipeline (group b) had the most species, followed by the unrestored (ab) plots and is lowest in the restored (group a) and reference (group a) areas (Figure 8).

To better understand which life history groups are reacting to show changes between years and plot types, I again parsed the perennial groups out into separate life forms: grasses, forbs, shrub subgroups, and cactus. Like the previous section, there was only one perennial grass, *Dasyochloa pulchella* and one perennial forb-subshrub, *Tiquilia latior. D. pulchella* grows in the 1968 pipeline and the unrestored plots and *T. latior* was found in unrestored plots only. The sole cactus species, *Opuntia basilaris,* occurred in all four plot types. While not significant, it had the greatest cover in the undisturbed and the restored areas, as well as a 100% survival rate in 2001 (Abella et al. 2007).

The cover of all shrub types combined was significantly lowest in 2006 and increased through all three monitoring years. Subshrubs, which consisted of the species *Porophyllum gracile* and *Stephanomeria pauciflora,* had low overall percent cover in all plots, but were significantly higher in 2006 pipeline plots only. The 2006 cover declined with time; pipeline plots still had the highest subshrub cover. Subshrub-shrub cover was not significant and never exceeded 5.0%. Subshrub-shrubs, consisting of *Bebbia juncea* and *Encelia farinosa*, grew on both unrestored plot types (1968 and 1998) and were never detected on any reference plot in any year. Long-lived shrub species increased throughout the monitoring years. Unlike total shrub cover combined, each year represents a homogenous group.

Overall, two perennial trends emerge regarding the four plot types and three monitoring years: cover and species richness tended to be highest in the pipeline plots and means typically increased temporally.

Most perennial density measurements were significant by year × treatment. Exceptions included the cactus *O. basilaris* and subshrub *P. gracile.* There was a single shrub individual in all years on one pipeline plot of *P. gracile* and *O. basilaris* had about one to three individuals in all plot types. Total shrub density was highest in the pipeline plots every year. Each plot type peaked in 2018 and then decreased slightly by 2019. The reference area stayed consistent throughout the years.

Among shrubs, early colonizers, except *A. dumosa* and *L. tridentata,* frequently varied in composition through time and space. Most colonizer densities were low or zero in 2006 aside from pipeline plots. By 2018, the density of unrestored and pipeline plots greatly increased and then slightly fell. The stable reference plots maintained a value of zero throughout all years. Reference plots would have no necessity for early colonizers unless a severe, potentially natural stress event (e.g. fire, flash flood) affected the area. Treatment plots seemed to match reference responses: densities stayed relatively low and stable throughout the monitoring period.

Among specific early colonizers, the most influential factors were the year 2019 and pipeline plot type. *Bebbia juncea*, a medium-lived perennial, only appeared on pipeline plots at an approximate density of 120 plants hectare^{-1}. *Encelia farinosa* was not observed on reference plots and exhibited the highest values in the final two years between both unrestored plot types. *Ambrosia salsola* occurred on pipeline plots in all years with a maximum density in 2018 that was double that of 2006 and 2019. In 2018, a small amount was detected on 1998 unrestored plots that disappeared the next year. To substantiate this familiar behavior, a study at an abandoned Nevada ghost town also had a higher *A. Salsola* density on compacted soil after 51 years than in reference areas (Webb & Wilshire 1980). *Stephanomeria pauciflora* varied among years, but only appeared on unrestored and pipeline plots. These individual densities clearly illustrate that early colonizers are still the main shrub type on pipeline and unrestored plots.

Larrea tridentata density was lowest in 2016 among all plot types with moderate significance and also significantly differed by treatment (Figure 11; Table 3). Reference plots had about double the density as restored plots, triple the density of unrestored plots, and nearly twenty times the density of pipeline plots. Treatment plots were more similar to reference conditions than both unrestored plots types were, although this similarity is not significant. *Ambrosia dumosa* exhibited a year × treatment interaction like the majority of the other shrub populations (Figure 11). Most individuals grew in pipeline plots followed by unrestored plots; the highest density occurred in 2018. Literature suggests that *Ambrosia* is a much more prevalent colonizer than *Larrea*, and so a higher density would be expected on disturbed over undisturbed plot types (McAuliffe 1988).

Density data, like cover data, highlights that the 1968 pipeline perennials community still indicates a disturbed area. Treatment plots are more similar to reference plots than other types, except with a little less temporal stability.

The statistics for annual communities among the four plot types only reflect the years 2018 and 2019 to avoid inter-seasonal variability. All non-native plants were exotic forbs and grasses. In all significant metrics, the effects were either, year, treatment, or year and treatment, but never an interaction between the two.

Exotic cover was significantly highest in the pipeline plots, followed by the reference plots like Section I results. The lowest covers were in 1998 plot types. The pipeline is similar to Section I results in that the more shrub cover or greater density present, the more exotic cover there is. This alone is not enough evidence to determine that shrub cover is a primary factor controlling exotic growth. In addition to plot type effects, combined exotic cover was significantly higher in 2019 than 2018 with both life history groups (grasses, forbs) separately displaying the same results; however, exotic forbs and grasses differed in their response to plot type (Figure 12). Exotic forb cover was significantly highest in the unrestored plots, followed by the treatment plots, very low in the reference, and undetected in the pipeline plots. Exotic grass cover, on the other hand, follows the same trend as overall exotic cover. Exotic grasses are the most influential component of overall exotic cover.

Native annual forb percent cover and species richness were significantly higher in 2019 than 2018 (Figure 12). The reference and treatment area had significantly more forbs than both unrestored areas. The reference sites have the best composition of native annuals, followed by the treatment area. The two undisturbed areas show the worst compositions (Figure 12).

3.4 Discussion

Evaluating the success of a restoration project has long been mostly a question of "how?" and "why?" Oftentimes, the success of a project is unclear and makes it difficult to support spending resources on restoration at all. Nevertheless, ecologists and managers must set certain goals and then examine metrics to determine if these goals were achieved or that the restored area is similar to reference conditions. If, many seasons later, these goals are met, the project is generally considered a success. Unfortunately, the goals for each project are often subjective, difficult to achieve, or vary in their success between years. The most commonly assessed metrics determine the status of vegetative structure, diversity, and ecological processes (SER 2004). The first and second metrics are the easiest and most popular methods. Evaluating processes requires a lot of time and resources that are usually not budgeted for (Ruiz-Jaen & Aide 2005). The Society of Ecological Restoration International (SER) distributed a primer in 2004 listing nine attributes that, if partially achieved, consider a project successful or on a trajectory to success. All attributes are compared with a reference, or undisturbed conditions. Some of the attributes suggest recovered sites should be resilient, have a decent assemblage of native species, contain all functional groups necessary for stability, and possess sustainable and self-sufficient ecosystem function that is fully integrated and will continue indefinitely (SER 2004).

The Fish Hatchery treatment plots, in many respects, can be considered moving towards the status of "a successful restoration project" since its genesis. When comparing the three plot types without the 1968 pipeline corridor (Results, Section I), positive impacts on perennial and annual plants are present. In multiple cases, the restored plots hold an intermediary position between the unrestored plots and the reference, but usually have more similarities to either one or the other. For example, *L. tridentata* is both a climax and colonizer species (Vasek 1979), a codominant, the longest-lived of the area, and one of the chosen outplanted species. Within the three plot types, it possessed an intermediate mean density between the reference and unrestored, leaning towards the unrestored. In other metrics, treatment plots were more similar to the reference such as native annual cover among the four plot types described in Section II. Of the 96 outplants, four years after the restoration treatment we know that 92% of them survived. These approximate 88 outplants survived the critical two-year period where most perennial outplants in the desert perish (Ackerman 1979). It is likely that the survival of these plants to maturity will eventually replenish the seed bank and perhaps establish even more creosote permitting favorable future climatic conditions.

A couple of issues arise when considering the site decently "restored." For example, there are very few perennial forbs and grasses in the area, even within the reference. While grasses are not a very common component of the plant community and typically make up less than 3% of a community's total cover (Beatley 1975; Wallace & Thomas 2008), they were present around the site but simply were not colonizing on the study areas. Perennial grasses tend to green up much later in the season than forbs and both can be an integral food source during the hot summer months (Wallace & Thomas 2008). Perennial forbs and grasses may be more subjectable to climatic differences because they have herbaceous material that dies and regrows each season (Beatley 1975), but the presence of only one species in the earliest year is problematic. The absence of these two life history groups, especially since they are in the larger ecosystem matrix, suggest that the site is not fully integrated into the larger landscape and that the vegetative structural functionality to make it a diverse ecosystem is not fully present. For example, Vasek (1979) considered an ecosystem functionally and structurally heterogenic because it contained plants that utilized different photosynthetic pathways. Also, the structurally diverse ecosystem would have plants of all statures and shapes, even in arid ecosystems (Vasek 1979). There were only five species of forb-subshrubs, subshrubs, and subshrub-shrubs combined and the other approximate fifty species classified as either forbs, shrubs, the single cactus species, or an exotic. Whether this is an issue depends largely on what part of the species-area curve the site itself would theoretically land on. Also uncertain are the effects of the desert pavement and biological soil crust in the reference. These features naturally cause changes in hydrology and a decrease or redistribution of vegetation because overflow water is siphoned elsewhere (Wood et al. 2005).

Annual cover consists mostly of native rather than invasive species, suggesting that a characteristic assemblage of flora at the Fish Hatchery is native and representative of the low-elevation desert scrub community. Cover and richness did vary by year, however inter-annual variability is expected and with only two years of data, has yet to cause concern that this particular life history group is struggling. If after several years, exotic cover spikes or native richness plummets, other restoration actions may be necessary to shift the trajectory back towards health. There was typically more cover on reference and restored plots, and among other metrics, this tended to make them similar. The restored plots are also still similar to the two 1998 disturbed plot types. This may suggest that while restored plots are inching towards reference-like conditions, there is still noticeable progress to be made before the site can be considered reference-like and recovered.

The 1968 pipeline corridor does not meet the majority of the SER attributes and after fifty years cannot be considered as reference-like. It has more attributes of a disturbed ecosystem than a mature and undisturbed one. The pipeline is not particularly resilient or self-sufficient for its TSR because there are major fluctuations in perennial plant levels between years that is unobserved in the reference and restored plot types. While percent cover is decent and fauna have been observed utilizing the area, vegetative composition is mainly made up of early colonizers and lacks sufficient numbers of dominant species like *L. tridentata.* Extremely low cover of *L. tridentata* on the pipeline plots suggests that the recruitment of new *L. tridentata* seedlings will be minimal even fifty years following a major construction disturbance in this plant community. Vasek (1979) did not encounter even a single seedling until nearly four years after a disturbance. There was nearly twenty times the number of *Larrea* individuals on reference plots compared to pipeline plots. Restored plots were more similar to reference conditions than the other plot types, although this similarity was not significant. The few *L. tridentata* present on the pipeline plots were very large, and are possibly just relics that escaped the 1968 construction activities. It appears that with the pipeline plots, species richness fluctuated most often. Upon the exclusion of the 1968 pipeline, there were more changes in plot type rather than year, although interactions were often significant at the pipeline. A road separates the 1968 plots from the others but the distance is short. The pipeline tended to have more types of the early-colonizing species than any other plot type.

The outplanting of *L. tridentata* and *Opuntia basilaris* helped the trajectory of the restored sites become more similar to the reference, rather than follow the 1968 pipeline corridor path. Of course, it is impossible to know exactly what trajectory it would have followed sans restoration; however, several discussed similarities between the unrestored and pipeline plots suggest that they would be alike.

One of the SER attributes suggests that threats from surrounding areas should be reduced or eliminated. Within the National Park System, this is becoming a more impossible task each year because of increasing recreation and visitation. The Fish Hatchery is situated near one of the most highly visited areas in the park and will likely always feel the effects associated with frequent human use.

Little information is known about artificial desert varnish other than it restores rocks to a more natural-looking state. It made the area aesthetically more pleasing to look at compared to the unrestored plots. Future research could determine whether the dark color alters the ambient infrared heat index and causes perennial effects. Belnap and Eldridge (2001) suggested that the trampling of dark-colored soil crusts and subsequent exposure of light soils increased albedo. These changes in surface color could lead to regional climate pattern changes (Belnap & Eldridge 2001). It is possible the loss of the dark color of varnish would have a similar effect as soil crust loss. The spreading of rocks and the reapplication of topsoil likely had an effect, but these abiotic treatments could not be separated to determine individual effects. Topsoil replacement has been shown to bolster the success of perennial germination and survival (Abella et al. 2015). The layer of desert pavement and biological soil crust on the reference area seemed correlated to low shrub cover, but increased native annuals cover, which has been observed before (Musick 1975, Quade 2001). In other research sites that compared interspaces to under-shrub spaces, native cover was most prominent in, surprisingly, interspaces. Desert pavement can often be such a tight mosaic that it is difficult for seeds to catch and small roots to anchor. Thus, interspaces often have less annual cover than under shrubs. Several native annual species have uniquely developed the ability to colonize zones like this as opposed to following the elevation gradient (Abella & Newton 2009). One particular species found in great abundance was *Chorizanthe rigida.* While desert pavement cannot be restored and the natural recovery process will take a very, very long time, executing an outplanting in a fashion that mimics the surrounding reference land could allow them to ecologically connect more easily.

Perennial cover, especially larger or longer-lived shrubs increased over the monitoring years. Shrub cover is much more variable compared to simple natural revegetation at a recovering site (Webb et al. 1987). Something quite interesting is the increase between 2006 and 2018 compared to the increase between 2018 and 2019. The magnitude of change of cover is relatively the same despite the substantially different amount of years. When examining precipitation patterns of the past several decades, the time between 2006 and the 2018 monitoring period shows less than average annual precipitation. Average or above-average rainfall is necessary for a large-scale recruiting event to occur for many climax species. It is essential that long-lived shrubs begin to colonize the area, in part because there are a number of other perennials that require the created microhabitat to germinate and survive into adulthood (McAuliffe 1988). If the plant community continues to be dominated only by short-lived shrubs, the chances of long-lived shrub recruitment will be greatly reduced for the foreseeable future.

Density data suggests that the 1968 pipeline community still represents a very disturbed area, while treatment plots are more similar to reference plots except with less temporal stability. The 1968 and 1998 unrestored plots shared similar fluctuations in cover and density levels at a higher magnitude than the treatment plots. The disturbed plots are sensitive to outside influences, rendering them much less self-sufficient from year to year. For example, the 1968 pipeline plots have high shrub cover, but this shrub cover is primarily made up of early colonizers like *Encelia, Ambrosia,* and *Stephanomeria.* Small-statured shrubs like these colonizers typically dominate disturbed, unrestored areas (Vasek 1979). Through the monitoring years, drastic changes in density and cover levels occurred. This suggests that not only is the community still far from its climax, but that it lacks resilience and self-sufficiency. On the other hand, reference plots are characterized by low shrub cover because of the layer of desert pavement, and few to no colonizer species. It remains at the same cover and density levels throughout the study. The 1998 restored plots have species similar in number and type to the reference, but the levels fluctuate with year, suggesting that treatment plots have a similar resilience and self-sufficiency of both unrestored plot types. The treatment plots are an intermediate in many ways between unrestored and reference plots.

3.5 Conclusion

It is absolutely necessary to state clear goals prior to beginning any restoration project, especially since this gauges the cost-effectiveness of the restoration (Kimball et al. 2015). It is quite clear that the pipeline 1968 corridor has very high shrub cover. If the goal was to reduce soil erosion and dust in the air pollution, the site may be considered restored because plants are keeping sediments from blowing away; however, the corridor is nothing like reference conditions. If the project goal was to mimic and maintain the surrounding landscape within the revegetation area, then the project could be considered as almost a failure and also almost a success for this short time since restoration. The restored plots tended to mirror reference conditions more often, suggesting that outplanting did alter the trajectory towards natural. The restored plots and the other disturbed areas lacked the stability over time that the reference showed. The disturbed areas may not be resilient enough to withstand periodic stress, or, perhaps there is a high turnover of plants indicating that long-lived, dominant species have not yet colonized the area. Once the two corridors can maintain a population of *L. tridentata* and associated species for several years, the area can be considered restored. With restoration, this may take several decades. Without restoration, it will take, at the very least, well over fifty years.

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3.7 Tables and Figures

Table 3.1 Significant effects for perennial plant variables measured at the Fish Hatchery restoration site in Lake Mead National Recreation Area, NV USA.

Years analyzed included 2006, 2018, and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at $\alpha \leq 0.05$ and moderately significant at $\alpha \leq 0.10$. See Appendix II for all Fish Hatchery Type III Tests.

Variable	Significant effects	Num DF	Den DF	F Value	Pr > F
Perennial cover	Year	2	12	14.81	<0.001
Shrub cover	Year	2	12	12.21	0.001
Long-lived shrub cover	Year	2	9	9.03	0.007
Subshrub-shrub species richness	Treatment	2	6	8.73	0.017
Encelia farinosa density	Year × Treatment	4	12	11.03	<0.001
Larrea tridentata density	Treatment	2	6	9.39	0.014
Stephanomeria pauciflora density	Treatment	2	6	9.37	0.014
Total shrub density	Year × Treatment	4	12	10.77	<0.001
Early colonizer density	Year × Treatment	4	12	10.47	<0.001
Early colonizer & *A. dumosa*	Year × Treatment	4	12	11.37	<0.001
A. dumosa & *L. tridentata* density	Year	2	12	7.56	0.008
A. dumosa & *L. tridentata* density	Treatment	2	6	9.32	0.015

Table 3.2 Significant effects for annual plant variables measured at the Fish Hatchery restoration site in Lake Mead National Recreation Area, USA.

Years included are 2018 and 2019 only. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at $\alpha \leq 0.05$ and moderately significant at $\alpha \leq 0.10$. See Appendix II for all Fish Hatchery Type III Tests.

Variable	Significant effects	Num DF	Den DF	F Value	Pr > F
Exotic cover	Treatment	2	6	91.5	<0.001
Exotic grass cover	Treatment	2	6	26.53	0.001
Native annual forb cover	Treatment	2	6	8.21	0.019
Native annual (<1 yr) forb cover	Treatment	2	6	7.93	0.021

Table 3.3 Significant effects for perennial plant variables measured at the Fish Hatchery restoration site in Lake Mead National Recreation Area, USA.

Years included *are* 2006, 2018, and 2019. Treatment types included a 1968-disturbed unrestored corridor (1968 pipeline), *a* 1998-disturbed unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment*:* *outplanting*), and undisturbed reference conditions. Results were considered ignificant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

Variable	Significant effect(s)	Num DF	Den DF	F Value	Pr > F
Perennial cover	Year × Treatment	6	16	3.65	0.018
Perennial species richness	Treatment	3	8	11.53	0.003
Perennial grass species richness	Treatment	3	8	4.8	0.034
Forb-subshrub species richness	Year × Treatment	6	16	4	0.012
Long-lived shrub cover	Year	2	11	16.68	<0.001
Subshrub cover	Year × Treatment	6	16	5.14	0.004
Subshrub species richness	Treatment	3	8	9.87	0.005
Subshrub-shrub species richness	Year × Treatment	6	16	5.42	0.003
Shrub cover	Year	2	16	11.99	<0.001
Shrub species richness	Year × Treatment	6	16	2.92	0.041
Ambrosia dumosa density	Year × Treatment	6	16	3.17	0.03
Bebbia juncea density	Treatment	3	8	17.5	<0.001
Encelia farinosa density	Year × Treatment	6	16	9.66	<0.001
Ambrosia salsola density	Year × Treatment	6	16	4.89	0.005
Larrea tridentata density	Year	2	16	3.02	0.077
Larrea tridentata density	Treatment	3	8	14.94	0.001
Stephanomeria pauciflora density	Year × Treatment	6	16	44.41	<0.001
Total shrub density	Year × Treatment	6	16	8.67	<0.001
Early colonizers density	Year × Treatment	6	16	9.74	<0.001
Early colonizers & *A. dumosa* density	Year × Treatment	6	16	8.33	<0.001
A. dumosa & *L. tridentata* density	Year	2	16	9.61	0.002

Table 3.4 Significant effects for annual plant variables measured at the Fish Hatchery restoration site in Lake Mead National Recreation Area, USA.

Years included 2018 and 2019. Treatment types included a 968-disturbed and unrestored corridor, 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Results were considered significant at α≤0.05 and moderately significant at α≤0.10.

Variable	Significant effect(s)	Num DF	Den DF	F Value	Pr > F
Exotic cover	Year	1	8	45.75	<0.001
Exotic cover	Treatment	3	8	17.76	<0.001
Exotic forb cover	Year	1	8	10.59	0.012
Exotic forb cover	Treatment	3	8	4.96	0.031
Exotic forb species richness	Year	1	8	12	0.009
Exotic grass cover	Year	1	8	42.57	<0.001
Exotic grass cover	Treatment	3	8	10.17	0.004
Native forb cover	Year	1	8	7.6	0.025
Native forb cover	Treatment	3	8	9.08	0.006
Native forb (<1 yr) cover	Year	1	8	9.16	0.016
Native forb (<1 yr) cover	Treatment	3	8	8.97	0.006
Native forb (<1 yr) species richness	Year	1	8	7.21	0.027

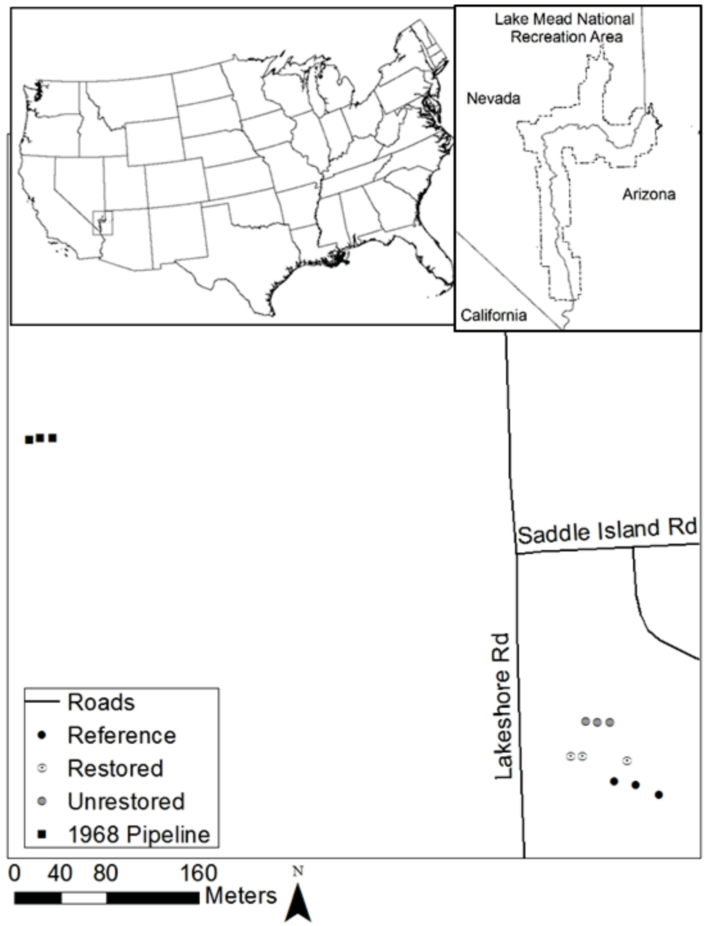


Figure 3.1 The Nevada Fish Hatchery is located in Lake Mead National Recreation Area. Lake Mead National Recreation Area is in the southwestern United States and follows the Colorado River system. Lake Mead National Recreation Area encompasses parts of Arizona and Nevada; the fish hatchery study is located only on the Nevada side. The 1998 and reference plots are on the east side of Lakeshore Road and are split by the River Mountain Loop trail, which was installed after plot installation. The pipeline 1968 plots are separated from the 1998 plots by Lakeshore Road.

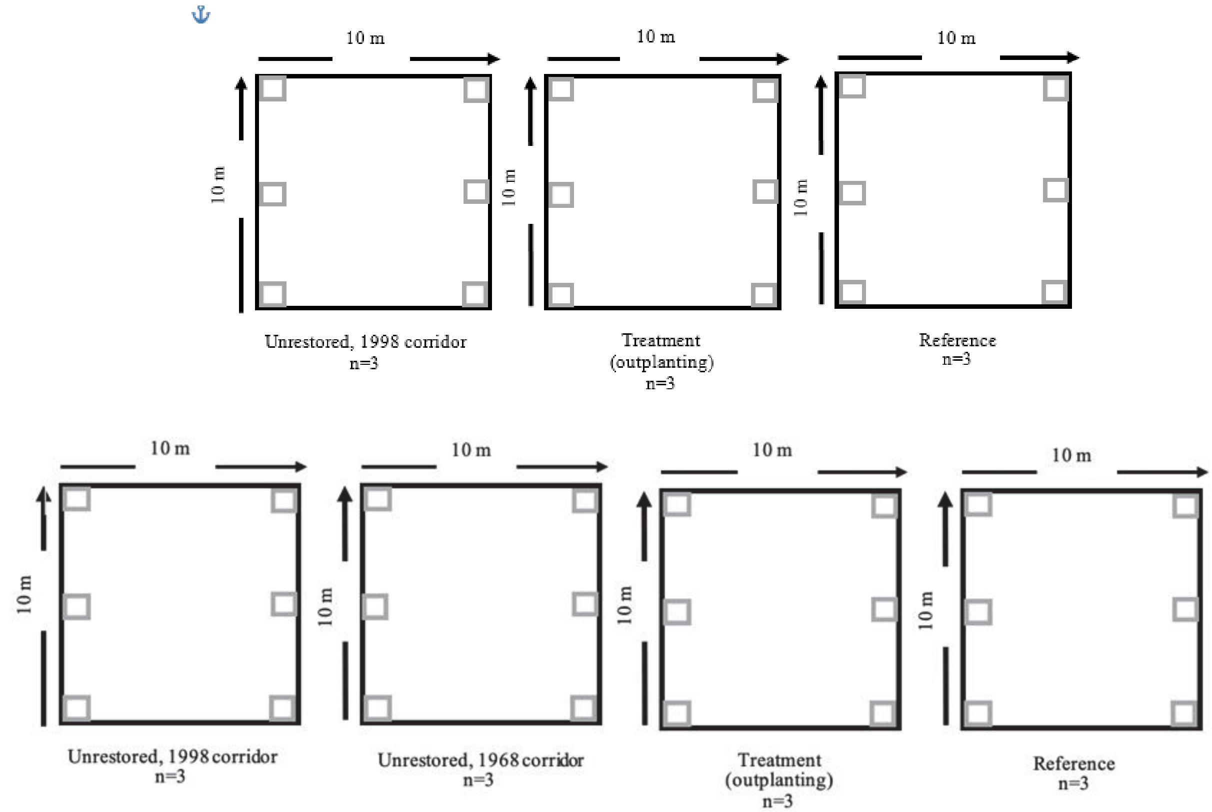


Figure 3.2 Sampling schematic and experimental design for the Fish Hatchery project site in Lake Mead National Recreation Area, Nevada, USA.

Disturbance (water pipeline installation) occurred in 1968 and 1998. Restoration included outplanting and artificial desert varnish spraying in 1998 only. The black squares indicate the whole plot scale (100 m_2). Grey squares indicate the six nested 1 m × 1 m subplots in standard locations at the four corners of the whole plot and at the midpoints along two opposite north-south axes within the whole plot. Plants that were not captured within the six subplots were recorded and assigned a percent at the whole plot level. Top: section 1 sampling schematic that does not include the 1968 pipeline corridor. Bottom: Section 2 sampling schematic including all four plot types.

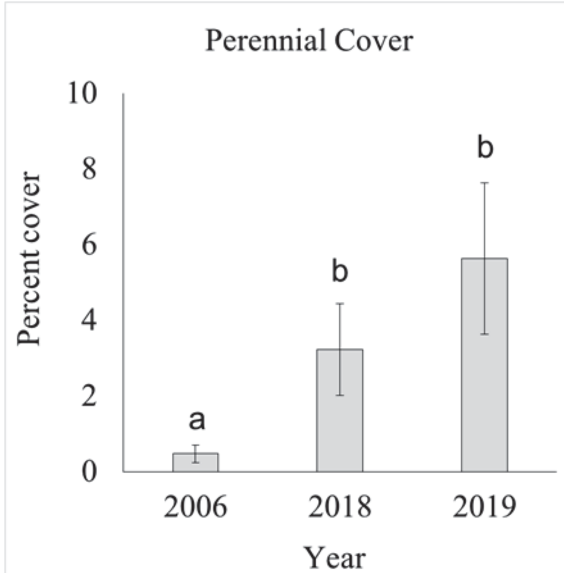


Figure 3.3 Perennial cover among 1998-corridor plots.
Total perennial cover increased over the monitoring period, however 2018 and 2019 did not have significantly different cover percentages. Different letters indicate significant groupings at $\alpha \leq 0.05$. Years analyzed included 2006, 2018, and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at $\alpha \leq 0.05$ and moderately significant at $\alpha \leq 0.10$. See Appendix II for all Fish Hatchery Type III Tests.

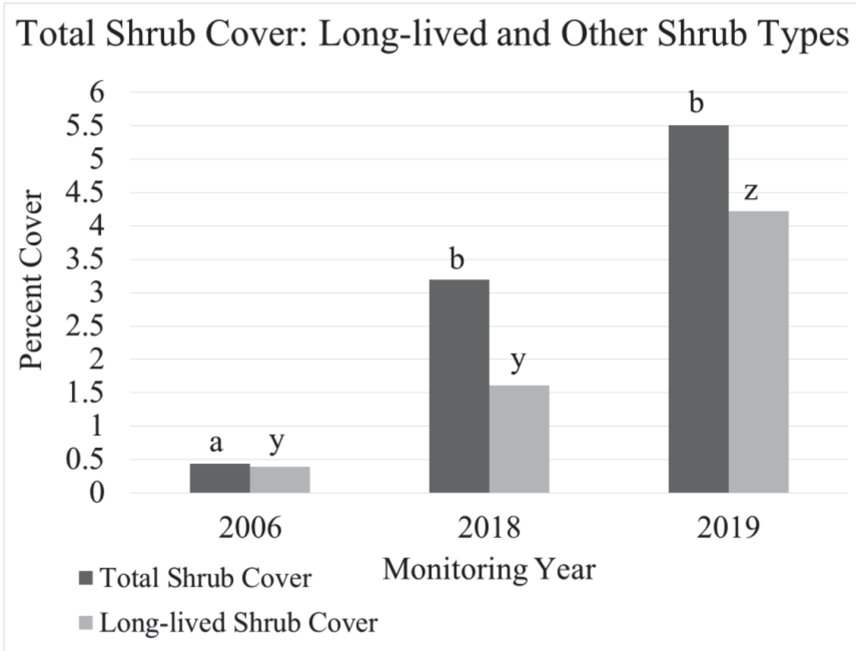


Figure 3.4 Shrub cover among 1998 disturbed corridor plots and undisturbed plots.
Total shrub percent cover plot^{-1}, which includes subshrubs, subshrub-shrubs, and shrubs, increased throughout the monitoring years. It follows a remarkably similar pattern to total perennial cover in Figure 4. When considering long-lived shrubs only, 2006 had significantly less cover than the following monitoring years. Different letters indicate significant groupings at α≤0.05. Years analyzed included 2006, 2018, and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

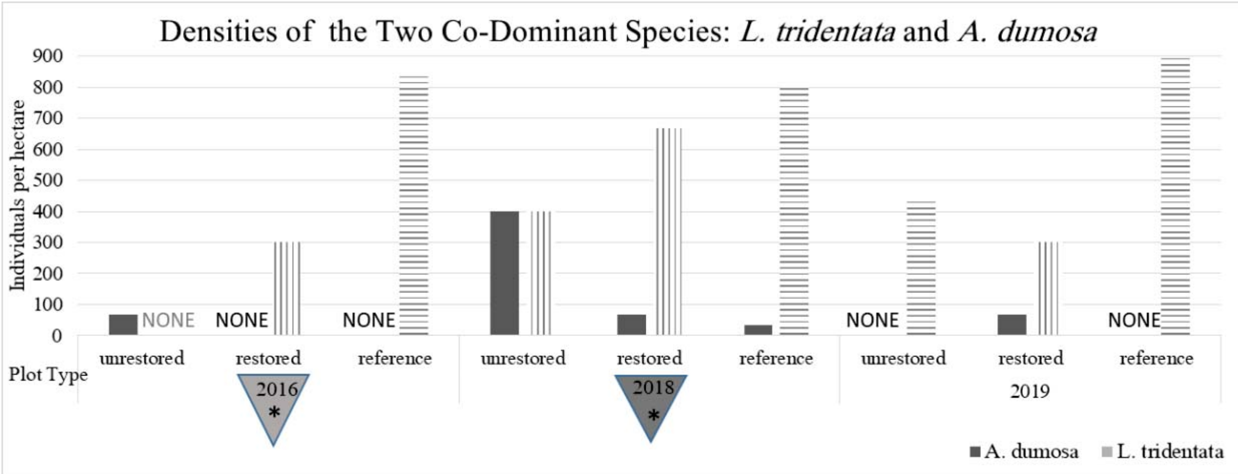


Figure 3.5 the densities of the two codominant species, *A. dumosa* and *L. tridentata* on 1998 disturbed corridor plots and undisturbed plots.

A. dumosa (dark grey) was moderately significantly highest in 2018 compared to other years (see asterisk in dark grey triangle). *L. tridentata* (light grey) was moderately significantly lowest in 2016 (see asterisk in light grey triangle). *A. dumosa* was not significant by plot type. *L. tridentata* had significantly more individuals hectare^{-1} in the reference plots (horizontal bars) compared to the unrestored and restored plots (vertical bars). Year × Plot type interactions were not significant for either species. Years analyzed included 2006, 2018, and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

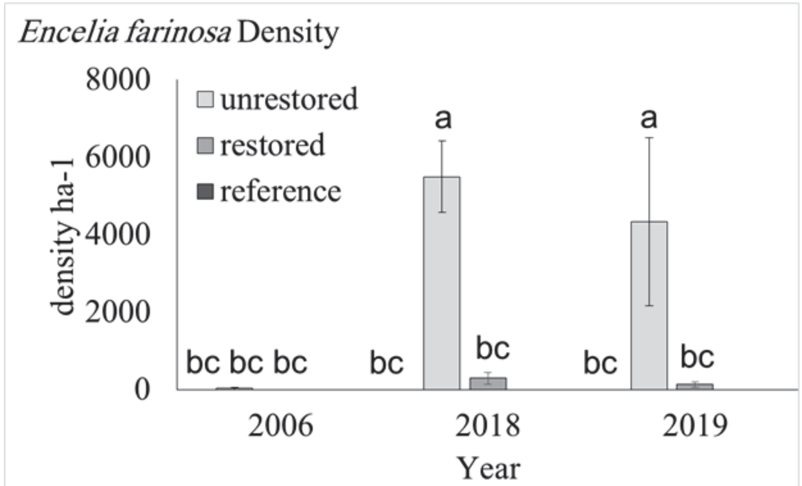


Figure 3.6 The density of Encelia farinosa increased over time in disturbed plots.
A substantial increase occurred in unrestored plots. The species was never detected in the reference area. Different letters indicate significant groupings at α≤0.05. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

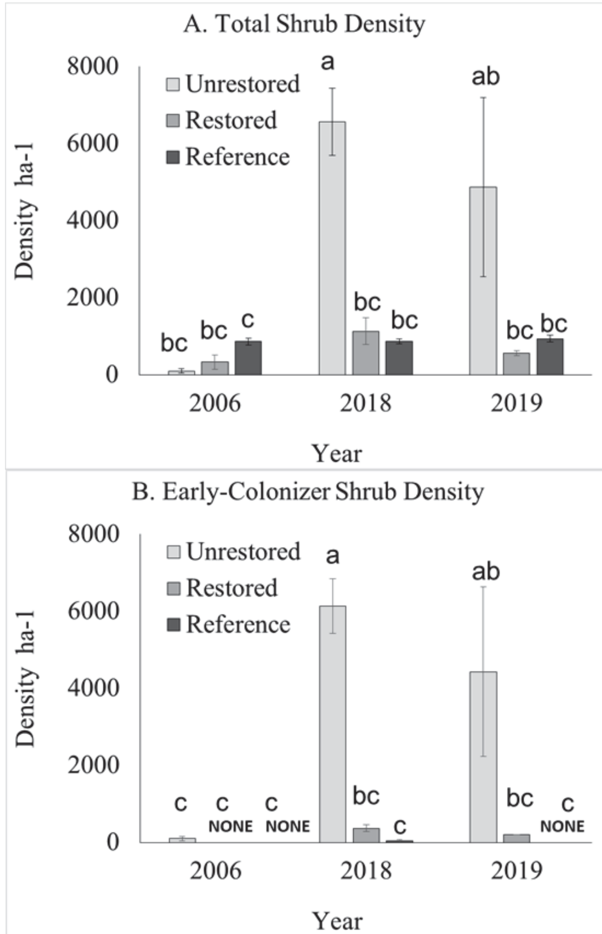


Figure 3.7 Densities of total and early-colonizer shrubs among three monitoring years and three plot types. A) Total shrub density increased in all plot types over the years, with unrestored plots exhibiting the largest increase. B) Early colonizer shrubs exhibited a similar trend to total shrubs in that they greatly increased in unrestored plots. They were scarce in 2006, 8 years after restoration. Reference densities remained low. Different letters indicate significant groupings at α≤0.05. Years analyzed included 2006, 2018, and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

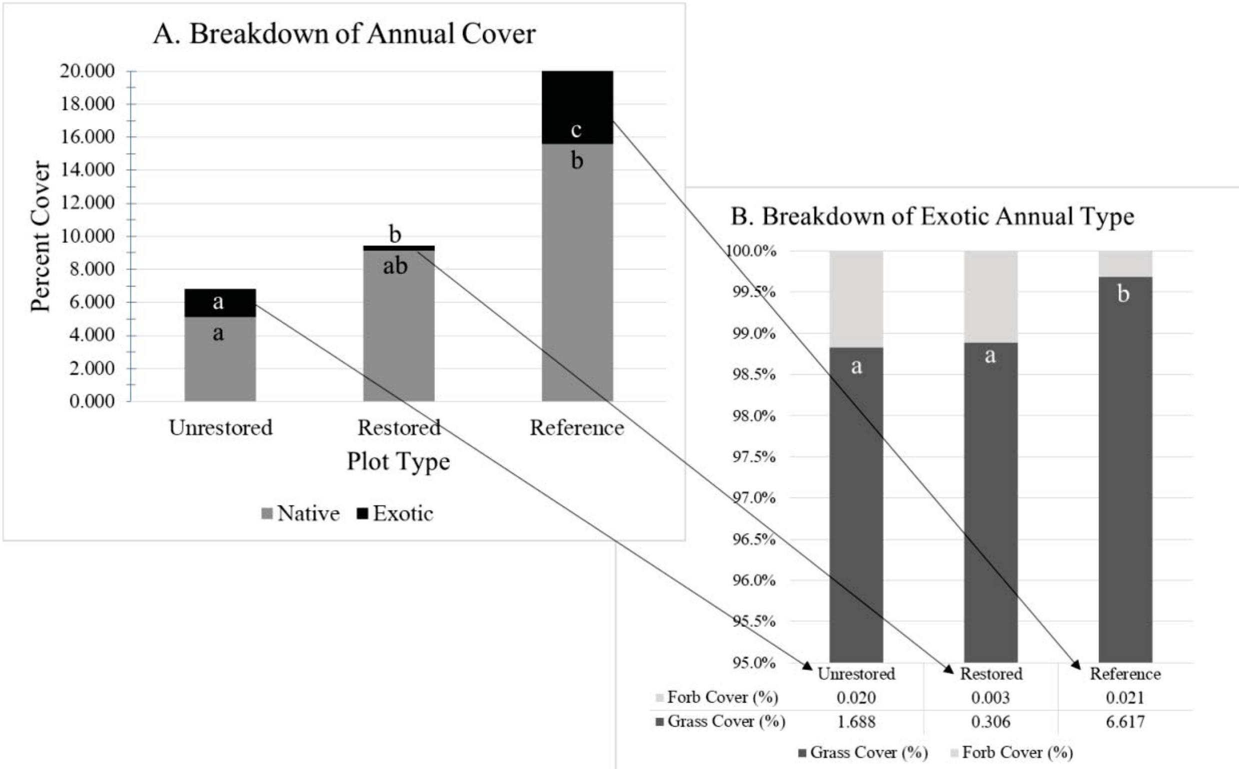


Figure 3.8 the 2018 and 2019 annual cover breakdown by plot type and nativity among three 1998-pipeline plot types.

A) Native annual cover (grey) was higher in every plot type compared to exotic annual cover (black). The native cover in restored plots was similar and in between in value of the unrestored and reference plots. B) Out of exotic annuals, the majority of them were grasses and not forbs. Y-axis begins at 95% and percentages represent the proportion of the total exotic annual cover of the plot type. Different letters indicate significant groupings at $\alpha \leq 0.05$. Years analyzed included 2018 and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at $\alpha \leq 0.05$ and moderately significant at $\alpha \leq 0.10$. See Appendix II for all Fish Hatchery Type III Tests.

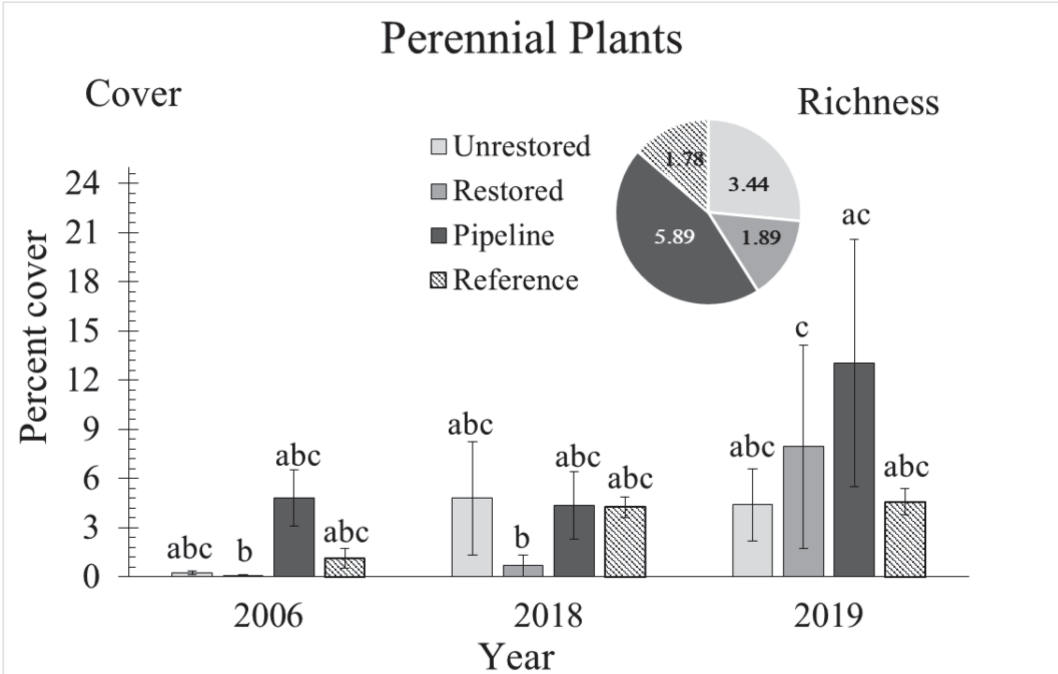


Figure 3.9 Perennial plant percent cover and species richness among three years at four plot types. Bar: Pipeline plots typically had the highest cover values. The greatest changes among plots besides the reference, which, maintained stability, occurred between 2018 and 2019. While every plot type except pipeline plots increased slightly between 2006 and 2018, none was significant. Pie: the restored (group a; SE ± 0.2) and the reference (group a; SE ± 0.278) plots are very similar in value and differ from the pipeline (group b; SE ± 0.455) plots. The unrestored (group ab; SE ± 0.475) plots are related to all plot types. Different letters indicate significant groupings at α≤0.05. Years analyzed included 2018 and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

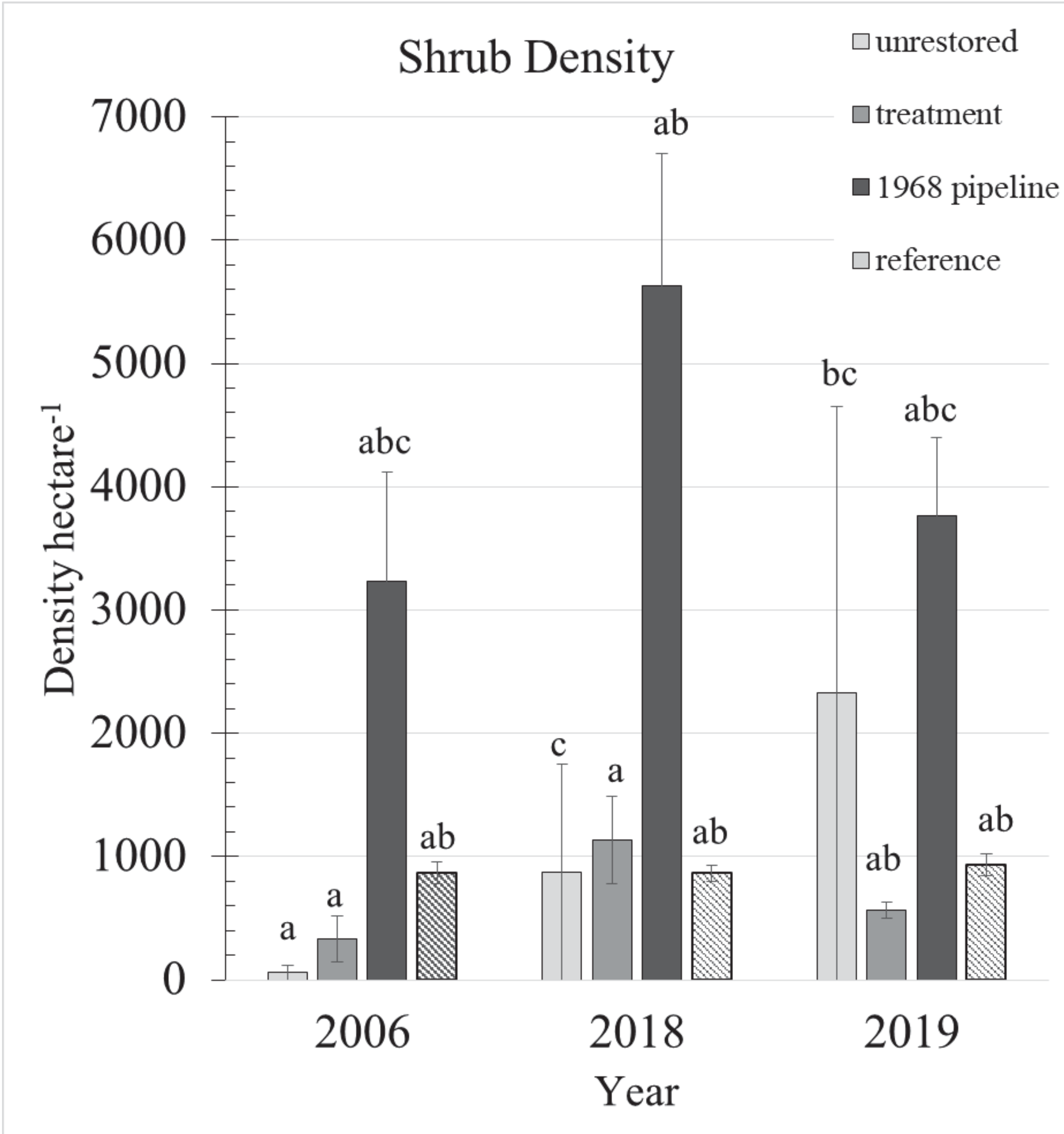


Figure 3.10 Shrub (all shrub subgroups) density among four plot types and three years.
Pipeline plots tended to have the highest cover and greatest variance while the reference remained at an even level throughout the years. Different letters indicate significant groupings at α≤0.05. Years analyzed included 2018 and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

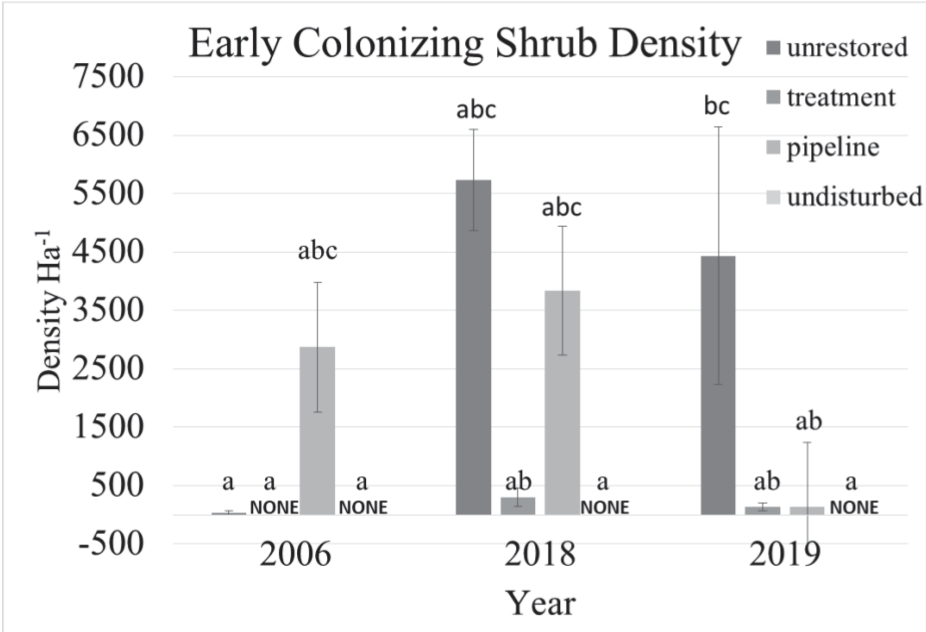


Figure 3.11 Densities per hectare of colonizing shrubs on four plot types.

The reference remained stable throughout the years. Both the unrestored and pipeline plots showed extreme fluctuation. Treatment plots remained relatively low and stable. Different letters indicate significant groupings at α≤0.05. Years analyzed included 2018 and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

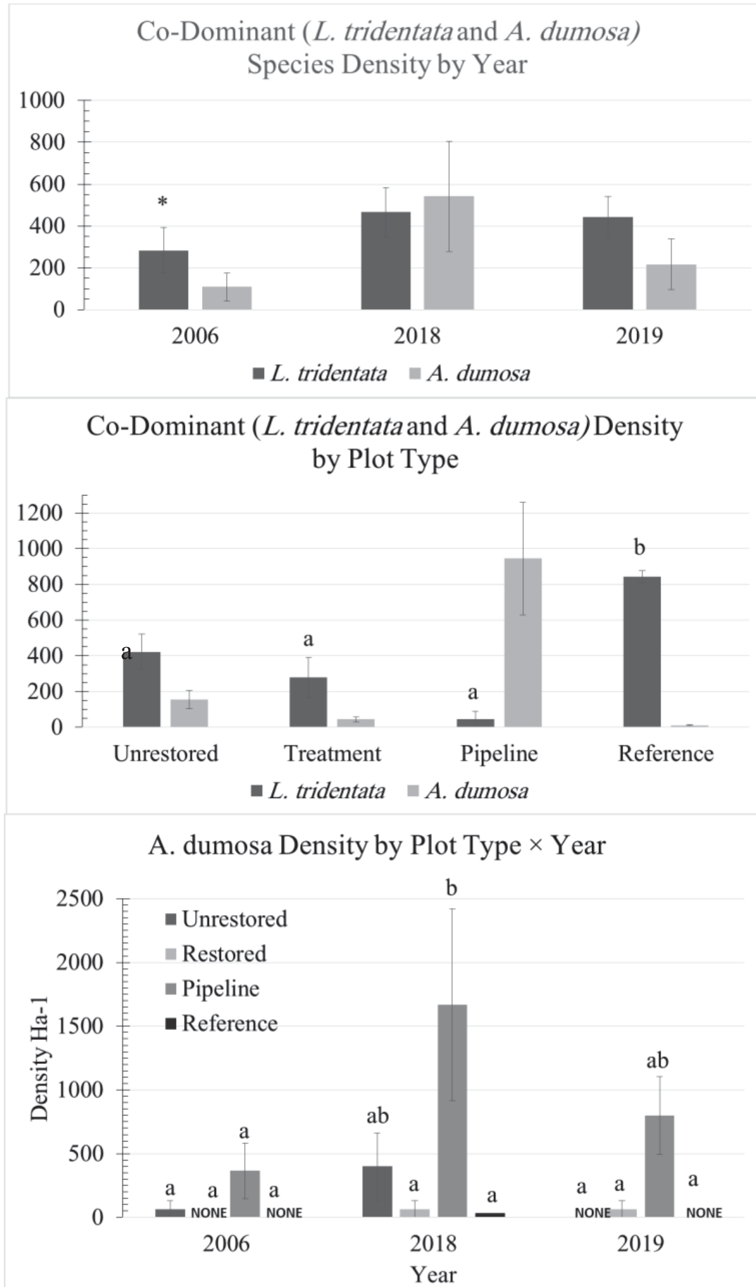
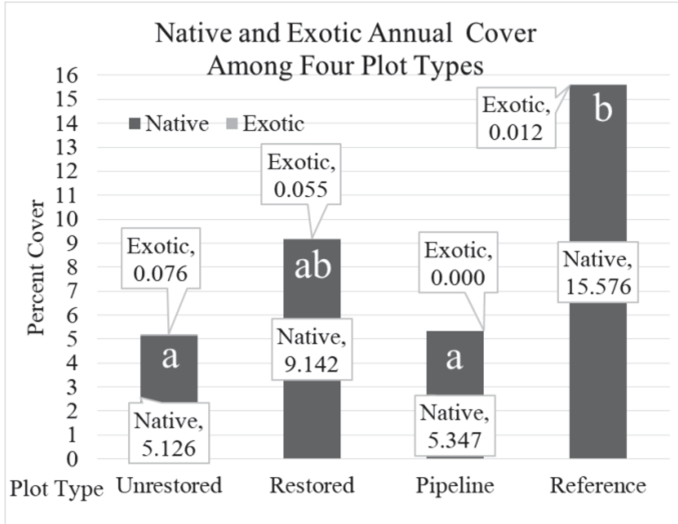


Figure 3.12 Codominant Density.

Top: *L. tridentata* is moderately significantly less in 2006 compared to 2018 and 2019. *A. dumosa* is always less than *L. tridentata* except in 2018. Middle: There are significantly higher densities in the reference plots compared to any other plot type. Interestingly, the unrestored area has more than the restored area. Bottom: *A. dumosa* is significant by year × plot type interactions. It tends to fluctuate greatly and be highest in the pipeline plot types. The unrestored plots do have some number and show variability as well. Different letters indicate significant groupings at α≤0.05. Years analyzed included 2018 and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.



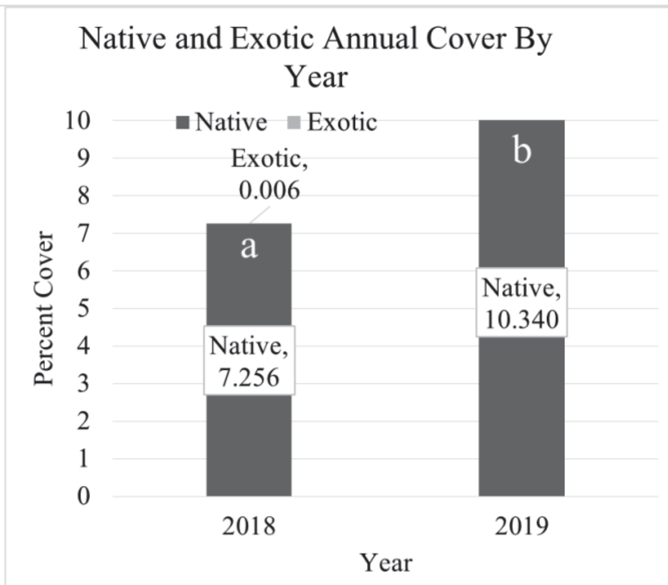


Figure 3.13 Annual Cover based on nativity for four plot types in two years at the Fish Hatchery, NV USA. Top: Native cover was much more prevalent than exotic cover. It was highest in the reference and restored areas, suggesting that they are similar. The restored plots, however, are still similar to the unrestored plots and so there may be some restoration effect that still needs to take place. Bottom: Native cover was predominant over exotic cover. There was more native cover in 2018 than 2019. . Different letters indicate significant groupings at α≤0.05. Years analyzed included 2018 and 2019. Treatment types included a 1998-disturbed and unrestored corridor (unrestored), a 1998-disturbed restored corridor (treatment), and undisturbed reference conditions. Main effects and interaction effects were tested for significance. Results were considered significant at α≤0.05 and moderately significant at α≤0.10. See Appendix II for all Fish Hatchery Type III Tests.

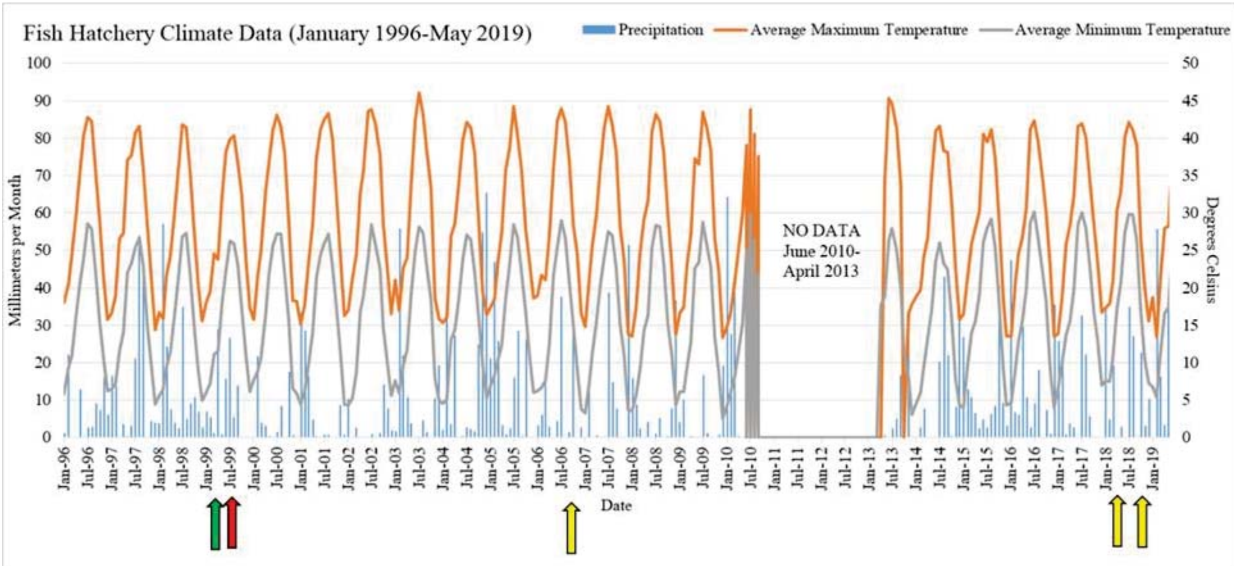


Figure 3.14 Fish Hatchery climate data from three years pre-restoration to current time.
Primary y-axis is mean monthly precipitation per year (mm). Secondary y-axis is daily temperature in degrees Celsius. Green arrow denotes approximate commencement of restoration.

Appendix 1: Site-Specific Methods, Results, and Statistics

A1.1 Fiber Optic Cable

Background and Study Area

Within Mojave National Preserve at an average elevation of 981 m (35° 9'7.62"N, 115°46'59.76"W), an AT&T fiber optic cable was constructed in the late 1990s (Figure A1). Between 2000 and 2001, part of the area was revegetated with *Larrea tridentata*. The *L. tridentata* outplants were then caged with wire mesh to deter herbivory. Evidence of DriWater®, a slow-release irrigation gel placed in a plastic tube near the root mass of an outplant, was found in some parts of the site. The cages remained on the plants as of 2019 sampling and were used as an identifier for the outplants in the treatment plots.

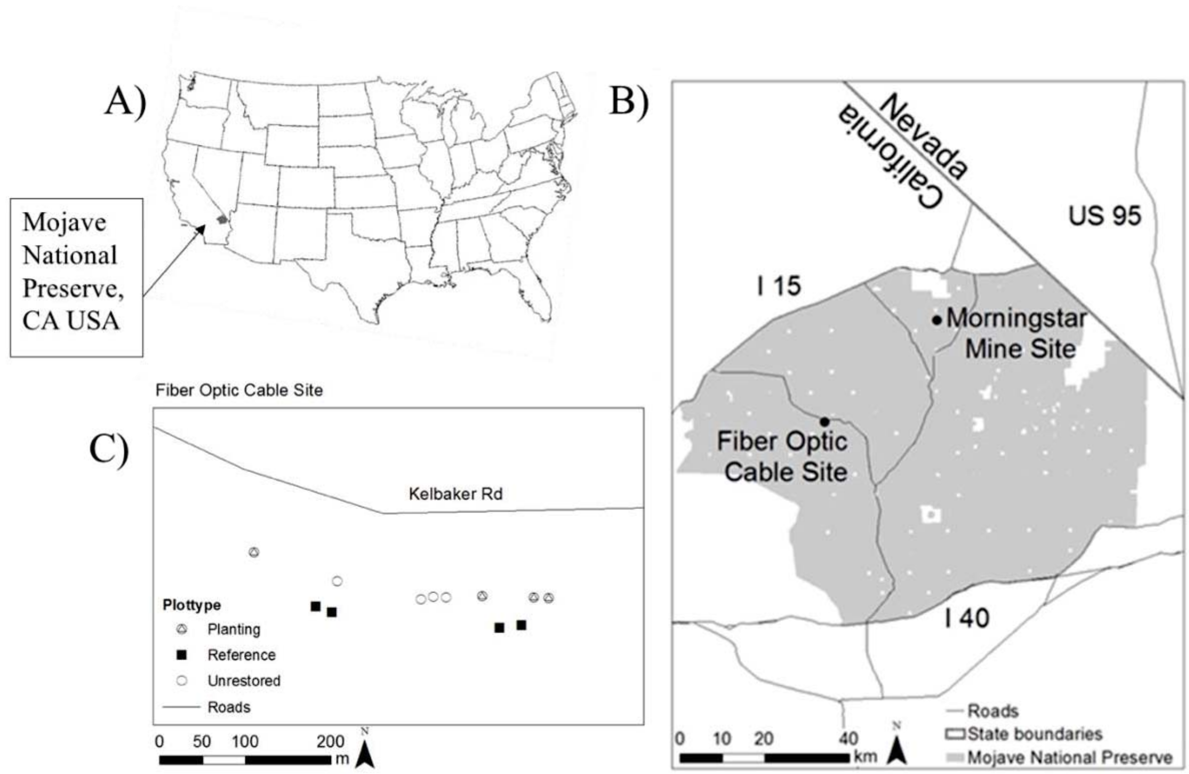


Figure A1. Location of the Fiber Optic Cable.
A) The Mojave National Preserve is located in southeastern California, USA. B) The Fiber Optic Cable was installed near the central part of the Mojave National Preserve (35.152, -115.785). C) The site is on the south side of Kelbaker Rd. Between the plots, a small dirt road runs north and south. While this road is still in use, a barrier of wooden stakes protects the restoration site to either side from off-road traffic.

Because of the limited amount of information and no available previous monitoring data, plot types had to be identified and delineated by other means, rendering multiple year measurements impossible. The areas of unrestored, treatment, and reference plots were easily determined despite a dearth in records. Treatment plots still had outplant caging; undisturbed plots had surface soil blatantly denuded of older, larger vegetation that ran directly adjacent to the treatment sites. The reference sites showed no indication of disturbance because of the large long-lived shrubs and large boulders. Following delineation, thirty plots total were established on 24 February 2018. Ten plots were established in the undisturbed area adjacent to the Fiber Optic

Cable (reference), ten plots were established in an area that did not receive restoration following disturbance (unrestored, control), and ten plots were established in the outplanting restoration (treatment) area. Plots with and without restoration were readily identifiable based on the presence or absence of the metal cages intended to prevent herbivory. All plots had slope aspects of roughly 300-335° and slope gradients for all plots ranged from 1-7°.

To accommodate the linear configuration of the disturbance, plot sizes were 12.5 m × 8 m (100 total m^2), with a minimum of a 1-m buffer between plots. A random subset of four plots per plot type was obtained using a random number generator. Plots were measured on 03 and 04 May 2019 during peak annual bloom.

Data Collection

For each plot, two types of subplots were measured: shrub microsites and interspace microsites (Figure A2). Analyses for differences resulting due to subplots were interpreted; however, to maintain the large-scale theme of this chapter, microsites types were combined to make up "subplot" effects for the following results. In all three of the plot types, six interspaces subplots were placed at 2.5 m and 5 m in from the long axis and at 3, 6, and 9, m along the long axis. If a *L. tridentata* was growing in the area where the subplot was supposed to be measured, the interspace subplot was randomly adjusted to lie at least 1 m away from the shrub but still as close as possible to the original coordinates. For treatment plots, twelve subplots were measured: six caged *L. tridentata* microsites and six interspace microsites. For unrestored plots, six interspace subplots and any available *L. tridentata* were measured. In most of the plots, no or very few *L. tridentata* were available for measure. Because no other shrub was chosen as a surrogate for the outplant, the number of shrub subplots ranged from 0-3 among all measured treatment plots. For reference plots, six *L. tridentata* were randomly chosen for measuring. Using a 1m^2 quadrat, percent cover of all species present was estimated using cover classes (1=<0.01%, 2=0.01-0.1%, 3=1-2%, 4=2-5%, 5=5-10%, 6=10-25%, 7=25-50%, 8=50-75%, 9=75-95%, 10=>95%, modified from Peet et al. 1998). Following the measurement of all subplots, species that had not yet been encountered in subplots were recorded and given a cover class at the whole plot level. All perennial species within the plot were counted to determine the density of each perennial species.

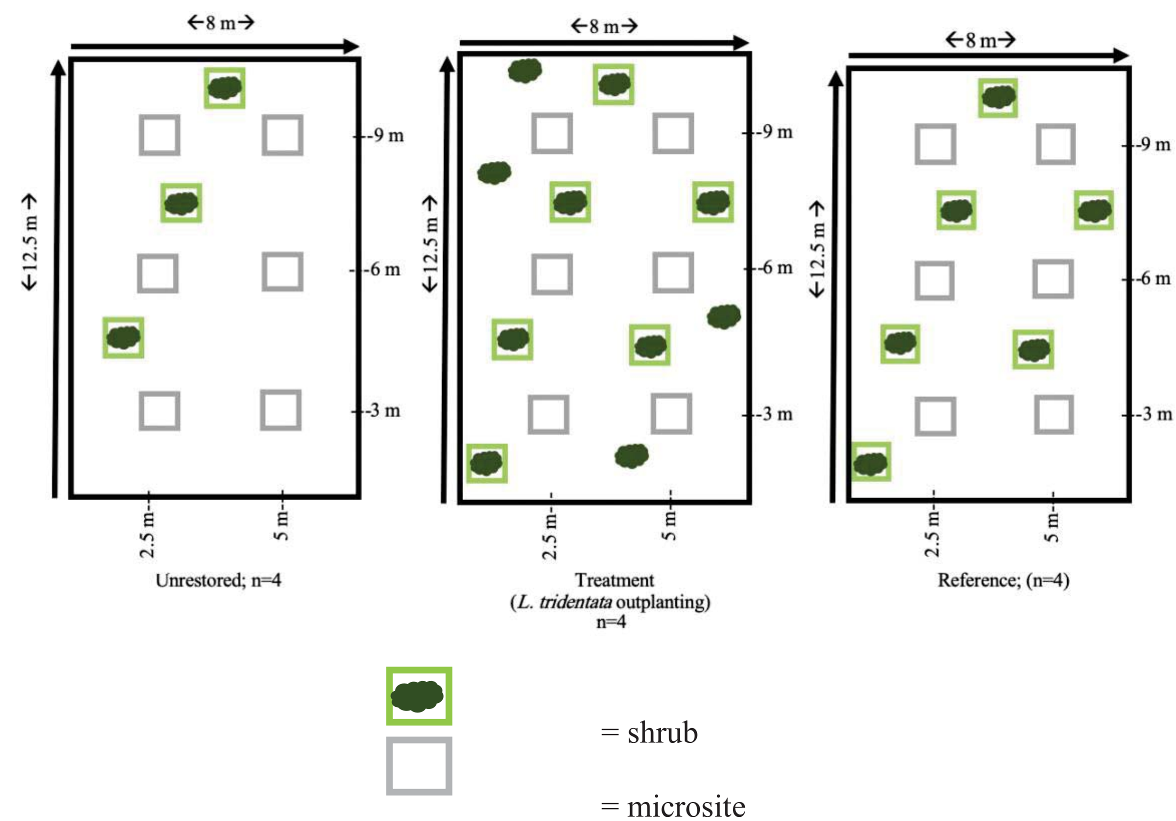


Figure A2. Sampling schematic and experimental design for assessing disturbance and restoration (outplanting) at the Fiber Optic Cable in the Mojave National Preserve, California, USA.

The black rectangle represents an entire plot (100 m^2; n=4). Grey squares represent 1 m × 1 m interspace subplots located at a standard location to avoid perennial cover. Green squares represent 1 m × 1 m shrub subplots. Only *L. tridentata* was used for shrub subplots. Because only *L. tridentata* was used for characterization of a shrub subplot, plots in unrestored area have 0-3 shrub subplots. Up to six individuals of *L. tridentata* were measured within restored and undisturbed plots.

Data Analysis

Planting treatment effects on native and exotic cover, species richness, and perennial plant density at a plot level were analyzed using one-way analysis of variance, generalized mixed models (PROC GLIMMIX, SAS version 9.4 2013), with plot type (disturbed/unrestored, disturbed/restored, and unrestored/reference) set as a fixed effect and plot as a random effect. If necessary, variables were either transformed to improve normality (cover, $\log_{10}+1$ or arcsine-square root transformed; species richness, square-root transformed), or, if normality was not improved by transformations, distributions were assessed and applied to models (lognormal distribution for cover; Poisson distribution for species richness) and goodness-of-fit tests were examined. Post-hoc tests with Tukey-Kramer adjustments for multiple comparisons were used to further assess significant effects or interactions. Tests were considered significant at $\alpha \leq 0.05$ and moderately significant at $\alpha \leq 0.10$.

Climate

Four weather stations were located proximally to the mine: 1) Horse Thief Springs California, CA US (elevation 1524 m, 69.2 km away); 2) Mitchell Caverns, CA US (elevation 1325.9, 31.6 km away); 3) Baker, CA US (elevation 293.2 m, 29.0 km away); 4) Mojave River Sink California (elevation 289.6, 29.0 km away). These stations, despite being further away and higher in elevation than desired, were still the best options; this particular region is lacking in weather stations compared to other regions in this chapter. Climate data from this site, specifically, should be evaluated with caution because of the large elevation and distance differences.

The Fiber Optic Cable was restored between 2000 and 2001 and weather data for this period was gathered from NOAA (Figure A4). Three years prior to the start of restoration through to the end of restoration, the average precipitation was about 12.1 cm year-1. The average high temperature was approximately 25.9° C and the average low was 12.0° C. From the end of restoration in 2001 to the first and only monitoring in April 2019, the average high temperature was 25.4° C and the low was 11.9° C. There was not enough complete data from the stations to determine precipitation. However, the average annual precipitation from the end of restoration to 2013 was 10.1 cm year-1.

The long-term precipitation annual average using 17 years was 10.6 cm. The long-term high and low average temperatures using 22 years up to the present was 24.7°C and 11.9°C. With the data that is currently available, it appears that climate remained relatively consistent since 1997.

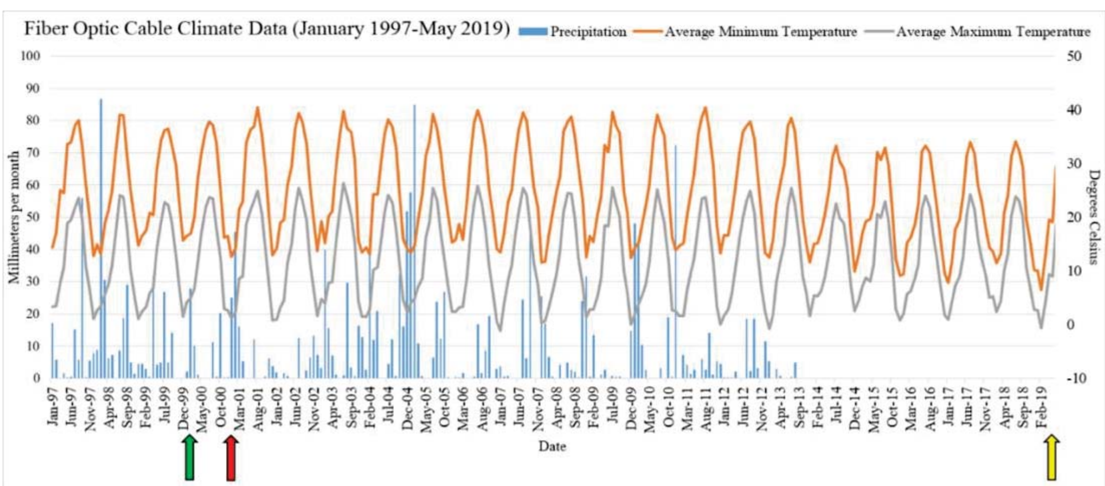


Figure A4. Fiber Optic Cable climate data from three years pre-restoration to current time. Primary y-axis is mean monthly precipitation per year (mm). Secondary y-axis is daily temperature in degrees Celsius. Green arrow denotes approximate commencement of restoration.

Results

Eighteen years later, the outplanting significantly affected perennial cover and density and exotic and native annual cover (Table A1). Perennial cover was highest in the reference and lowest in the unrestored area; the treatment area was at an intermediate cover level related to both of the other two. The reference plots had three and two times more cover than the unrestored and treatment plots, respectively. The treatment plots had 1.6 times more perennial cover than the unrestored plots. Upon parsing out which factors may be affecting perennial cover, I found that the only significant component at the plot level was woody perennial cover. Reference plots had the most coverage from woody perennials; plots were significantly different to the low cover found in the unrestored plots; treatment plots were related to both. Total perennial richness and all of its components were insignificant among plot types (Table A1).

Like cover, total perennial density significantly varied among plot types (Table A1) and increased from the unrestored, to the treatment, to the reference groups. However, unlike cover, the three plot types were distinctly significantly grouped apart. The outplanted plots had a mean density closer to reference conditions than unrestored conditions. Of the nineteen species that occurred within the area, three of the species had the highest density among all plots and two were dominant for the plant community. *Acamptopappus sphaerocephalus* had the highest density, interestingly, within the treatment area. *A. sphaerocephalus* numbers per treatment plots were well over double the numbers of both the reference and unrestored plots. The species has been noted to colonize a different disturbed pipeline at a higher density than the undisturbed land in previous studies; it is speculated to take a similar role as *Ambrosia dumosa* in sandier areas. (Webb et al. 1987). The two co-dominant species of the plant community were also significant among plot types (Table A1). *Ambrosia dumosa* had a significantly higher density within the reference area compared to both the treatment and unrestored areas. The unrestored and treatment areas did not significantly differ from one another. There were four times as many *A. dumosa* individuals in reference plots compared to treatment plots and three times more compared to unrestored plots. All three of the plot types were significantly distinct in their densities of *L. tridentata.* Treatment plots hosted the highest density, followed by reference plots, and the lowest density was found in unrestored plots. This represents decent survival among outplants within the treatment area. In the field, I noticed that there were no cages with a visibly dead *L. tridentata* or cages missing a plant entirely, suggesting that the outplants had indeed survived. If the cages of dead plants were to have been removed, it seems likely that the cages of the larger, healthy *L. tridentata* would also have been removed following the decreased herbivory threat.

Exotic cover was significantly highest in the reference area compared to the treatment and unrestored area (Table A1). The treatment area was an intermediary between the two other areas, but the actual mean was far more similar to the unrestored area. The reference plots had nearly four times the exotic cover compared to the treatment and unrestored plots.

Native annual cover and species richness were not significant among treatments or microsites (Table A1), possibly because plots were close to each other with sparse shrub cover within all of them. One concern for treatment plots is that the *L. tridentata* are still caged. While the plants seem to be somewhat successfully growing through the cages, the cages are likely a hindrance to the plant’s full growth potential.

Statistical Tests

Table A1. Effects of plot type on the cover, species richness, and density of native perennials and the effects of plot type on cover and species richness of native and exotic annuals.

Results are following restoration treatments for the Fiber Optic Cable, Mojave National Preserve, CA USA. Plot types include disturbed/unrestored, disturbed/treated and reference. Significant effects (α≤0.50) are in bold. Degrees of freedom for all models were 2, 9.

Fiber Optic Cable	Effect	Plot type	
	Variable	F Value	Pr > F
Exotic	Annual cover	6.05	0.022
	Annual richness	0.49	0.626
Native	Annual cover	0.46	0.644
	Annual richness	2.23	0.164
	Perennial cover	6.07	0.021
	Perennial richness	0.52	0.613
	Perennial forb cover	1.77	0.224
	Perennial forb richness	0.25	0.787
	Shrub cover	6.09	0.021
	Shrub richness	0.99	0.409
	Larrea tridentata density	77.40	<0.001
	Ambrosia dumosa density	9.94	0.005
	Acamptopappus sphaerocephalus density	7.27	0.013
	Total perennial density	21.17	<0.001

A1.2 Morningstar Mine

Background and Study Area

The Morningstar Mine is located in the northern section of the Mojave National Preserve in California, United States of America (Figure A5). The mine is in the southern Ivanpah Mountains (35°21’40, -115°29’26). The Morningstar Mine was an active mine for silver, gold, and other precious metals from 1907-1942. The mine closed between 1943 and 1963, until another private enterprise bought the land and resumed mining in 1964 until 1992, when mining operations officially ceased.

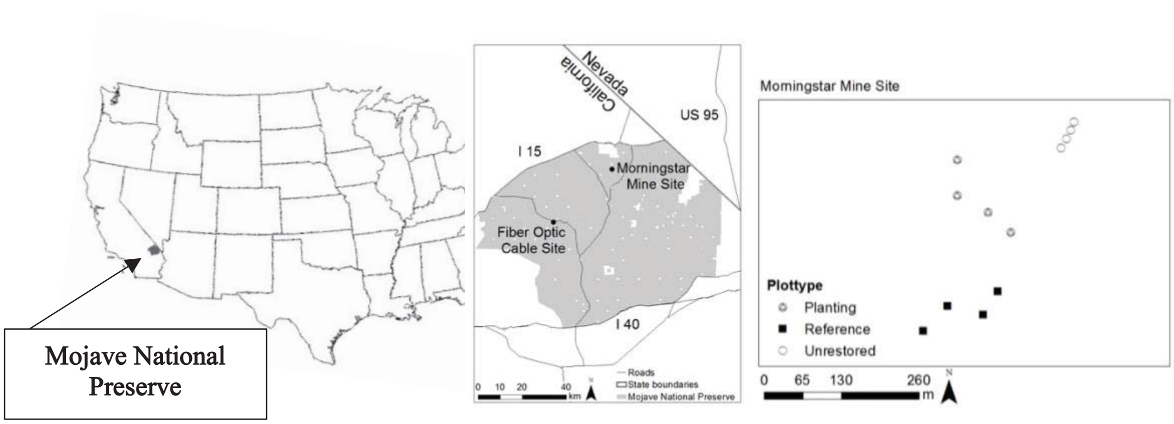


Figure A5. The Location of the Morningstar Mine.
A) The mine resides in the National Park Service-managed Mojave National Preserve, in the southwestern United States. B) The mine is located in the central portion of the Mojave National Preserve. C) Plots on the mine are located to the northwest of Morningstar Mine Rd. and is fenced and locked off to public use.

Sometime near the turn of the millennium, there was a noticeable cyanide leak from the leach pools that was permeating into the surrounding land. A temporary dam was constructed in 2002; there is no indication that any work has been completed on the dam since. Between 2008 and 2009, the 5-ha mine underwent extensive restoration effort. Old buildings and mining relict were removed. The land was then graded and recontoured to shallow the deep furrows and piles from mining operations. Work crews entered the surrounding undisturbed desert and dug up abundant (e.g. several large populations) species and cactus cuttings. These were used for transplantation on parts of the restoration sites. Transplants consisted of nearly 2,000 individuals of eleven different species that were planted on a 4-ha area. Following the transplantation, crews hydroseeded annual and perennial species in 2009. The amount of seed applied per acre varied between species, with a minimum of 0.5 lbs. and a maximum of 7 lbs. acre^{-1}. In total, 316.7 lbs. of seed was spread across the 4-ha area. In 2011, mechanical weed control efforts were conducted to reduce exotic cover in problematic areas.

While recontouring does represent an abiotic treatment, it occurred to all parts of the mine, and is thus eliminated for consideration as an individual treatment. The seeding and outplanting occurred to the same area and also represent a single treatment rather than two separate treatments. Therefore, the three plot types assessed are: disturbed (unrestored), treatment (seeding and transplanting), and the undisturbed (reference) plots. Ten years following restoration, the site was monitored for several metrics.

Monitoring

Plot sizes are 10 × 10 meters, with a minimum of a 1-meter buffer between plots in the smaller, unrestored area and up to several meters in the two larger areas. A random subset of four plots per plot type was chosen using a random number generator. Plots were measured between 10-11 April 2019. Plot elevations ranged from1374-1406 m. Slopes gradients ranged between 2-12° and slope aspects were between 70-130°.

For each plot, six subplots were measured at each the four plot corners and along the midpoint of the x-axis, as illustrated in Figure A6. Using a 1-m^2 quadrat, percent cover of all species present was obtained and given a cover class: (1=<0.01%, 2=0.01-0.1%, 3=1-2%, 4=2-

5%, 5=5-10%, 6=10-25%, 7=25-50%, 8=50-75%, 9=75-95%, 10=>95%, modified from Peet et al. 1998). Following measurements of all subplots, species that had not yet been encountered were recorded and given a cover class at the whole plot level. All perennial species within the plot were counted to determine the density of each perennial species.

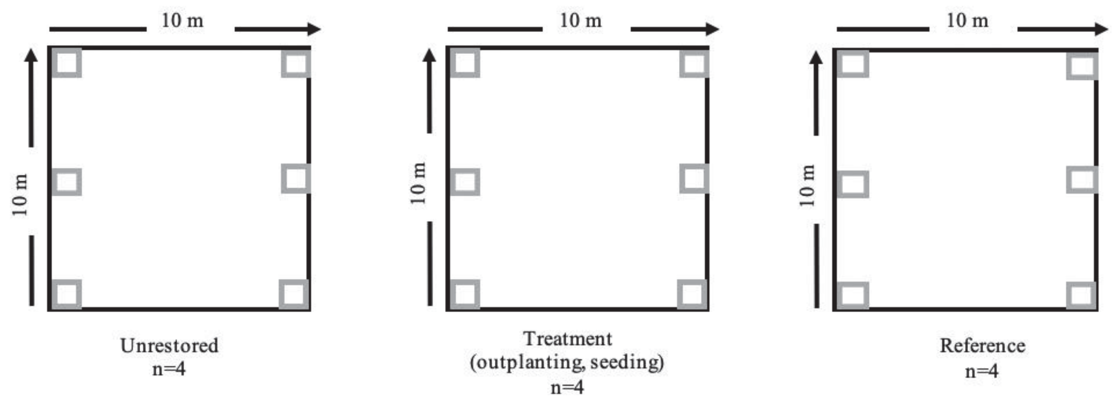


Figure A6. Sampling schematic and experimental design for assessing restoration treatments at the Morningstar Mine, Mojave National Preserve, California, USA.
The black squares represent the entire plot (100 m_2; n=4). Grey squares represent the six nested 1 m x 1 m subplots in standard locations at the four corners and at the midpoints of the two north-south-oriented opposing axes within the whole plot. Species not captured in subplots were recorded and assigned a cover class based on the entire plot area.

Data Analysis

Restoration treatment effects on native and exotic cover, species richness of life history groups, and perennial plant density were analyzed using generalized mixed models (PROC GLIMMIX, SAS version 9.4 2013), with plot type (disturbed/unrestored, disturbed/restored, and unrestored/reference) set as a fixed effect and plot as a random effect. If necessary, variables were either transformed to improve normality (cover, $\log_{10}$+1 or arcsine-square root transformed; species richness, square-root transformed), or, if normality was not improved by transformations, distributions were assessed and applied to models (lognormal distribution for cover; Poisson distribution for species richness) and goodness-of-fit tests were examined. Post-hoc tests with Tukey-Kramer adjustments for multiple comparisons were used to further assess significant effects or interactions. Tests were considered significant at $\alpha \leq 0.05$ and moderately significant at $\alpha \leq 0.10$.

Climate

Three weather stations were located somewhat proximally to the mine: 1) Mountain Pass 1 SE, CA US (elevation 1456 m, 11.9 km away); 2) Mountain Pass, CA US (elevation 1441.7, 13.2 km away); 3) Mid Hills California, CA US (elevation 1649.9 m, 26.6 km away). These stations, like the Fiber Optic Cable, were still the best possibilities despite elevation and geographic differences; this particular region was deficient in weather stations with complete data compared to other regions in this chapter. Despite using three different weather stations, precipitation data is missing from June to August 2006 and April 2013 to September 2019. This leaves the misfortune of not knowing precipitation patterns just prior to and during the only monitoring year in 2019.

Morningstar Mine was restored between October 2008 and February 2009. Three years before restoration began to its end, the average precipitation was about 21.5 cm year^{-1} (Figure A7). The average high temperature was approximately 20.0° C and the average low was 8.2° C. From the end of restoration in February 2009 to the first and only monitoring in April 2019, the average high temperature was 20.1° C and the low was 8.7° C. There was not enough complete data from the stations to determine precipitation. However, the average yearly precipitation from the end of restoration to 2013 was 20.6 cm.

On a longer-term climate trend, the temperature over a 14-year period had an average high of 20.0°C and an average low of 8.7°C. Precipitation, based off a seven year data set from

2005 to 2012, was 21.2 cm year^{-1}. There was considerable consistency in climate over the time period for the data available, but should be evaluated with caution.

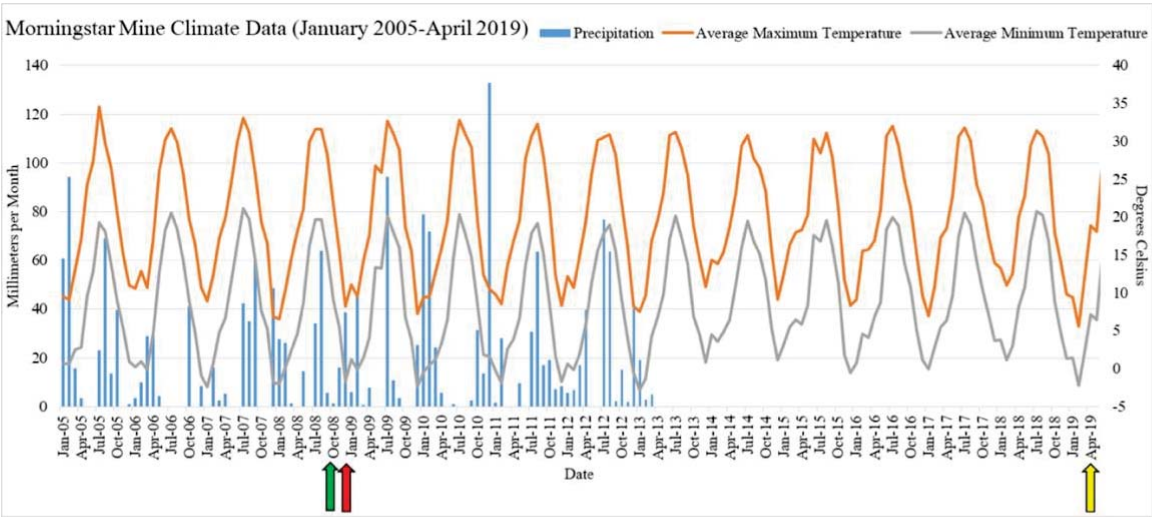


Figure A7. Morningstar Mine climate data from three years pre-restoration to current time. Primary y-axis is mean monthly precipitation per year (mm). Secondary y-axis is daily temperature in degrees Celsius. Green arrow denotes approximate commencement of restoration. Red arrow represents approximate finish of restoration. Yellow arrows indicate monitored years included in analysis.

Results

After 10 years, the transplanted perennials and the seeded annuals (one species) and perennials (twelve species) treatments significantly affected perennial richness, cover, and density, and exotic annual grass cover between the plot types (Figure A8).

Generally, the species richness of all perennials significantly increased from the unrestored, to the treatment, to the reference plots (Table A2). Treatment plots closely resembled reference plots.

Yucca richness moderately varied by plot type (Table A2). Specifically, the reference and treatment plots did not differ, and the treatment and unrestored area did not differ. The two species, *Y. brevifolia* and *Y. schidigera,* which comprised the total species richness of *Yucca* within the plot types but not the entire project site, were detected in the reference and treatment areas only. According to the 2011 National Park Service report, 173 *Y. brevifolia* and 91 *Y. schidigera* were transplanted to the restoration area. *Y. brevifolia* seeds were also included in the revegetation seed mix applied to the restoration area. In 2011, crews counted 1066 live *Y. brevifolia* individuals (including volunteers) and 79 live *Y. schidigera* within the five restoration area transects (61 m × 7 m) they delineated. While cactus richness alone and woody perennial richness alone were not significant by plot type, woody perennial and cactus species richness combined significantly increased, with treatment plots having more similarity to the most-diverse reference sites than to the least-diverse unrestored sites (Table A2). Woody perennial and cactus cover combined significantly increased from the unrestored, to treatment, to the reference plots. Woody perennial cover alone and cactus cover alone were significantly different between plot types, as well. The 2011 report stated that there were large populations of *Opuntia* species, among other cacti, in the surrounding area. Because of such high densities, the *Opuntia* species were used to revegetate the restoration area as cuttings. In 2011, the total number of surviving transplants was 77%, with barrel, hedgehog, and beavertail doubling in number. While I was unable to locate transplants due to the limited amount of data, my results appear to be consistent with the success of transplants: cactus density and cover was significantly higher in the restored and reference areas. While not significant, it should be noted that the reference area had eight times more cacti than the restoration area. There were no cacti in the unrestored plots, a mean density of less than two in the treatment plots, and a mean density of approximately eleven in the undisturbed plots. Perennial forb cover exhibited a moderate significance between plot types, with the treatment plots having the most percent cover. Perennial forb cover in the restored area was more than double the reference area and more than three times that of the unrestored area.

According to the 2011 official monitoring report, four different species of perennial forbs were seeded. All of the seeded species were relocated during 2019 surveys in both the reference and the restoration areas.

Perennial grasses did not have any statistical significance among plot types concerning cover and richness, however this life history group added only a very small contribution to overall perennial cover.

Perennial density of the treatment area was significantly more similar to the reference plots than the unrestored plots, which exhibited the lowest perennial density. Perennial density was significantly lower in the unrestored area and differed from both the restored and the reference. Unrestored plots had a mean density of approximately 15 individuals plot^{-1}, while restored plots had nearly seven times that amount and reference plots had nearly ten times that amount.

Cactus density in the treatment area was similar to the reference density. The unrestored area had no cactus present. In fact, the reference had more than eight times the density of the treatment area. *Yucca* density followed a similar trend as the cactus; however, the treatment areas were more similar to the unrestored areas where no *Yucca* was found. The density of perennial forbs was moderately significantly greater in the treatment plots compared to reference and unrestored plots. The unrestored and reference area had means of approximately 3.8 and 3.5, respectively, while the restored area had a mean well beyond those at 39.8. Perennial grass density was again not significant because it did not constitute a very large portion of total perennial density. The only perennial grass detected within all plots, *Achnatherum speciosum*, was the only graminoid in the revegetation seed mix, and was also only detected in the treatment area. Woody perennial densities increased from the unrestored, to treatment, to reference plots.

According to the 2011 annual monitoring report, three shrub species were transplanted in 2009 from the reference to the restoration area; eleven were seeded. Overall, density, species richness, and cover measurements were highest or second highest in the reference area, typically followed or preceded by the treatment area, and in every case was lowest in the unrestored area. This is unsurprising because during monitoring, it was clear that the unrestored area was just short of completely denuded of long-lived or even intermediate vegetation.

Native annual cover and species richness were not significant by plot type. Exotic cover significantly increased from the unrestored, to treatment, to reference plots. Treatment means were intermediate between the two other plot types. The driving factor was primarily an increase in exotic annual grasses, which exhibited the same trend as exotic annuals overall. There was a higher cover of both perennial and total plants and higher weed cover in the reference area.

Figure A8. Morningstar mine plots ten years after restoration (2019).

The plot type on the left is unrestored after extensive mining operations for nearly sixty years. The severely compacted soil is mostly denuded of vegetation. The middle photo shows the restoration of recontouring, seeding, and transplanting. The right picture is undisturbed desert conditions.

Statistical Significance Tests

Table A2. Effects of plot type on plant life history groups following restoration (seedling and planting) from Morningstar Mine, Mojave National Preserve, USA.

Plot types include disturbed/unrestored, disturbed/treated, and undisturbed/reference. Shrubs include all plants which exhibit a shrub growth habit, cactus, and yucca. Degrees of freedom for all models were 2, 9.

Morningstar Mine	Effect Variable	Plot type F Value	 Pr > F
Exotic	Annual cover	5.39	0.029
	Annual richness	0.05	0.956
	Annual forb cover	0.70	0.520
	Annual forb richness	1.29	0.323
	Annual grass cover	6.31	0.019
	Annual grass richness	0.21	0.816
Native	Annual cover	0.00	0.999
	Annual richness	0.53	0.605
	Perennial cover	7.95	0.010
	Perennial richness	7.88	0.011
	Perennial grass cover	0.68	0.533
	Perennial grass richness	0.79	0.481
	Perennial forb cover	3.60	0.071
	Perennial forb richness	2.23	0.163
	Tall stature perennial cover	20.43	<0.001
	Tall stature perennial richness	11.63	0.003
	Woody perennial cover	20.43	0.001
	Woody perennial richness	7.28	0.013
	Cactus cover	486.1	<0.001
	Cactus richness	0.64	0.549
	Yucca density	5.79	0.024
	Cactaceae density	11.64	0.003
	Total perennial density	8.19	0.009

A1.3 Northshore Road

Project Site and Background

The experimental study was conducted within Lake Mead National Recreation area along North shore road at a latitude and longitude of approximately 36 degrees 18’42.870” N and 114 degrees 29’18.819” W (Figure A9). The soil consists of Gypsid and Calcid soils, types of Aridisols (Lato 2006). The plant community is typical for the region. It contains spare shrub cover, many gypsophilic plants, perennial grasses, and a healthy layer of biocrust in most undisturbed sections (Chiquoine 2016).

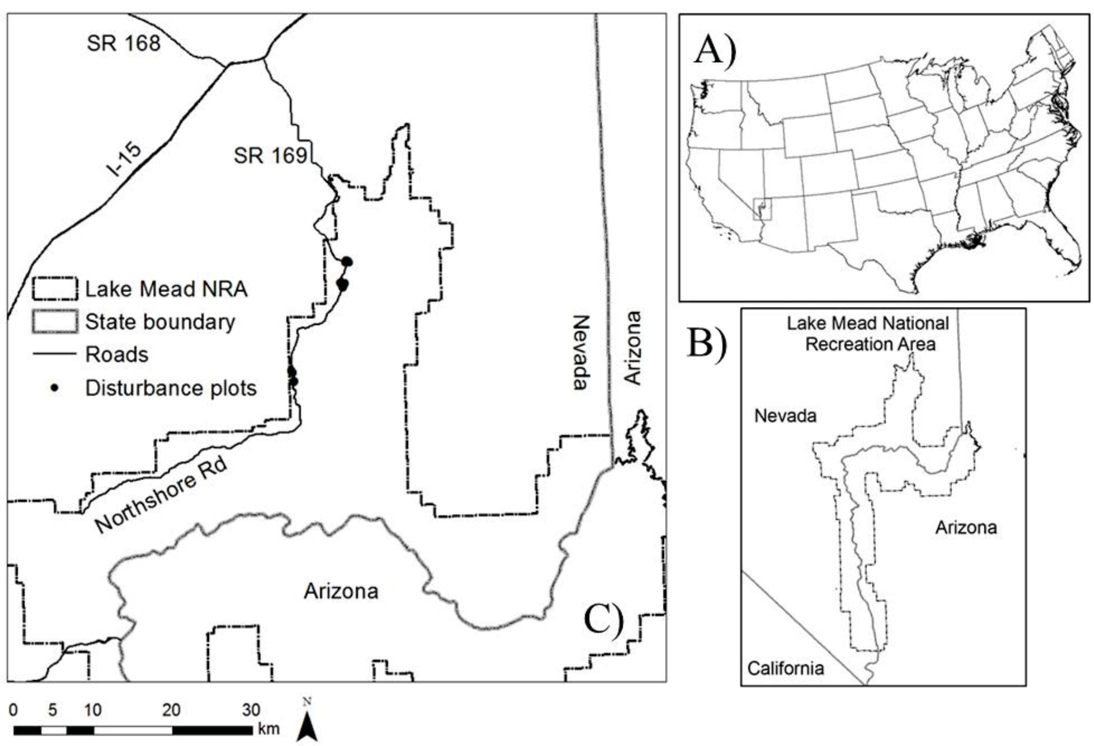


Figure A9. The Location of Northshore Road sites.
A) Lake Mead National Recreation Area is located in the southwest United States. B) Lake Mead National Recreation Area follows the Colorado River system and is made up of land from both Nevada and Arizona. C) Northshore Road is located in the eastern part of Lake Mead National Recreation Area in Nevada.

In order to improve driving conditions within the park, the old Northshore Road was realigned and widened with heavy machinery in 2008. During construction activities, topsoil was salvaged and stored in meter-high piles. Perennials in the path of destruction were dug up and stored in a temporary nursery for about a year. Areas previously covered in old pavement were recontoured and applied with a 5-cm thick layer of salvaged topsoil in 2009. The salvaged plants were planted in January 2010. The plants were then irrigated by three different methods. Treatments were implemented as a mixed effect design for the presence or absence of topsoil and perennial planting per each monitoring year. Each treatment plot was paired with a corresponding undisturbed pair, representative of the reference condition. Treatment levels included: 1) disturbed, topsoil, no planting, and undisturbed pair plot; 2) disturbed, no topsoil, no planting, and undisturbed pair plot; 3) disturbed, no topsoil, planting, and paired plot; 4) and disturbed, topsoil, planting, and paired plot. Overall, there were forty plots. Plots were usually parallel to the road and measured 50 m × 2 m, equaling 100 m_2 per plot. As part of a nested design to estimate whole plot cover, six 1 m × 1 m subplots were placed along the longitudinal axis at 0.5, 10.5, 20.5, 30.5, 40.5, and 49.5 meters (Figure A10).

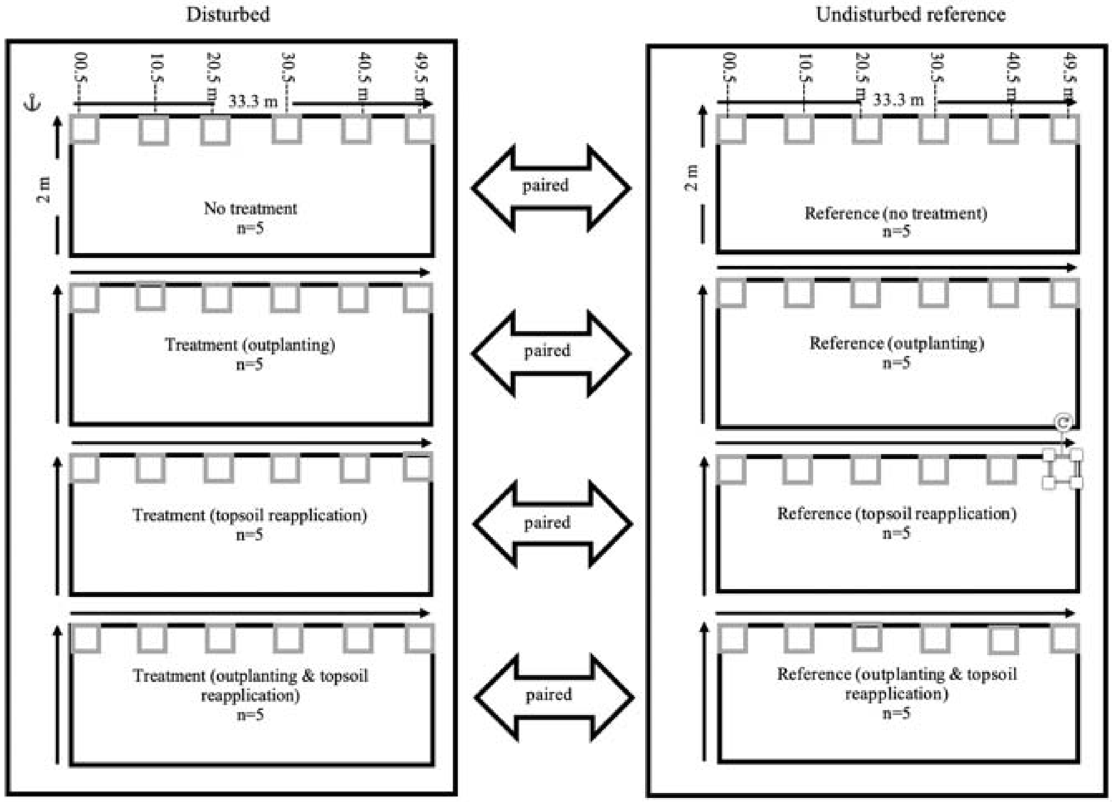


Figure A10. Sampling schematic and experimental design for assessing disturbance and restoration treatments applied along Northshore Road in Lake Mead National Recreation Area.

Twenty 2 m × 50 m plots were located within the disturbed areas along the 19 km stretch of Northshore Rd, which underwent reconstruction between 2008-2010. Disturbed areas varied in size and plot dimensions were selected to fit within disturbances and treatments. Plots were established in 2016 within areas that received the following treatments between December 2009 and January 2010 (n=4): no treatment (control), outplanting, topsoil reapplication, or topsoil reapplication and outplanting combination. The back rectangles represent different plot types and their respective plot pairs. Within 100 m of each disturbed plot, an undisturbed reference plot was also established in 2016 on the same or similar landform in which the disturbed plot was located to control for variability among soil types and site along the linear disturbances. Within all plots, six nested 1 m × 1 m subplots (illustrated as grey squares) were placed in standard locations centered at standardized in equal increments along one of the long axes. Plants that were not captured within subplots were recorded and assigned a cover class at the whole plot level.

Northshore Road was assessed for perennial and annual percent cover, species richness and for perennial density in 2016, 2017, and 2019. A 1 m × 1 m quadrat was used to measure subplots in a nested design. Percent cover followed the cover classes: 1=<0.01%, 2=0.01-0.1%, 3=1-2%, 4=2-5%, 5=5-10%, 6=10-25%, 7=25-50%, 8=50-75%, 9=75-95%, 10=>95%. Species that did not appear in subplots were recorded and given a cover class based off their cover for the entire plot.

Data Analysis

Year and treatment effects on native and exotic cover and species richness of life history groups and perennial plant density were analyzed using generalized mixed models (PROC GLIMMIX, SAS version 9.4 2013), with year (2016, 2017, and 2019) and treatments (outplanting, yes or no; topsoil reapplication, yes or no) set as fixed effects and with paired disturbed and undisturbed plots blocked as a random effect to ensure pairing. All main effects, two-way, and three-way interactions between year and treatments were analyzed. If necessary, variables were either transformed to improve normality (cover, $\log_{10}+1$ or arcsine-square root transformed; species richness, square-root transformed), or, if normality was not improved by transformations, distributions were assessed and applied to models (lognormal distribution for cover; Poisson distribution for species richness) and goodness-of-fit tests were examined. Post-hoc tests with Tukey-Kramer adjustments for multiple comparisons were used to further assess significant effects or interactions ($\alpha \leq 0.05$). Non-significance of disturbance indicates that variability among sites for that variable is greater than the variability between disturbed and undisturbed reference plots.

Climate

Three weather stations were located proximally to the road: 1) Valley of Fire State Park, NV US (elevation 609.6 m, 8.5 km away); 2) Echo Bay, NV US (elevation 381.0, 6.1 km away); and 3) Overton, NV US (elevation 381.0 m, 9.2 km away).

Northshore Road was restored between December 2009 and January 2010, for the topsoil and outplanting. Three years prior to the start of restoration through to the end of restoration, the average precipitation was about 12.9 cm year^{-1}. The average high temperature was approximately 28.5° C and the average low was 13.3° C. The average of the two restoration years, 2009 and 2010 was very close to the 13-year average, but restoration years had over a centimeter more rainfall. When divided between the year that topsoil was applied, 2009, and the year the outplanting was completed, 2010, the earlier year was drastically worse with about half the precipitation as the long-term average. The year 2010 had over 10 cm more rain fall than the long-term average. Thus, the year of the outplanting was a favorable precipitation year. From the end of restoration in 2010 to the first monitoring year in March 2016, the average annual precipitation that fell was 15.11 cm. The average high temperature was 28.3° C and the low was 13.8°C. During the first year of monitoring, precipitation levels were 14.5 cm year^{-1}, the average high temperature was 28.1°C and the average low was 18.6°C. During the second year of monitoring, precipitation levels were 14.4 cm year^{-1}, the average high temperature was 29.0°C and the average low was 8.5°C, which is substantially colder than any other year or the long-term average. The final year of monitoring, from January 2019 to September 2019 had an average annual precipitation of 14.2 cm, the average high temperature was 29.0°C and the average low was 16.6°C. Except for outliers in 2009, 2010, and 2017, all years remained consistently close to long-term averages.

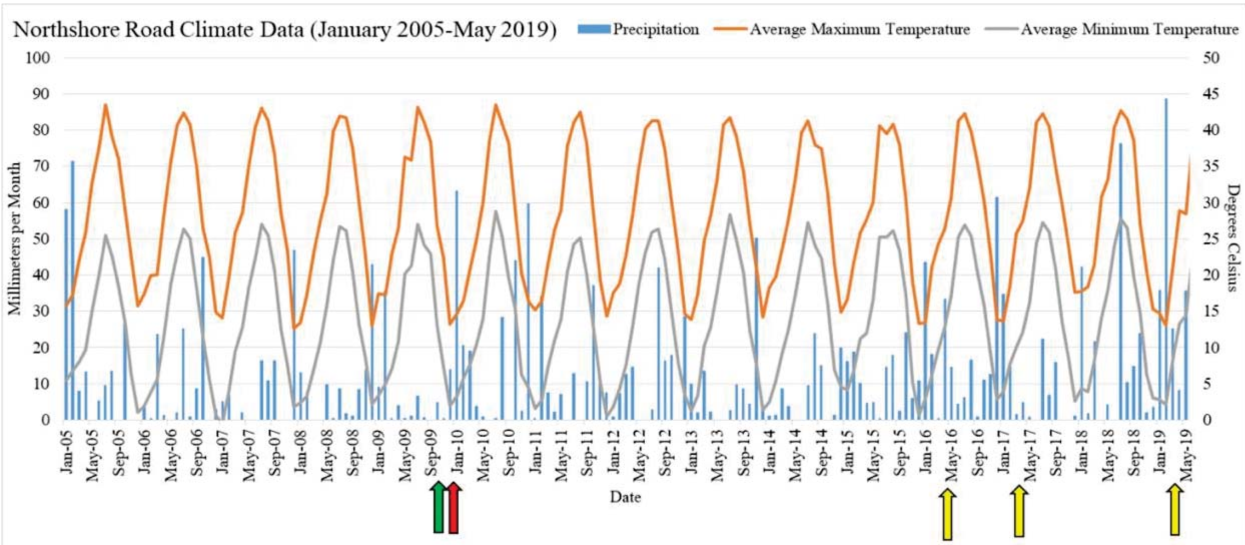


Figure A11. Northshore Road climate data from three years pre-restoration to current time.
Primary y-axis is mean monthly precipitation per year (mm). Secondary y-axis is daily temperature in degrees Celsius. Green arrow denotes approximate commencement of restoration.

Results

Northshore plots reacted significantly with perennial cover, density, species richness, and some annual native and exotic measurements (Table A3). Planting did not appear to have a big impact on restoration, however topsoil replacement did.

Table A3. Significant effects on native and exotic annual and perennial plant life history groups.

Main effects include year surveyed (2016, 2017, and 2019), topsoil reapplication (yes, no), and outplanting native perennial forbs and shrubs (yes, no).

Variable	Effect	Num DF	Den DF	F Value	Pr > F
Native annual cover	Year	2	55	23.28	<.0001
Native annual species richness	Year	2	64	25.95	<.0001
Perennial cover	Year × disturbance × topsoil reapplication	2	64	3.87	0.026
Perennial species richness	Year × topsoil reapplication	2	64	3.72	0.030
Native perennial forb cover	Year × topsoil reapplication × outplanting	2	44	3.55	0.037
Native perennial forb richness	Year	2	64	5.23	0.008
Native perennial grass cover	Year	2	64	3.16	0.049
Shrub cover	Year × disturbance × topsoil reapplication	2	64	3.94	0.024
Shrub richness	Disturbance	1	64	3.88	0.053
Shrub richness	Year	2	64	5.53	0.006
Exotic annual cover	No significant effects	-	-	-	-
Exotic annual species richness	Year × disturbance	2	64	5.84	0.005
Exotic annual forb species richness	Disturbance	1	64	7.12	0.010
Exotic annual forb species richness	Year	2	64	6.07	0.004
Exotic annual grass species richness	Year × disturbance × topsoil reapplication	2	64	3.18	0.048
Perennial forb density	Year	2	64	3.56	0.0343
Shrub density	Year x disturbance x plant	2	64	2.65	0.0783

The cover of perennial species exhibited a three-way interaction between year, disturbance, and soil (Table A3). Despite the direct effects of outplanting increasing perennial cover and density, no plant effects were detected. There was less cover in disturbed plots regardless of treatment compared to their undisturbed pair every year; highest cover values were always in undisturbed plots. The next highest plots are those that received topsoil coverage, indicating that topsoil application positively affected perennial cover. Cover increased in all categories between 2016 and 2017. Between 2017 and 2019, all categories except the topsoil undisturbed pairs decreased. There are no 2018 measurements, but perennials were likely negatively impacted by 2018 drought conditions, apparent through either reduced size or die-off. When year is removed from the model, undisturbed plots have more cover than treatment plots followed with moderate statistical significance of topsoil-applied plots having more cover than plots without topsoil. All perennials found at the site were considered native and consisted of mostly perennial forbs and shrubs with some species of perennial grasses scattered sparingly about. *Dasyochloa pulchella, Hilaria rigida, Sporobolus cryptandrus, Achnatherum hymenoides* were all perennial grasses found within plots, but the cover was not significant; the highest value was under 0.4%. For perennial forb cover, there was a significant year × topsoil reapplication × outplanting interaction (Table A3). It should be noted, however that disturbance was not significant. Thus, there was more variability between plot type than there was due to the disturbance. Highest values were in 2017 plots with topsoil and outplanting, followed by 2019 plots with outplanting and both with and without topsoil application. The lowest values occurred during 2016 in plots that received no outplanting or topsoil application. This is one of the only cases in which the transplants seemed to have an impact on restoration success. This success, however, may be of little ecological consequence: the maximum percent cover among all plot types was only. Nine of the twelve plot types were below 1% cover. Shrub cover significantly increased in plots with topsoil, except in 2019 where the cover dropped 0.3%. In 2019, percent coverages typically did not vary between plot types, except by a large amount in the topsoil reference plots. In all years, reference plots were higher than their pairs that had topsoil applied. In all values except 2016 undisturbed sites and 2019 disturbed sites, topsoil increased the percent cover of shrubs. Most interactions among the years remained similar or completely nonsignificant, suggesting that the addition of topsoil may play an important role in shrub cover. Disturbed, no-topsoil plot types were extremely low compared to the other plot types in 2016 and 2017, but then increased to about equal amounts in 2019.

Perennial species richness had significant interactions between year and soil (Table A3). Year appeared to play a larger role than the addition of topsoil, although topsoil did increase species richness for each monitoring year except 2019. This is substantiated when year is excluded from the model and soil no longer has a significant p value. Topsoil allowed for a slight increase in the number of plants, according to the means. There is no statistical significance, however, between topsoil treatments within years (Table A3). There was no significant effect within the outplanting treatment. Perennial graminoid species richness was not significant. Perennial forbs were significantly lower in 2016 compared to 2017 and 2019, where it decreased by nearly half. While not significant, there was a slight decrease in diversity in 2019 compared to 2017. Shrub species richness varied with moderate significance among disturbance and significantly among year (Table A3). Between 2016 and 2019, there was a total increase of 77.5%. The majority of this increase, 65%, occurred between 2017 and 2019. Expectedly, although marginal, undisturbed plots had slightly more species diversity. In most categories, including undisturbed pairs regardless of topsoil application, 2016 appeared to have greater species richness than either 2016 or 2019. With year excluded as a repeated measure, reference plots had moderately significantly more species.

The outplanting treatment tended to significantly affect individual species' densities more often than the general tests. The density measurements taken were perennial grasses, perennial forbs, woody perennials, and certain ecologically important species. Total density was significantly higher in disturbed plots in 2019 than any other pear or plot type. Interestingly, densities were lower in undisturbed plots compared to disturbed plots each year. In 2019, there were a very large number of perennial seedlings, sometimes several hundred in a plot. These numbers are largely driving this significance. Many plants die within the first two years (Ackerman 1973), so while some may survive, this hardly counts as a restoration success.

While not significant, disturbed plots tended to have less than 1 ± 0.3 (mean ± SE) perennial grass individuals per plot, while undisturbed plots tended to have only slightly higher number of individuals per plot but with greater variability among plots (1.5 ± 5; mean ± SE). Additionally, in undisturbed plots, there was a progressive increase in the mean number of perennial grass individuals among plots between the first survey and the last survey. Perennial forb density was similar between 2016 and 2019, while also similar between 2017 and 2019. An increase between 2016 and 2017 is indicative of a good year, and the sharp decrease into 2019 may have been caused by drought die-off in 2018. Total shrub density was also affected by the sheer number of seedlings and exhibited the exact trends seen with total perennials, including similar means. The density of *Ambrosia dumosa* was significantly lower in 2016 compared to 2017 and 2019. Plots with topsoil application had 2.7 times the density compared to plots that were treated with just the subsurface soil. Interestingly, although only moderately significant, plots with planting treatments had less than half of the number of individuals compared to plots without planting.

Native cover, which includes both annuals and perennials, was significantly higher in plots that were spread with topsoil regardless of year, and also higher in undisturbed plots compared to their paired treatment and unrestored plots. Native cover significantly increased throughout the monitoring period (Table A3); the middle monitoring year, 2017, was similar to both the first and most recent monitoring years. Native species richness, in most categories and including undisturbed pairs regardless of topsoil application, was greater in 2017 than 2016 and 2019. Plots that received the topsoil treatment were more diverse than plots that did not in all three monitoring years. As time progressed, their species composition drastically diverged. Undisturbed pairs always surpassed both disturbed soil type plots in native diversity except in an insignificant reversal where in 2016, the disturbed topsoil plot exhibited better species composition than its paired reference plot. This is a small example of the treatment actually showing better results; although, the difference in species richness for the pair in 2017 and 2019 was one species. With year as a random repeated measure, the conclusions above did not vary. Disturbed plots with no topsoil had just a third or less of the species diversity of undisturbed and disturbed, topsoil-treated.

Annuals, regardless of nativity, are expected to vary considerably in both composition and cover with year because of precipitation patterns. This trend was observed in most tests, unless the results were insignificant. Native annual cover was significantly lower in 2016 compared to 2018 and the highest cover percentage in 2019. Native annual species richness was lowest in 2016, tripled in 2017, and slightly, but significantly, decreased by 2019. Interestingly, when year is a repeated measure variable only, disturbance and soil interactions became moderately significant in that disturbed plots with no topsoil application had half the diversity of other plot types.

All encountered exotic species were either annual grasses or annual forbs. Exotic cover did not vary significantly across year, soil, or disturbance, but there is a developing trend of increasing cover throughout the years. In 2006, plots were 0.31% ± 0.10 covered with exotic species. By 2017, this number almost quadrupled. In 2019, 2017 percentages nearly tripled for an average cover of 3.25 ± 0.80. Exotic species richness varied with year and soil treatment and also year and disturbance. Plots with topsoil were higher than plots without in every year except 2019, but the unrestored plot had a value that was less than a tenth of a percent higher. Disturbed plots showed a higher richness than undisturbed plots, and exotic richness increased in 2017 and declined slightly in 2019.

Table A4. Effects table for Northshore Rd.

Effects included year of survey [YR] (2016, 2017, and 2019), disturbance [DB] (yes, no), topsoil reapplication [TS] (yes, no), and outplanting [OP] (yes, no). Each treatment plot (topsoil reapplication × outplanting) had its own reference (undisturbed) paired plot. Topsoil reapplication and outplanting treatments were applied to disturbed plots only. Interpretation of significance of treatments is dependent on if disturbance is a significant main effect or a part of a significant interaction with treatments. If disturbance is not significant but topsoil reapplication or outplanting is significant, significance likely indicates a high variability of that variable among plots.

Northshore Road												
Effect		Variable	Perennial									
			Native									
			total	forb			grass			shrub		
			density	cover	richness	density	cover	richness	density	cover	richness	density
YR		df	2, 64	2, 44	2, 64	2, 64	1, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	6.06	0.37	5.23	3.56	0.15	2.19	0.05	0.7	5.53	7.03
		Pr > F	0.0039	0.691	0.008	0.0343	0.697	0.120	0.9503	0.500	0.006	0.0017
PLOT	OP	df	1, 64	1, 44	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64
		F Value	1.6	0.3	0.24	1.08	0.07	0.04	0.32	1.6	0.16	2.16
		Pr > F	0.2111	0.586	0.628	0.3022	0.791	0.838	0.5736	0.211	0.689	0.1461
	TS	df	1, 64	1, 44	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64
		F Value	1.29	0.02	0.08	0.06	0.37	0.17	0.32	3.59	0.89	1.6
		Pr > F	0.2607	0.886	0.785	0.8131	0.547	0.682	0.5736	0.063	0.349	0.211
	DB	df	1, 64	1, 44	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64
		F Value	6.01	1.02	0.07	0.95	0.15	0.12	0.01	9.32	3.88	4.99
		Pr > F	0.0169	0.317	0.795	0.3338	0.697	0.733	0.9359	0.003	0.053	0.029
YR × PLOT	YR × OP	df	2, 64	2, 44	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	3.13	0.39	0.1	2.01	0.03	0.08	0.05	0.02	0.5	2.51
		Pr > F	0.0505	0.681	0.902	0.1421	0.975	0.924	0.9503	0.985	0.611	0.0896
	YR × TS	df	2, 64	2, 44	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	0.11	0.58	1.34	0.39	0.07	1.52	0.05	0.1	2.16	0.13
		Pr > F	0.899	0.566	0.270	0.6806	0.932	0.227	0.9503	0.906	0.124	0.8795
	YR × DB	df	2, 64	2, 44	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	5.08	0.04	1.85	1.07	0.27	0.65	0.18	0.46	1.84	6.11
		Pr > F	0.009	0.957	0.165	0.3485	0.766	0.523	0.8386	0.635	0.168	0.0037
YR × PLOT × PLOT	YR × OP × TS	df	2, 64	2, 44	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	0.11	3.55	0.86	1.72	0.36	0.08	0.05	0.46	0.76	0.02
		Pr > F	0.9004	0.037	0.426	0.1877	0.697	0.924	0.9503	0.635	0.471	0.9802
	YR × OP × DB	df	2, 64	2, 44	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	2.75	0.03	0.79	1.09	0.07	1.24	0.18	0.82	1.17	2.65
		Pr > F	0.0717	0.974	0.459	0.343	0.931	0.295	0.8386	0.447	0.318	0.0783
	YR × TS × DB	df	2, 64	2, 44	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	0.13	0.2	0.74	0.7	0.1	0.65	0.18	3.94	0.55	0.18
		Pr > F	0.8819	0.822	0.479	0.5023	0.901	0.523	0.8386	0.024	0.582	0.8395
YR × PLOT × PLOT x PLOT	YR × OP × TS × DB	df	2, 64	2, 44	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64	2, 64
		F Value	0.1	0.78	0.55	2.06	0.36	1.47	0.18	1.77	0.34	0.01
		Pr > F	0.908	0.464	0.579	0.1363	0.696	0.238	0.8386	0.179	0.714	0.9858
PLOT × PLOT	OP × TS	df	1, 64	1, 44	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64
		F Value	0	0.24	0.01	0.16	0.48	1.06	0.32	1.07	0	0
		Pr > F	0.9821	0.629	0.939	0.6869	0.493	0.308	0.5736	0.306	0.968	0.9807
	OP × DB	df	1, 64	1, 44	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64
		F Value	0.6	0.44	0.2	0.24	0.09	0.47	0.01	1.9	2.15	0.85
		Pr > F	0.4424	0.513	0.653	0.6243	0.770	0.496	0.9359	0.173	0.147	0.3598
	TS × DB	df	1, 64	1, 44	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64
		F Value	0.66	2.56	1.93	0.08	0	0.79	0.01	0.12	0.1	0.18
		Pr > F	0.4213	0.117	0.170	0.7783	0.975	0.377	0.9359	0.734	0.749	0.6764
PLOT × PLOT × PLOT	DB × OP × TS	df	1, 64	1, 44	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64	1, 64
		F Value	0	0.16	0.26	0.44	0	0.08	0.01	0	0.15	0.03
		Pr > F	0.9973	0.690	0.615	0.5118	0.971	0.785	0.9359	0.949	0.704	0.8669

Northshore Road						
Effect		Variable	Annual			
			Native total		Exotic total	
			cover	richness	cover	richness
YR		df	2, 55	2, 64	2, 64	19.66
		F Value	23.38	25.95	1.78	<0.001
		Pr > F	<0.001	<0.001	0.176	2, 64
PLOT	OP	df	1, 55	1, 64	1, 64	0.13
		F Value	0.55	0.05	0.5	0.718
		Pr > F	0.460	0.820	0.483	1, 64
	TS	df	1, 55	1, 64	1, 64	1.03
		F Value	1.13	2.58	1.42	0.314
		Pr > F	0.292	0.113	0.238	1, 64
	DB	df	1, 55	1, 64	1, 64	0.72
		F Value	0.04	1.57	0	0.401
		Pr > F	0.835	0.215	0.955	1, 64
YR × PLOT	YR × OP	df	2, 55	2, 64	2, 64	0.16
		F Value	0.69	0.97	0.57	0.856
		Pr > F	0.507	0.384	0.566	2, 64
	YR × TS	df	2, 55	2, 64	2, 64	4.98
		F Value	0.76	1.09	1.15	0.010
		Pr > F	0.472	0.344	0.323	2, 64
	YR × DB	df	2, 55	2, 64	2, 64	7.11
		F Value	0.2	0.7	0.04	0.002
		Pr > F	0.823	0.499	0.960	2, 64
YR × PLOT × PLOT	YR × OP × TS	df	2, 55	2, 64	2, 64	2.32
		F Value	0.21	0.38	0.57	0.106
		Pr > F	0.811	0.688	0.568	2, 64
	YR × OP × DB	df	2, 55	2, 64	2, 64	1.04
		F Value	0.37	0.22	0.04	0.358
		Pr > F	0.693	0.807	0.962	2, 64
	YR × TS × DB	df	2, 55	2, 64	2, 64	0.74
		F Value	0.18	0.28	0.03	0.483
		Pr > F	0.832	0.759	0.974	2, 64
YR × PLOT × PLOT x	YR × OP × TS × DB	df	2, 55	2, 64	2, 64	1.04
		F Value	0.13	0.08	0.04	0.358
		Pr > F	0.877	0.921	0.962	2, 64
PLOT × PLOT	OP × TS	df	1, 55	1, 64	1, 64	0.05
		F Value	0.1	0.24	0.64	0.829
		Pr > F	0.755	0.628	0.426	1, 64
	OP × DB	df	1, 55	1, 64	1, 64	0.04
		F Value	1.59	0.6	0	0.847
		Pr > F	0.212	0.441	0.953	1, 64
	TS × DB	df	1, 55	1, 64	1, 64	1.29
		F Value	0.92	2.63	0.14	0.260
		Pr > F	0.342	0.110	0.714	1, 64
PLOT × PLOT × PLOT	DB × OP × TS	df	1, 55	1, 64	1, 64	0.46
		F Value	0.05	2.63	0.05	0.501
		Pr > F	0.822	0.110	0.818	1, 64

A1.4 Road 108

Study Area and background

Within Lake Mead National Recreation Area at a latitude and longitude of approximately 36°22'58.32"N 114°25'35.99"W, a road leading from the main road, Northshore, to the waterfront of Lake Mead was decommissioned in 2002 (Figure A12).

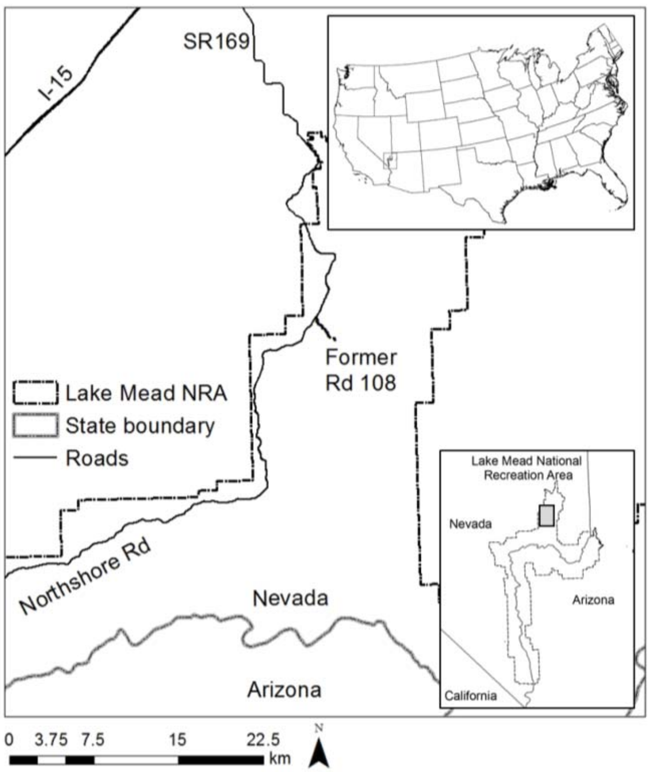


Figure A12. Location of Road 109 restoration site.
The closed road is located in Lake Mead National Recreation Area, which encompasses parts of Nevada and Arizona along the Colorado River system. Road 108 is in the northern part of the recreation area and is only accessible by foot from Blue Point Springs.

The old road was deeply disturbed and compacted from such regular use for access to Lake Mead. Later that year, National Park Service employees used heavy machinery to rip sections of the road in order to decompact the soil and leave textured scars. Ripped and unripped sections alternate every 0.1 mile along the closed Road 108. A walking trail still exists and, at some points, does intersect the project site. The soil is a Gypsid Aridisol (Lato 2006) with creosote-bursage and gypsophilic vegetation plant communities coexisting. Elevations range from 375-437 m. Slope gradients range from 1-18%, with an average of around 6%.

In 2009, 18 vegetation plots were established based off the ripping treatment locations (Figure A13). Six plots were established on the road in unripped portions (unrestored), six plots were established in the ripped portions of the road (treatment), and six plots were established proximal to the road but not close enough to have been affected by driving activities (undisturbed). Plots were developed in a nested design and measured 33.3 m × 3 m, or 99.9 m_2 in total area per plot. The nested design was observed using six 1 m × 1 m subplots. Plots were placed along the longitudinal axis and measured every 0.5, 6.0, 11.5, 17.0, 22.5, and 28.0 m along the longitudinal axis. Monitoring occurred between March and May, depending on the development of the annual plant community, in 2009, 2016, and 2019.

The subplots were monitored for perennial and annual cover and species richness and perennial density and total shrub volume. Visual estimates for percent cover followed cover classes: 1=<0.01%, 2=0.01-0.1%, 3=1-2%, 4=2-5%, 5=5-10%, 6=10-25%, 7=25-50%, 8=50-75%, 9=75-95%, 10=>95% (modified from Pet et al. 1988). Species that did not appear in subplots were recorded and given a cover class based off of their cover for the entire plot.

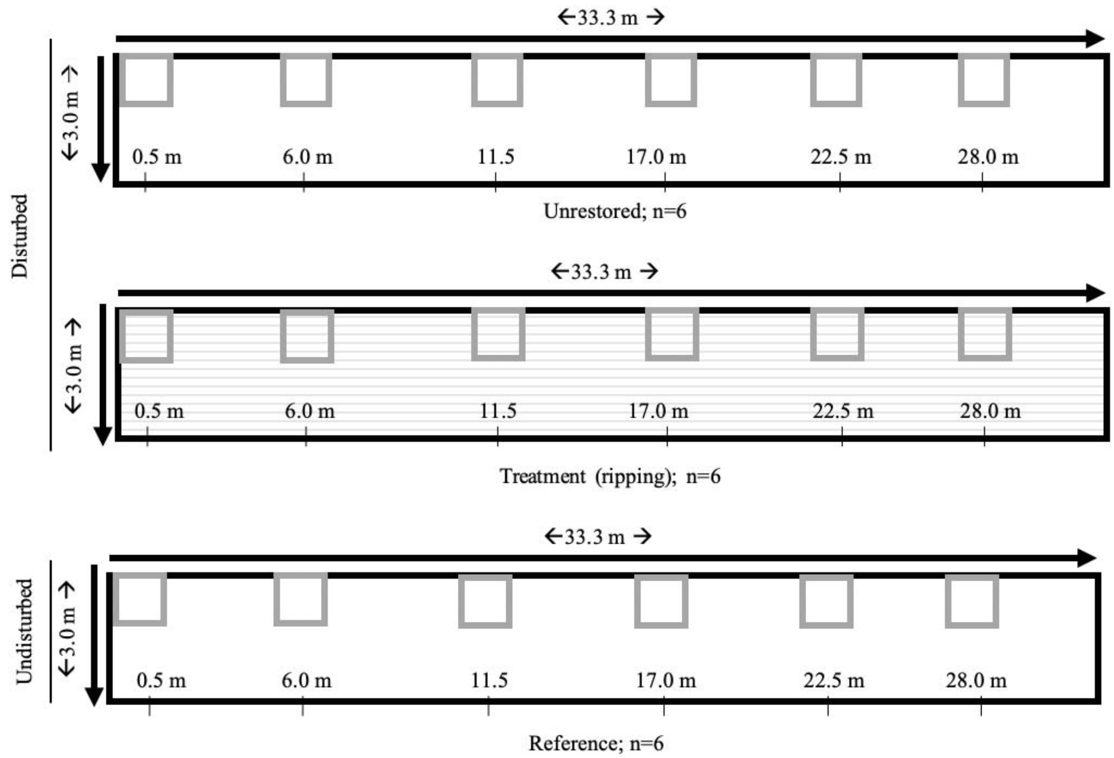


Figure A13. Sampling schematic and experimental design for assessing decommissioned Road 108 in Lake Mead National Recreation Area, Nevada, USA.

Road 108 was decommissioned in 2002. Approximate 100 m sections of the 3-km long closed road were ripped by heavy machinery with gaps between ripped sections remaining unrestored. Plot size was selected to fit within treatment sections and road width. Six triplet paired plots were established along the decommissioned road: unrestored (not ripped), ripped (treatment), and undisturbed reference plots. Ripped and unripped sections were paired with adjacent undisturbed reference sites to control for variability in vegetation along the linear road disturbance. Black rectangles represent whole plots (99.9 m_2; n=6 per plot type). Grey squares represent the six 1 m × 1 m nested subplots centered at 0.5 m, 6.0 m, 11.5 m, 17.0 m, 22.5 m, and 28.0 m along one of the long axes. Plants that were not captured within subplots were recorded and assigned a cover class at the whole plot level.

Data Analysis

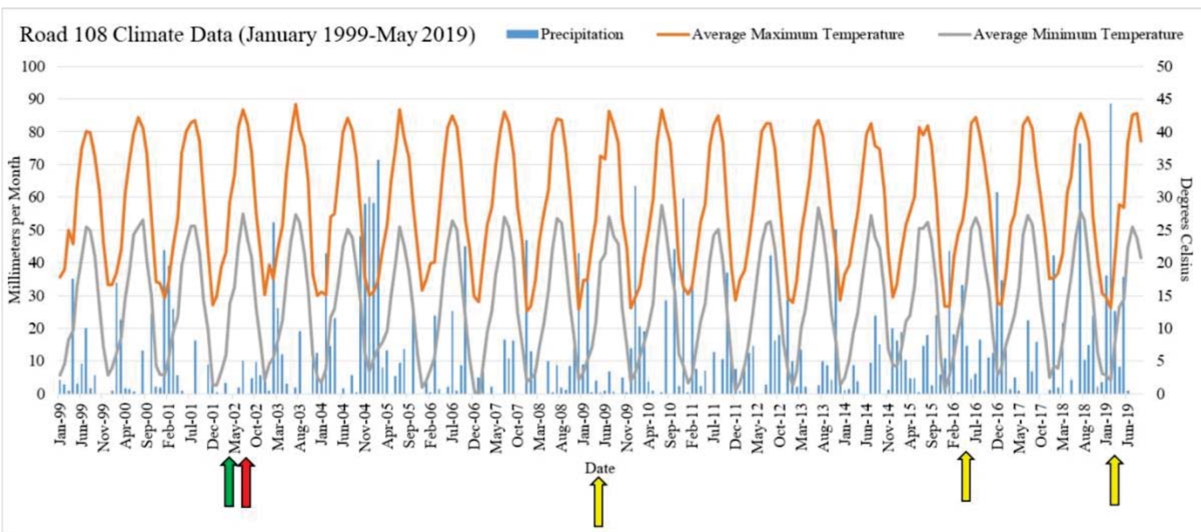
Year and ripped treatment effects on native and exotic cover and species richness of life history groups and perennial plant density were analyzed using generalized mixed models (PROC GLIMMIX, SAS version 9.4 2013), with year (2009, 2016, and 2019) and plot type (disturbed/unrestored, disturbed/ripped, and undisturbed/reference) set as fixed effects and triplet pairs blocked as a random effect. Paired unrestored and ripped treatment plots were blocked with respective undisturbed reference plot to ensure pairing. All main effects and two-way interactions between year and plot type were analyzed for significance ($\alpha \leq 0.05$). If necessary, variables were either transformed to improve normality (cover, $\log_{10}+1$ or arcsine-square root transformed; species richness, square-root transformed), or, if normality was not improved by transformations, distributions were assessed and applied to models (lognormal distribution for cover; Poisson distribution for species richness) and goodness-of-fit tests were examined. Post-hoc tests with Tukey-Kramer adjustments for multiple comparisons were used to further assess significant effects or interactions ($\alpha \leq 0.05$).

Climate

Three weather stations were located proximally to the decommissioned road: 1) Valley of Fire State Park, NV US (elevation 609.6 m, 0.7 km away); 2) Echo Bay, NV US (elevation 381.0, 8.1 km away); 3) Overton, NV US (elevation 381.0 m, 18.7 km away).

Road 108 was decommissioned in 2002 and restored in that same year. Three years prior to the start of restoration through to the end of restoration, the average precipitation was about 9.81 cm year^{-1} (Figure A14). The average high temperature was approximately 28.7° C and the average low was 13.7° C. From the end of restoration in 2002 to the first monitoring year in April 2009, average annual precipitation was 13.0 cm; the average high temperature was 28.6° C and the low was 13.3° C. The second year of monitoring in April 2016 had an average yearly precipitation of 22.29. The average high was 28.5 °C and the low was 14 .5° C. In the third monitoring year, 2019, average yearly precipitation was 19.42 cm, the average high was 29.9°C, and the average low was 14.8°C.

The past 20-year average from January 1999 to September 2019 had an average annual precipitation of 13.9 cm. The average high temperature was approximately 28.6° C and the average low was 13.7° C. The year of restoration showed that the precipitation that fell was only about 27.9% of the long-term average, however Road 108 had abiotic soil manipulation only,



and had no new plants or seeds depending on a favorable rain season.

Figure A14. Road 108 climate data from three years pre-restoration to current time.
Primary y-axis is mean monthly precipitation per year (mm). Secondary y-axis is daily temperature in degrees Celsius. Green arrow denotes approximate commencement of restoration.

Results

Closed Road 108 within Lake Mead National Recreation Area was decommissioned in 2002 and was restored later that year. The National Park Service used heavy machinery to create deep furrows in parts of Road 108, a technique called “ripping”. The main effects of the ripping treatment are in Table A4.

Table A5. Significant effects table for closed Road 108.

Variables show different significant effects ($p \leq 0.05$; moderate $p \leq 0.10$) on native and exotic annual and perennial plant life history groups. Main effects include year (2009, 2016, 2019) plot type (unrestored, treatment, reference) and year by plot type interactions.

Variable	Effect	Num DF	Den DF	F Value	Pr > F
Native annual cover	Year	2	21	22.18	<.0001
Native annual cover	Plot type	2	21	4.64	0.0214
Native annual species richness	YEAR	2	30	30.81	<.0001
Native annual species richness	Plot type	2	30	4.77	0.0159
Perennial species richness	Plot type	2	30	13.44	<.0001
Shrub cover	Year	2	23	3.41	0.0504
Shrub richness	Type	2	30	15.16	<.0001
Shrub density	Year x Type	4	30	4.75	0.0043
Exotic annual cover	Year	2	30	9.92	0.0005
Exotic annual species richness	Year	2	30	19.36	<.0001
Perennial density	Type	2	30	6.63	0.0041
Native perennial forb density	Type	2	30	3.01	0.0642

Perennial cover as a whole did not significantly differ among treatments or monitoring years (Table A5). Specifically, perennial forbs, which included the rare and endangered *Arctomecon californica* (Las Vegas Bearpoppy), were not significant among plot type or years, but did not constitute as large of percent cover as shrub species did. Shrub cover was marginally significantly greater in 2016 compared to 2009 and 2019. 2019 exhibited the lowest cover among the three monitoring years; however, shrub cover was very low in all plot types. Means ranged from 1.94%±0.81% in 2019 to 3.09%±0.86% in 2016. There were no perennial grasses detected at any time or plot throughout the study.

Perennial species richness was significantly greater in the reference area, which hosted a mean of 7±1.10 species, compared to the unrestored and treatment areas. The treatment area hosted the lowest number of species with a mean of 2.6 species, although not significantly less than the unrestored plots. Within specific lifeforms, perennial grasses for absent from the site, perennial forbs did not significantly differ, but shrub diversity was significantly affected by plot type. Shrub species richness was significantly greater in the reference compared to the unrestored and treatment areas. The mean number of shrub species per plot type increased from the unrestored area (mean=1.11 ± 0.25 species) to the treatment area (mean=2.39 ± 0.30 species), to the significant maximum in the reference area (mean=4.39 ± 0.27 species). While not significant, disturbed plots exhibited an opposite trend than the rest of the data, in that the treatment area is actually outperforming the unrestored area.

Perennial density was significantly less in the unrestored area compared to the treatment and reference area (Table A5). The treatment and reference, which were similar to each other, areas hosted between five and six times as many individuals as the unrestored plots. Perennial forb density provided another example of the treatment area performing well. Perennial forb density was moderately significantly largest in the treatment area, followed by the reference area, and smallest in the unrestored area. The density of shrubs varied with an interaction between treatment and year. Overall, shrub density increased over time except for a dramatic decrease in unrestored plots. Treatment plots tended to be similar to reference plots. To determine which shrub species were causing the statistical significance, I analyzed the densities of abundant shrub species. *Ambrosia dumosa, Lepidium fremontii, and Sphaeralcea ambigua* were not major contributors. *Psorothamnus fremontii* and *Stephanomeria pauciflora* both showed statistical significance regarding year and treatment interactions. There were significantly less *P. fremontii* individuals in 2009 compared to 2016 and 2019, although this increase is only by two or three individuals. Its density was twice as high in the reference and treatment plots compared to the unrestored plots. *S. pauciflora* in treatment plots increased over time, while reference plots decreased. Unrestored plots' *L. tridentata* densities were similar to both treatment and reference plots, although the total mean density of all plots combined does not equal up to two.

Native cover was moderately significantly highest in the reference plots followed by the unrestored plots and was significantly highest in 2009. While this is surprising for plot type, it is unsurprising for year because native cover includes both perennial and annual natives. As previously mentioned, native perennial cover was not significant. Native annual cover varied between main effects, year and plot type. Native annual cover was highest in the reference plots, followed by the unrestored plots. Treatment plots only had a mean native annual cover of 0.24% ± 0.06. The year 2019 had significantly more annuals than either 2009 or 2016. All of the native annuals were forbs except for one grass, *Vulpia octoflora*. Total exotic cover, which would include both grasses and forbs, was significantly greater in 2019 compared to 2009 and 2016. In fact, it was 19.5 times higher than in 2009, and 9.8 times higher than in 2016. This is interesting in that 2018 may possibly be considered a bad drought year. Therefore, with less rainfall one would expect less cover of all plants. It is possible that these exotic plants were able to outperform native forms in the adverse drought conditions. Invasive plants are typically very hardy; although this does not explain why cover would have been so low in the 2009 and 2016 monitoring years.

Native species richness was significantly greater in 2019 compared to 2009 and 2016, which is hardly surprising because annuals fluctuate drastically with year. 2016 had slightly more species than 2009, although the numbers are not significant. The minimum number of species, which occurred in 2009, was a mean value of 6.5 per plot and the maximum, occurring in 2019, reached 11.9 species per plot. Native annual species richness was significantly greater in the reference area compared to the treatment area; the unrestored area was similar to both. Exotic species richness was also significantly higher in 2019 compared to 2009 and 2016. Unlike cover, 2009 and 2016 are unrelated to one another. It follows the same trend as total exotic cover in that the variable increases over time. Exotic species richness was lowest within the treatment area, followed by the unrestored area and the reference area. The unrestored area was an intermediary between the treatment and reference areas. Perennial density and shrub density were highest in the reference area.

Table A6. Effects of year and plot type on the cover, species richness and density of native perennials and the effects of year and plot type on cover and species richness of native and exotic annuals.

Results are following restoration treatments (abiotic, ripping) for Road 108, Lake Mead National Recreation Area, NV USA. Plot types include disturbed/unrestored, disturbed/treated, and reference. Significant effects (α≤0.50) are in bold and moderately significant effects (α≤0.10) are italicized.

Road 108	Effect	Year			Plot type			Year × Plot type		
	Variable	df	F Value	Pr > F	df	F Value	Pr > F	df	F Value	Pr > F
Exotic	Cover	2, 30	9.92	**0.0005**	2, 30	2.16	0.133	4, 30	1.24	0.316
	Richness	2, 30	19.36	**<0.001**	2, 30	5.97	**0.007**	4, 30	2.55	*0.060*
Native	Cover	2, 29	2.53	*0.097*	2, 29	2.75	*0.081*	4, 29	0.94	0.455
	Richness	2, 30	24.99	**<.0001**	2, 30	9.13	**<0.001**	4, 30	1.12	0.365
	Annual cover	2, 21	22.18	**<0.001**	2, 21	4.64	**0.021**	4, 21	0.54	0.706
	Annual richness	2, 30	30.81	**<0.001**	2, 30	4.77	**0.016**	4, 30	1.06	0.393
	Perennial cover	2, 30	1.25	0.301	2, 30	0.54	0.589	4, 30	0.37	0.830
	Perennial richness	2, 30	0.3	0.742	2, 16	13.44	**<0.001**	4, 30	0.69	0.606
	Perennial forb cover	2, 16	0.95	0.406	2, 16	0.02	0.983	4, 16	0.05	0.996
	Perennial forb richness	2, 30	0.04	0.966	2, 30	1.67	0.206	4, 30	1.48	0.234
	Shrub cover	2, 23	3.41	*0.050*	2, 23	1.89	0.174	4, 23	1.03	0.411
	Shrub richness	2, 30	0.47	0.632	2, 30	15.16	**<0.001**	4, 30	15.16	**<0.001**
	Perennial density	2, 30	1.35	0.275	2, 30	6.63	**0.004**	4, 30	1.85	0.145
	Shrub density	2, 30	1.57	0.225	2, 30	9.27	**<0.001**	4, 30	4.75	**0.004**
	Perennial forb density	2, 30	0.27	0.765	2, 30	3.01	*0.064*	4, 30	0.65	0.634
	Larrea tridentata density	2, 30	2.12	0.138	2, 30	5.46	**0.010**	4, 30	1.74	0.167
	Psorothamnus fremontii density	2, 30	7.98	**0.002**	2, 30	3.83	**0.033**	4, 30	2.56	*0.059*
	Stephanomeria pauciflora density	2, 30	0.02	0.976	2, 30	1.5	0.240	4, 30	3.86	**0.012**

A1.5 Keys View Road

Study Area and Background

The Keys View Road experimental design was created to compare annual and perennial plant communities over time and determine whether restoring live or dead perennials facilitated successful recovery. Within Joshua Tree National Park at a latitude and longitude of approximately 33°57'21.43"N 116°10'23.75"W, construction activities to render the road safer by widening, repaving, resurfacing, and adding berms was completed early in 2008 (Figure A15). Later in 2008, parts of the adjacent roadside were recontoured and outplanted with 800 perennials; other parts of the roadside received a vertical mulch treatment; and other parts had both treatments. With plot establishment, this led to the designation of five plot types: disturbed and unrestored plots (unrestored), plots that received outplanting (OP), plots that received vertical mulch (VM), plots that received vertical mulch and outplanting combinations (OPVM), and plots undisturbed by construction activities (reference). The plots, which are situated alongside the road, range in elevation from 1292 m up to 1539 m.

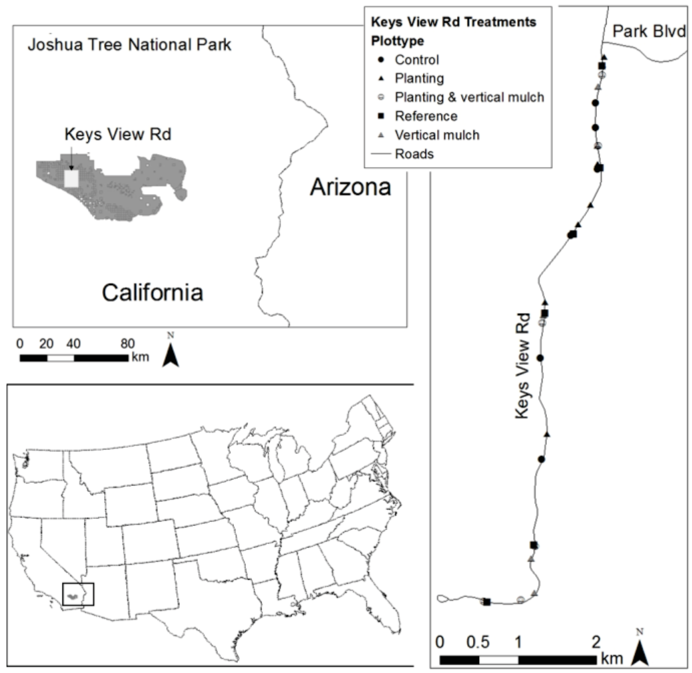


Figure A15. Location of Keys View Road, Joshua Tree National Park, California, USA.
Joshua Tree National Park is located in the southeastern United States. It occupies the southeastern end of California. Keys View Road extends from Park Boulevard to Keys View Point, in the western side of the park. The project site extends a long portion of the road and each plot is directly adjacent to the roadway.

Each plot type has six replicates per treatment. Each plot is 2 m × 20 m, equaling 40 m^2 per plot (Figure A16). A 0.5 by 0.5 frame was used to assess microsite subplots across the larger plot. The subplots were centered on five outplants, five vertical mulch structures, and in five interspaces based off the treatment that the plot received. Interspace subplots were placed in the center of the short axis (midpoint, at approximately 1 m) and at 3, 7, 11, 15, and 19 m. All three plot types had interspace subplots.

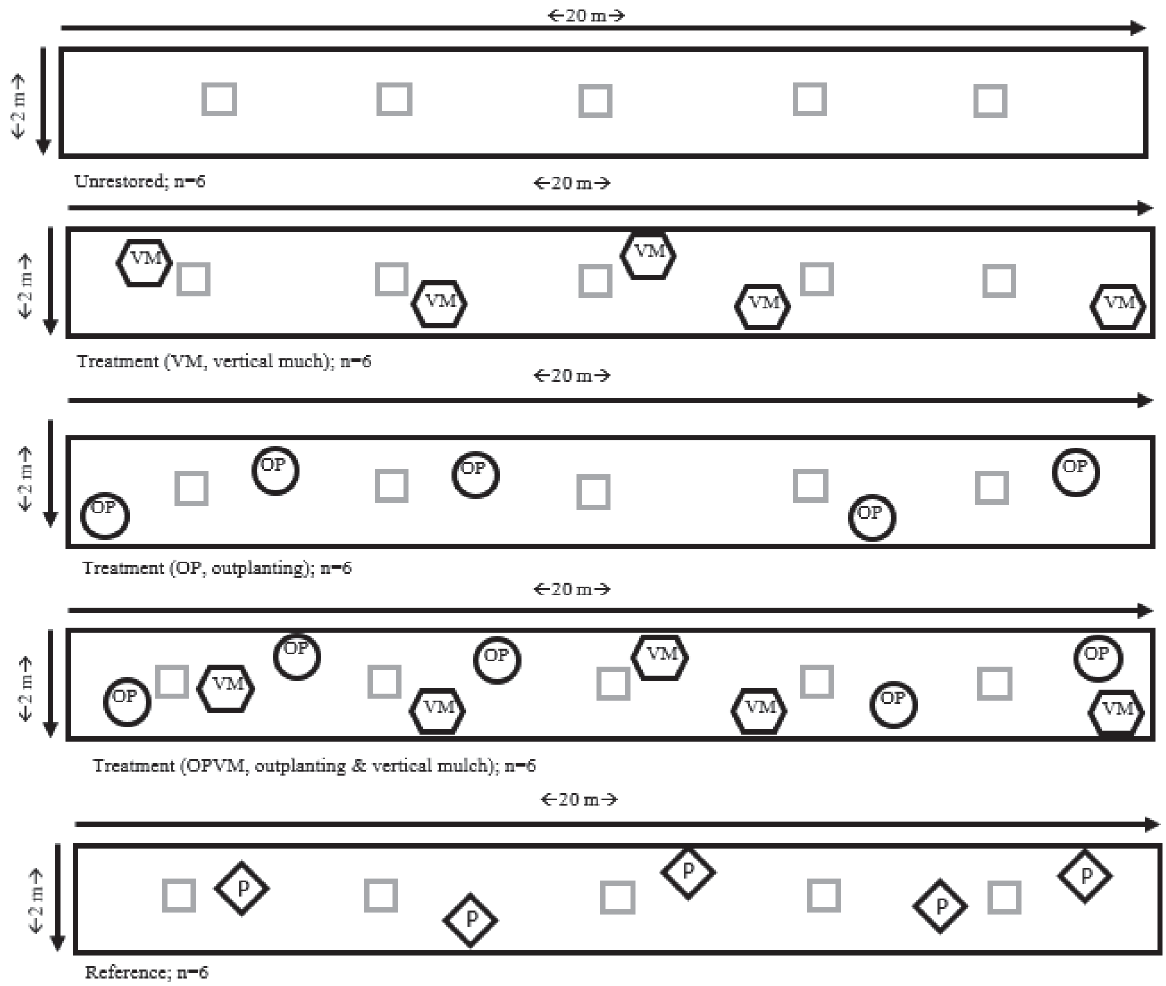


Figure A16. Monitoring schematic and experimental design for assessing disturbance and restoration treatments along Keys View Road in Joshua Tree National Park, California, USA.

The black rectangles represent whole plots (40 m_2; n=30). Gray squares represent interspace 0.5 m × 0.5 m subplots (five per plot). Interspaces were moved to avoid perennial cover or vertical mulch. Hexagons indicate vertical mulch structure 0.5 m × 0.5 m subplots (up to five per plot). Circles indicate outplanting 0.5 m × 0.5 m subplots (up to five per plot). Diamonds indicate live perennial natural plants. Outplanting and vertical mulch microsites varied on availability depending on the year of survey. The six plot types represented are the unrestored and the reference, and the three types of restoration treatment plots. On plots with vertical mulch, we collected the same data for outplants as for five vertical mulch structures within each plot, systematically selecting structures closest to the center of the plot at 2, 5, 9, 13, and 17 m along the 20-m plot axis. On each disturbed plot, we sampled an interspace (≥1 m from the canopy edge of a live perennial plant or vertical mulch structure) nearest to 3, 7, 11, 15, and 19 m (five interspaces total per plot) using a 0.5 by 0.5 quadrat. In reference plots in 2019, no perennial microsites were measured, however they were measured for the previous years' of monitoring.

On plots with just vertical mulch and just outplanting, up to five of each respective treatment structure was measured at the center of the short axis and along the longitudinal axis at 2, 5, 9, 13, and 17 m. Occasionally, there were not enough live outplants or vertical mulch structures and so the subplots number less for that particular plot. On plots that received an outplanting and vertical mulch combination treatment, five interspaces subplots, five vertical mulch and five natural shrub subplots were monitored as close to the standardized markings mentioned previously. In all plots, if there was a shrub or vertical mulch structure in the way, the subplot was adjusted and the new coordinates were recorded. Unrestored plots simply had five subplots spread evenly along the center of the plot. Undisturbed plots had five interspace subplots and five natural perennial plant subplots located at the same meterage as the outplants and vertical mulch. In the 2019 undisturbed plots, no data for perennial natural plants was measured, but was measured for interspace subplots. In cases where there were not a full five of the treatment, as many present were used. Monitoring procedures for 2018 and 2019 follow Abella et al. 2018. Each subplot was measured in the same location throughout the years.

For each subplot, the annual and perennial species richness and percent cover was visually determined using cover classes (1=0-1%, 2=1-2%, 3=2-5%, 4=5-10%, 5=10-25%, 6=25-50%, 50-75%, 8=75-95%, 9=>95%). Species that did not occur in any of the subplots were recorded and given a cover class based off their percent cover within the entire plot.

Because of the different number of outplanting and/or vertical mulch per survey year, microsites were scaled by the proportion of the plot area in which they covered to scale up to cover from microsite to 40-m_2 plot scale. The proportion of the plot area that each microsite (perennial plant, vertical mulch, or interspace) occupied was calculated per plot per year. This proportion was multiplied by the average cover a species had in the microsite. Entire plots were surveyed for species not already detected within microsites to categorize cover of these species at the plot scale.

The site was monitored in 2009, 2010, 2011, 2017, 2018, and 2019. A subset of randomly selected plots was measured in 2017. I conducted a sensitivity analysis to ascertain whether to include the partial 2017 data or to discard it because of its limiting sample size. The sensitivity analysis showed that for most variables, p values did not change the conclusions of the test. When one data set had significant main effects, the other had an interaction that was moderately or significantly different. On no test was the conclusion in direct contrast between the two tests.

Data Analysis

Year and treatment effects on native and exotic cover and species richness of life history groups were analyzed using generalized mixed models (PROC GLIMMIX, SAS version 9.4 2013), with year (2009, 2010, 2011, 2017, 2018, and 2019) and plot type (disturbed/no treatment control; disturbed/outplanting; disturbed/vertical mulch; disturbed/outplanting and vertical mulch; and undisturbed/reference) set as fixed effects and plot as a random effect. Main effects and two-way interactions between year and plot type were analyzed. A sensitivity analysis was performed for potential influences of sampling only three of the six plots per treatment in 2017 by conducting analyses with only the three replicate plots sampled all years compared to the full data set. Conclusions did not qualitatively differ between models and results using the full data available are reported. For all models, if necessary, variables were either transformed to improve normality (cover, $\log_{10}$+1 or arcsine-square root transformed; species richness, square-root transformed), or, if normality was not improved by transformations, distributions were assessed and applied to models (lognormal distribution for cover; Poisson distribution for species richness) and goodness-of-fit tests were examined. Post-hoc tests with Tukey-Kramer adjustments were used to further assess significant effects or interactions ($\alpha \leq 0.05$). Specifically for non-native plants, post-hoc tests with Tukey-Kramer adjustments were assessed using $\alpha \leq 0.10$ as a compromise between Type I and Type II error to reduce falsely inferring that restoration does not affect non-native plants.

Climate

While the region had a plethora of weather stations to choose from, most of the close ones were spotty; usually the data was only partial for the three measurements and also missing at random points in time. Because of this, six weather stations were averaged to get an understanding of the historical climatic patterns at Keys View Road. The weather stations used were as follows: 1) Desert Hot Springs 3.0 NW, CA US (elevation 408.1 m, 33.7 km away); 2) Joshua Tree 2.0 S, CA US (elevation 1039.1 m, 13.3 km away); 3) Joshua Tree 2.6 SE, CA US (elevation 989.7 m, 30.1 km away); 4) Palm Springs Asos, CA US (elevation 124.7 m, km away); 5) Palm Springs, CA US (elevation 129.5 m, 13.4 km away); 6) Thousand Palms 0.7 W, CA US (elevation 77.4 m, 24.5 km away). The weather stations are much lower than even the lowest Keys View Road plots. In addition, the weather patterns are purportedly influenced by the mountains in the region

The shoulders of Keys View Road were restored with an outplanting, a vertical mulch treatment, and a combination of the two in 2008. Three years prior to the start of restoration through to the end of restoration, the average precipitation was about 12.8 cm year$_{-1}$ (Figure A17). The average high temperature was approximately 28.2° C and the average low was 13.4° C. From the end of restoration in 2008 to the first monitoring year in April 2009, average annual precipitation was 11.9 cm; the average high temperature was 28.6° C and the low was 13.3° C. The first year of monitoring (2009) had a mean annual precipitation rate of 7.48 cm year$_{-}$1. The average high was 28.4 °C and the low was 12.9° C. The second year of monitoring in April 2010 had an average yearly precipitation of 30.1 cm. The average high was 27.7 °C and the low was 12.7° C. 6 years later, in the fourth monitoring year (2017), average yearly precipitation was 18.3 cm, the average high was 28.2°C, and the average low was 13.5°C. The fifth monitoring year, annual precipitation was 14.5 cm. The average daily high was 27.8° C and the low was 13.4° C. In the final monitoring year of 2019, the averages from January to September were calculated: yearly precipitation was 16.2. The high and low were 27.9° C and 13.2° C, respectively.

In considering long term data trend of 14 years, the average annual precipitation was 13.0 cm, the average high was 28.2° C and the low was 13.3° C. The restoration year showed overall better trends for plant growth than the long-term average. Precipitation from the end of restoration to the first monitoring year was 91.5% of the 14 year average. 2009 had the lowest precipitation, and it steadily increased to above the 14 year average for every year except 2011.

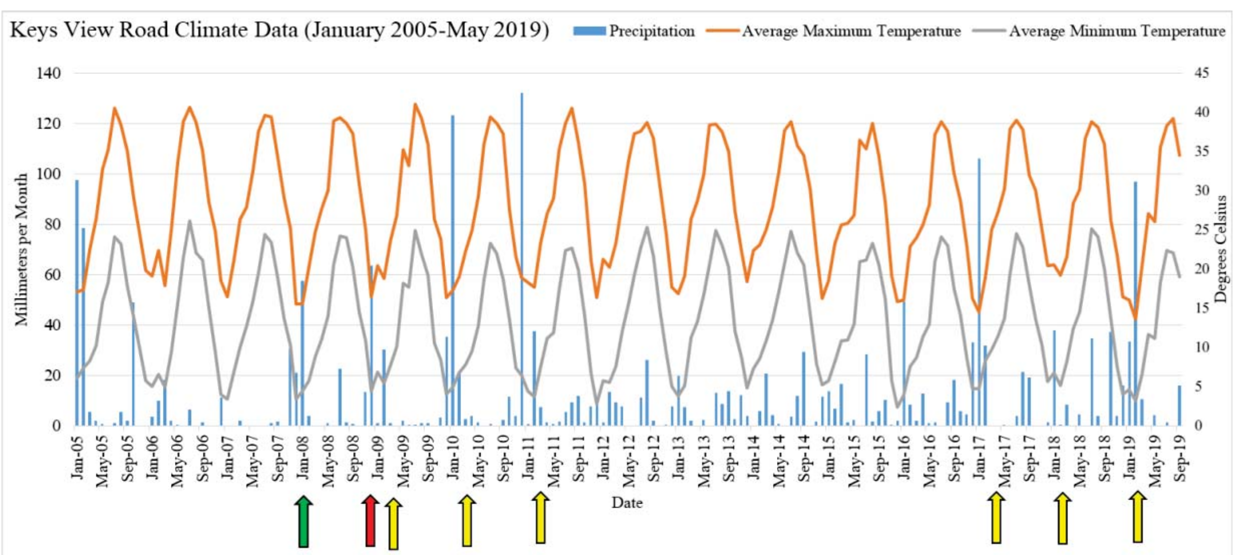


Figure A17. Keys View Road climate data from three years pre-restoration to current time.
Primary y-axis is mean monthly precipitation per year (mm). Secondary y-axis is daily temperature in degrees Celsius. Green arrow denotes approximate commencement of restoration.

Results

Year was often a significant effect among many variables in the study (Table A7). All treatment groups did significantly impact results in some way. Vertical mulch tended to bring down total means, even when included with outplanting treatments.

Table A7. Significant effects on native and exotic annual and perennial plant life history groups.

Main effects include year surveyed (2009, 2010, 2011, 2017, 2018, and 2019), vertical mulching (yes, no), outplanting native perennial forbs and shrubs (yes, no), and vertical mulching and outplanting combination (yes, no).

Variable	Effect	Num DF	Den DF	F Value	Pr > F
Native annual cover	Year x Plot type	20	106	2.61	0.0008
Native annual species richness	Year	5	106	18.41	<.0001
Perennial cover	Year x Plot type	20	85	2.24	0.0055
Perennial species richness	Year x Plot type	20	106	3.24	<.0001
Native perennial forb cover	Year	5	106	3.81	0.0032
Native perennial forb richness	Year x Plot type	20	106	2.26	0.0041
Native perennial grass cover	n/a	-	-	-	-
Native perennial grass richness	n/a	-	-	-	-
Shrub cover	Year x Plot type	20	65	1.97	0.0211
Shrub richness	Year x Plot type	20	106	1.81	0.0287
Exotic annual cover	Year x Plot type	20	105	2.15	0.0067
Exotic annual species richness	Year	5	106	27.98	<.0001
Exotic annual forb cover	Year	5	60	11.13	<.0001
Exotic annual forb species richness	Year	5	106	11.79	<.0001
Exotic annual grass cover	Year x Plot type	20	104	2.05	0.0102
Exotic annual grass species richness	Year	5	106	36.49	<.0001

Considering monitoring years overall, it appears that 2017 and 2019 were better years for plant cover and richness in most regards. 2017 had slightly higher abundances than 2019. The worst years tended to be this first years of monitoring and 2018. Halfway through the study, there was a 6-year monitoring hiatus. Upon examining many of the figures, the years tend to become components of 2 distinct groups: the earlier group with 2009, 2019, and 2011, and the latter group with 2017, 2018, and 2019.

Perennial cover in all categorical lifeforms (i. e. woody plants, forbs, and grasses) is alarmingly low even 20 years after restoration efforts.

Overall perennial cover significantly changed by year and plot type interactions. Perennial cover remained low in all treatment and bare plots in the first three years of monitoring (2009-2011). Gradually, perennial cover rose slightly among all treatment plots. This increase may be of little ecological consequence because restoration plots never exceeded a cover of 4%. Undisturbed plots had the highest cover percentages in all years. Interestingly, the highest value overall occurred in the undisturbed plots in 2018, where percent coverage means exceeded 50%. This is an unusually high value for this part of the Mojave in a good year. The year 2018 was a vegetation-poor season in almost all other regards. All other plot types followed the typical 2018 trend: cover decreased from 2017-2019. Perennial grass cover was moderately significant by year and plot type interactions, but they did not contribute a large portion to overall perennial cover.

The majority of the scant perennial cover can be attributed to perennial forbs and shrub cover, which includes yucca and Cactaceae member. These latter two lifeforms on their own would not have had significant impacts because of their small numbers. Instead, their inclusion with woody perennials, or shrubs, is because of their ecological role. All three lifeforms provide tall, aboveground overstory for smaller annual forbs and grasses, seedling perennials, and abundant fauna. Total shrub cover, which was significant in year and plot type interactions, was extremely low in all years and all treatments except reference plots. Among years and within restoration treatment plots, the highest cover percentage was only 2.1% in the 2017 OPVM plot type. Highest coverage occurred in 2017 and 2019, while lowest covers are found in 2009. Each year, reference plots had the most coverage among plot types. Treatment plots did not change among each other or among years. Perennial forb cover gradually increased over time, although a 2018 decline and a subsequent 2019 recovery was detected. Interestingly, 2018 had the third highest cover among all plot types. The 2017-2018 decline could be die-off or is very likely attributed to an overall smaller stature under water stress conditions. The year 2017 had the highest percentages of perennial forb cover and 2010 and 2011 had the lowest cover.

Generally, all treatments except reference plot types augmented in comprehensive perennial diversity over the years in a year and plot type interaction. Species richness in reference plots decreased by half over the ten-year monitoring period. Vertical mulch plots changed the least among all the treatments. While VM did increase over time, it did not gain substantial diversity like the other treatments and, aside from 2017, remained more similar to unrestored plots. More perennial species could be found per plot regardless of plot type in 2017 and 2019.

Perennial grasses did not make up a substantial portion of overall perennial richness. In a year and plot type interaction, perennial forbs increased over the monitoring period among all plot types, especially plot types that contained outplanting in some capacity (Table A7). It should be noted, however, that these values range from 1-3 species per plot. 2009 and 2010 showed the lowest native perennial forb richness with many plots indicating a mean value of zero until the end of the 2011 monitoring year. Among woody perennials, *Yucca*, and cacti, year and plot interactions indicate that undisturbed plots have the most species present per plot across all year except 2019, where it is surpassed by the two treatments plot types containing outplants. In the years prior to 2019, OP and OPVM species frequencies stay similar to each other while also increase Outplanting may be helping make the treatment plots more similar to the undisturbed plots.

Statistics for native annual cover included only forbs because there were no native annual grasses present at the plots monitored, although they do occur elsewhere in the park. Most percent covers were similar throughout plot type and year in a year and plot type interaction. Covers tended to average around 3% to 6%, although the 2019 reference plot type reached substantially higher than this. Although not significantly different, 2018 does exhibit clearly lower values compared to other years in every plot type, indicative of a poorly-vegetated season. This decline is counterbalanced by a subsequent surge in 2019. Within plot types, there is no clear trend, although there are several examples of a slight increase in cover from the beginning to the end of the monitoring period. Following trends found in other tests, native annual species richness generally increases over time with 2017 and 2019 significantly having the most diversity and 2018 the least diversity.

In the three early years, no difference in overall exotic cover, which includes forbs and graminoids, among plot types is apparent except a slight increase in 2011. In the later years, a sharp decline in 2018 and a subsequent increase in 2019 year and plot type interactions is detected. Overall, annual exotic cover was lowest in 2018, regardless of plot type followed by 2009 and 2010 in all plot categories except reference plots. In each plot type, aside from the outlier-like conditions of a poorly-vegetated 2018, exotic cover has been increasing with time. The bulk of the exotic cover is from exotic grasses, like *Schismus* and *Bromus* species and is driving the increase over exotic annuals because the trend is not as apparent in exotic forbs. Exotic forb cover significantly increased over time, especially in the last three years of monitoring save the dry 2018 conditions. Cover was extremely low in the three early years, never exceeding 0.05%. The year 2011 averages down to zero because while exotic forbs were present, the numbers were too inconsequential for detection. Exotic grasses, on the other hand ranged considerably. The lowest values occurred early in monitoring on bare plots, while values on vertical mulch treatments in 2011 and 2019 soared to nearly 25% cover. Vertical mulch appears to bring exotic cover percentages closer to unrestored conditions, rather than reference conditions. Exotic grass species richness was significant by year, but *Bromus* and *Schismus* were not being differentiated in early years.

Statistical Significance Tests

Table A8. Effects of year and plot type on native and exotic cover and species richness after restoration treatments for the Keys View Road project, Joshua Tree National Park, CA USA.

Plot types include disturbed/unrestored, disturbed/treated, and reference. Significant effects (α≤0.50) are in bold and moderately significant effects (α≤0.10) are italicized.

Keys View Road	Effect	Year			Plot type			Year × Plot type		
	Variable	df	F Value	Pr > F	df	F Value	Pr > F	df	F Value	Pr > F
Exotic	Total annual cover	5, 105	16.58	**<0.001**	4, 25	5.04	**0.004**	20, 105	2.15	**0.007**
	Total annual richness	5, 106	27.98	**<0.001**	4, 25	0.97	0.439	20, 106	0.66	0.853
	Annual forb cover	5, 102	21.98	**<0.001**	4, 25	3.48	**0.022**	20, 102	1.40	0.138
	Annual forb richness	5, 106	18.41	**<0.001**	4, 25	2.76	**0.050**	20, 106	1.22	0.251
	Annual grass cover	5, 104	14.70	**<0.001**	4, 25	8.70	**<0.001**	20, 104	2.05	**0.010**
	Annual grass richness	5, 106	36.49	**<0.001**	4, 25	1.88	0.145	20, 106	1.21	0.262
Native	Annual forb cover	5, 102	21.98	**<0.001**	4, 25	3.48	**0.022**	20, 102	1.40	0.138
	Annual forb richness	5, 106	18.41	**<0.001**	4, 25	2.76	*0.050*	20, 106	1.22	0.251
	Total annual forb cover*	5, 106	7.41	**<0.001**	4, 25	3.19	**0.030**	20, 106	2.61	**<0.001**
	Total annual forb richness*	5, 106	18.41	**<0.001**	?	?	?	?	?	?
	Perennial cover	5, 85	4.96	**<0.001**	4, 25	15.29	**<0.001**	20, 85	2.24	**0.006**
	Perennial richness	5, 106	3.68	**0.004**	4, 25	14.45	**<0.001**	20, 106	3.24	**<0.001**
	Perennial forb cover	5, 106	3.81	**0.003**	4, 25	0.95	0.453	20, 106	1.16	0.307
	Perennial forb richness	5, 106	11.73	**<0.001**	4, 25	0.71	0.595	20, 106	2.26	**0.004**
	Total shrub cover	5, 65	3.55	**0.007**	4, 24	18.90	**<0.001**	20, 65	1.97	**0.021**
	Total shrub richness	5, 106	2.40	**0.042**	4, 25	9.46	**<0.001**	20, 106	1.81	**0.029**

*Total annual forb category includes annuals, annual-biennials, and annual-perennials

A1.6 Fish Hatchery

Study Site and Background

This study was conducted within Lake Mead National Recreation Area, part of the Colorado River system in Clark County, Nevada at a latitude and longitude of approximately 36° 3'52.94"N and 114°49'8.27"W (Figure A18). Elevation of the plots range from 391 to 412 meters. The 0.21-ha area is divided by the River Mountain loop trail and Lakeshore Dr. The site is located on an alluvial fan with one soil association consisting mostly of Typic Torriorthents and the texture classified as mostly sandy loams (Abella et al. 2007). The plant community of the site is typical of low elevations in the Mojave Desert, *Larrea tridentata-Ambrosia dumosa* scrublands.

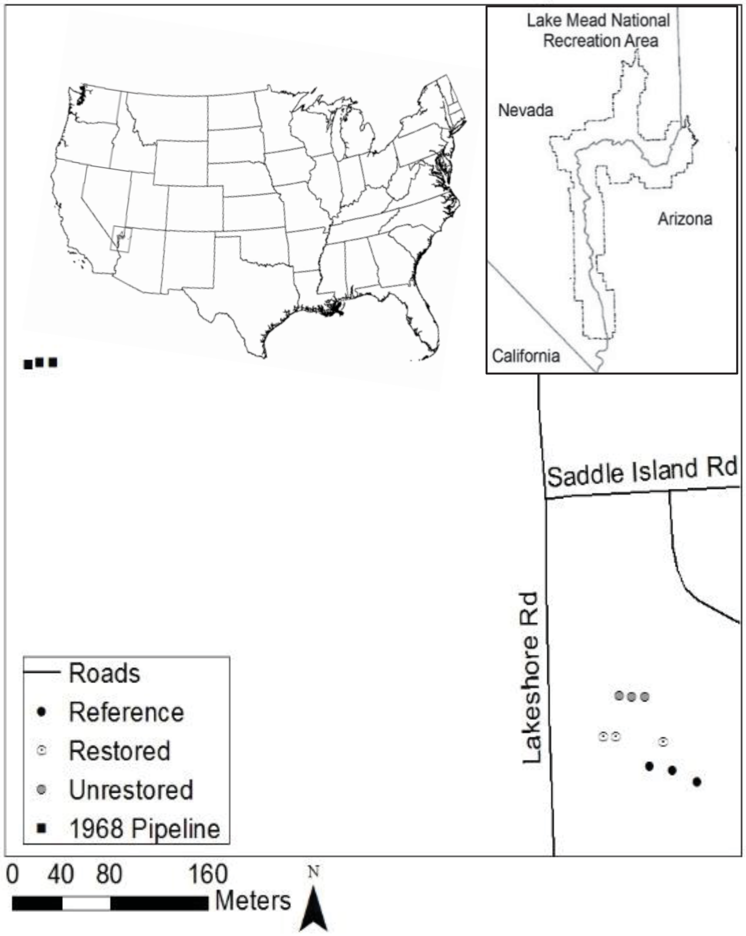


Figure A18. The Nevada Fish Hatchery is located in Lake Mead National Recreation Area. Lake Mead National Recreation Area is in the southwestern United States and follows the Colorado River system.
Lake Mead National Recreation Area encompasses parts of Arizona and Nevada; the Fish Hatchery study takes mace only in Nevada. The Plots are on the east side of Lake Shore Road and are split by the River Mountain Loop trail, which was installed after plot installation.

In 1998, a water pipeline was constructed by the Nevada Water Authority. This new pipeline corridor was in addition to another pipeline installed in 1968. For the purposes of this synthesis study, I have omitted the results involving the 1968 pipeline corridor.

In January and February 1999, the National Park Service implemented restoration to part of the 1998 disturbed pipeline corridor. The National Park Service bladed both the treated and non-treated sections of the 1998 corridor, reapplying the upper 20 cm of topsoil after construction. The National Park Service replaced topsoil, redistributed rocks, and hand-raked the soil surface to ensure an evenness to the topsoil layer to the treated 1998 corridor. They applied an artificial layer of desert varnish (Permeon, Soil-Tech Inc., Las Vegas, NV) to rocks and the soil surface for color restoration. The 1998 corridor then received an outplanting of 96 *L. tridentata* seedlings, 12 *A. dumosa* seedlings, 9 *Opuntia basilaris* seedlings, and 2 *Senegalia greggii* seedlings. The planting treatment is detailed in Newton (2001). 4 years later (2001), no planted *S. greggii* or *A. dumosa* survived, however *L. tridentata* (92%) and *O. basilaris* (100%) showed strong survival rates (Abella et al. 2007).

Four adjacent locations constitute three different plot designs: a 1998 disturbed, unrestored corridor, a 1998 disturbed, restored corridor, and an area that remained undisturbed by construction activities (Figure A19).

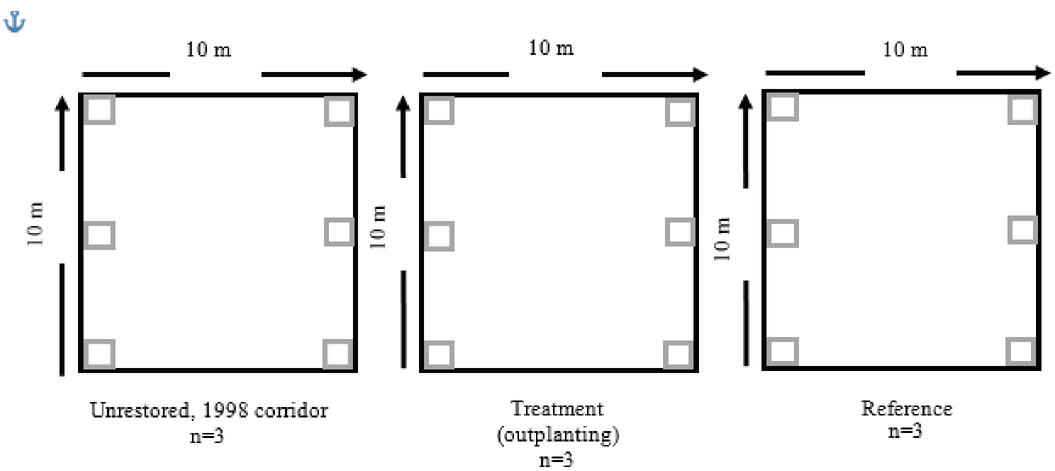


Figure A19. Sampling schematic and experimental design for the Fish Hatchery project site in Lake Mead National Recreation Area, Nevada, USA.

Disturbance (water pipeline installation) occurred in 1998. Restoration included outplanting in 1998. The black squares indicate the whole plot scale (100 m_2). Grey squares indicate the six nested 1 m × 1 m subplots in standard locations in the four corners of the whole plot and at the midpoints along two opposite axes within the whole plot. Plants that were not captured within subplots were recorded and assigned a percent at the whole plot level.

In 2006, a 30 m × 70 m section was delineated in the centers of each of the four areas. Within each of these four sections, seven 10 m × 10 m plots were established. Using simple random sampling, three plots were selected in each area for sampling. Within each plot, six 1 m × 1 m subplots were located at the plot corners and at the midpoints (5 m) of the southern and northern plot lines. All species within the subplot were identified and the aerial percent cover of each plant species was visually estimated. Plots were then surveyed for species that did not occur in the subplots. Nomenclature follows United States Department of Agriculture guidelines.

The first year of field sampling took place between 31 August and 25 October 2006. Field sampling in 2018 was conducted on 3 April 2018 and field sampling in 2019 was conducted on 29 March 2019. For temporal continuity and comparison, I used the same methods described in Abella et al. 2007. 2018 and 2019 monitoring seasons were designed to capture spring annuals, while the monitoring in 2006 captured some autumn annuals and dead remnant annuals. These represent two very different annual communities and thus 2018 and 2019 could not be directly compared to 2006.

To compare soils among the controls and various treatment plots, the top ten centimeters of soil was collected in an interspace between shrubs (>1 m away from a shrub) and underneath a live perennial shrub. Soil was collected on four regions underneath the shrub, halfway between the main stem and canopy edge and combined. In the absence of a long-lived perennial in the plot such as *L. tridentata,* an alternative, often shorter-lived "pioneer perennial" such as *A. dumosa* or *Encelia farinosa* was used (Vasek 1979).

Between the monitoring seasons of 2018 and 2019, one of the plots within the treatment area was destroyed by construction activities. For 2019 monitoring, a new plot was established in the area directly adjacent (within a couple meters) of the destroyed plot. This was then monitored. Data comparisons and a sensitivity analysis show that many of the significant variables between including and excluding the new plot remained the same. Because of the limited difference between them and the repercussions of having a sample size of only two instead of three, the new plot was used in analysis rather than using missing values. The treatment plots cover about 30% of the total area that received treatment and unrestored plots cover about 25% of the total area that did not see any type of restoration.

Data Analysis

Effects of planting treatment and assessment year on native and exotic cover and species richness of life history groups and perennial plant density were analyzed using generalized mixed models (PROC GLIMMIX, SAS version 9.4 2013), with plot type (disturbed/unrestored, disturbed/restored, and unrestored/reference) and year (2006, 2018, and 2019) set as fixed effects and plot as a random effect. If necessary, variables were either transformed to improve normality (cover, $\log_{10}$+1 or arcsine-square root transformed; species richness, square-root transformed), or, if normality was not improved by transformations, distributions were assessed and applied to models (lognormal distribution for cover; Poisson distribution for species richness) and goodness-of-fit tests were examined. Post-hoc tests with Tukey-Kramer adjustments for multiple comparisons were used to further assess significant effects or interactions. Results are considered significant at $\alpha \leq 0.05$ and moderately significant at $\alpha \leq 0.10$.

Climate

Three weather stations were located close to the Fish Hatchery: 1) Alan Bible Visitor Center (elevation 500.2 m, 6.3 km away); 2) Boulder City, NV US (elevation 762.0 m, 9.5 km away); 3) Willow Beach, AZ US (elevation 225.6 m, 25.3 km away). Weather stations are all close to site, but the three stations did not provide any climate data from June 2010 to April 2013. The next nearest weather stations in Henderson, NV USA were missing the same data.

After the water pipeline was installed at the Fish Hatchery, part of the area was outplanted with native perennials and a layer of artificial desert varnish was applied to the ground in 1999. Three years prior to the start of restoration through to the end of restoration (1996-1999), the average precipitation was about 12.9 cm year^{-1}. The average high temperature was approximately 28.6° C and the average low was 15.6° C. From the end of restoration in 1999 to the first monitoring year in 2006, the average yearly precipitation was 11.5 centimeters; the average high temperature was 29.7° C and the low was 15.7° C. During the first monitoring year (2006), the average precipitation was 10.7 cm year^{-1}, the average daily high was 30° C and the low was 15.4° C. In the second monitoring year in 2018, the average annual precipitation of that year was 15.9 cm, the average high was 29.8° C and the average low was 18.6° C. In the final 2019 monitoring year for the months of January to the end of September, the average annual precipitation was 11.53 cm, the average high temperature was 28.6°C and the low was 15.6°C.

The averages for the past 23 years list an average annual precipitation of 11.0 cm, the average high was 28.6° C and the average low was 15.6° C. The year of restoration was relatively close to the 20-year average. The three years following the 1999 restoration, however, showed extremely poor precipitation and an average maximum temperature that was over 1°C hotter. The percentage of the average long-term precipitation that fell in 2000, 2001, and 2002 was only 51.2%, 85.6%, and 34.7%, respectively. After these years, the years remain close to long-term averages until about 2008 through 2010. The first known measurements were taken in 2001 (right in the middle of the drought period) and showed relatively good survival of outplants. Because the next data is not available until 2006, speculating whether this drought did cause a die-off is impossible.

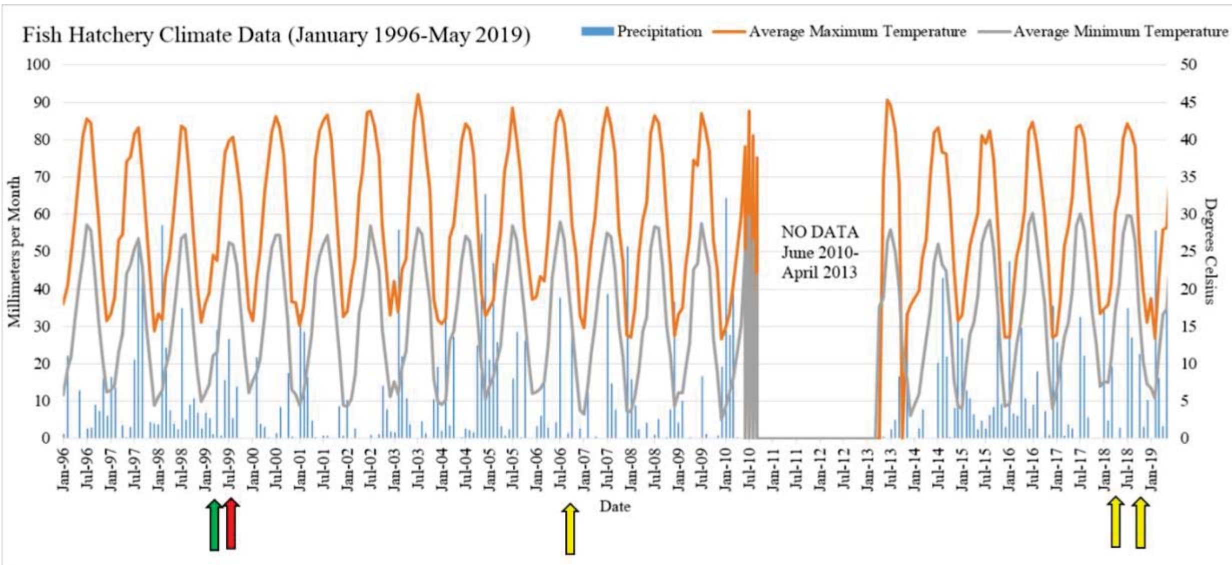


Figure A20. Fish Hatchery climate data from three years pre-restoration to current time.
Primary y-axis is mean monthly precipitation per year (mm). Secondary y-axis is daily temperature in degrees Celsius. Green arrow denotes approximate commencement of restoration.

Results

The Fish Hatchery had some differences among variables, but they were mostly due to the year of monitoring. The variables that were significant by plot type only included: native annual cover, shrub diversity, exotic annual cover, and exotic annual grass cover. Three other variables were significant by year and plot type interactions. Section I: Perennial and annual results without the 1968 pipeline corridor.

To ascertain whether treatment, year, or treatment × year interaction effects are effecting results of the restoration treatment, the 1968 was excluded from this portion of analyses. There are thus three plot types: three 1998-pipeline unrestored, three 1998-pipeline restored (treatment), and three reference plots. Perennial metrics include measurements from 2006, 2018, and 2019. Annual metrics include measurements from 2018 and 2019, only.

Perennial cover significantly differed by year. The year 2006 had less cover than either 2018 or 2019. Cover percentages increase between each monitoring year. During the twelve-year monitoring hiatus of 2006 and 2018, cover increased approximately sevenfold. This averages to an increase of only 0.56% each year, if divided equally across years. During the single year hiatus between 2018 and 2019, perennial cover increased approximately 1.7%. Perennial species richness remained homogenous in both time and space. In fact, no effects were detected for almost all of the life history groups for species richness. Upon further investigation of the exact perennials present within each plot type per year, the only species that was detected only once was *Tiquila latior*. This suggests that between years and treatments, the species richness was relatively homogenous and representative of the creosote-bursage plant community.

To determine what was providing the significant differences in perennial cover, perennials were parsed into more specific life history groups: forbs, grasses, woody perennials (shrubs), and cacti. I found that shrubs were the most substantial component affecting perennial cover because other perennial life history groups were minimal.

The only perennial grass present at the site was *Dasyochloa pulchella*, which did not occur very frequently. D*. pulchella* was only found in unrestored plots and had less than a 0.05% cover. The sole perennial forb, *Cuscuta californica*, is a holoparasitic vine that inhabited in one undisturbed plot in 2019. No significant effects for cactus cover were detected; the only cactus species was *Opuntia basilaris.*

Following from the previous results, it is unsurprising that the "shrubs" category, which here includes plants classified as shrubs, subshrubs, and subshrub-shrubs, varied significantly by year but not by treatment as well. They followed a remarkably similar pattern to total perennial cover, although percentages are different. Cover increased over the years, with 2006 being distinctly lower than the high percentages found in 2018 and 2019. Much of this increase occurred between 2018 and 2019, although the two groups did not significantly differ from one another.

Shrubs were further divided based on their stature. In order of increasing size, the shrub types are forb-subshrubs, subshrubs, subshrub-shrubs, and shrubs. I found that typically larger shrubs types were undergoing significant changes within the monitoring period. The only forb-subshrub present was *Tiquilia latior*, which was only detected in 2006 unrestored plots. Subshrubs which included *Stephanomeria pauciflora*, did not vary within the 1998 pipeline sites. *S. pauciflora* is often implicated as a short-lived and early-colonizing species found in disturbed areas (Vasek 1979). It would be expected to potentially appear in either treatment or unrestored sites, depending on how quickly the recovery trajectory is moving. Subshrub-shrubs included *Bebbia juncea* and *Encelia farinosa.* Subshrub cover was not significant; however the unrestored plots typically had slightly higher values. Subshrub richness was significant. The two species were only found in disturbed plots (e.g. reference and unrestored). Long-lived shrub cover, which included the two co-dominant species and a third *Ambrosia* species, significantly differed by year only. 2019 exhibited significantly more cover of the three species than either 2006 or 2018. In order to determine whether this statistical significance was an artifact of the 2018-2019 plot replacement, the three species were counted in each plot; I determined that the statistic is indeed significant.

Perennial cover and all of its components were only affected by year and not by treatment. Long-lived shrubs increased in cover between the longer period of time (2006-2018) while smaller shrub species increased between 2018 and 2019. The density of perennials that are considered as early invaders, regardless of their life spans, exhibited the most significant changes, especially between year × treatment interactions. Of the six perennial species occurring at the site, five of them are considered to be early colonizers (Vasek 1979), although only three significantly changed.

Larrea tridentata is not only the co-dominant and longest-living species in the area, but is also posited to have a unique role as both a colonizer and climax species (Vasek 1979). *L. tridentata* density changed by plot type. Restored plots had intermediate values between the unrestored and reference plots, although the restored plots are more similar to the unrestored plots. Reference plots had about twice the density per hectare as restored plots and about three times the density compared to unrestored plots. *L. tridentata* density moderately increased over the monitoring year as well.

Ambrosia dumosa, the other co-dominant of the plant community and its relative *Ambrosia salsola* did not significantly change. Both have conflicting literature suggesting whether they are short- or long-lived (Vasek 1979; USDA Plants), but share roles as early colonizers. *A. salsola* is much more likely to be identified as a colonizer in the creosote-bursage community than its relative.

The short-lived *Stephanomeria pauciflora*, an extremely prevalent colonizer, and *Encelia farinosa*, a common plant and also a colonizer both significantly changed. Stephanomeria pauciflora was only found in unrestored, disturbed plots.

E. farinosa exhibited year and treatment interactions. In 2006, eight years after restoration, the density is extremely low among all plot types. By 2018 and 2019, disturbance plots increase. Restored plots in the later years were not significantly different from 2006 results.

In unrestored plots, however a major increase occurred between 2006 and 2018, followed by a slight non- significant decline. Reference plots never had any individuals among years.

Shrubs of all statures and longevities exhibited significant year × treatment interactions. The density of shrubs in the three plot types did not differ in 2006, however they increased in 2018 and 2019. There was a significant increase in shrub density specifically among unrestored plots. Between 2018 and 2019, a slight decrease among unrestored plots suggests some plant die-off between the two years. Shrubs that are considered early-colonizers increased. There was a significant year × plot type interaction, which included all of the aforementioned except *A. dumosa*. During the first surveys, early colonizing shrub density did not significantly differ among plot types. However, in 2018 and 2019, this shrub species group had significantly more individuals in the unrestored plots compared to treatment or reference plots. Early colonizing shrubs densities remained at none to very low in reference plots.

Annual plant communities are subject to precipitation patterns, including both the amount of train and the timing of the rain event. Annuals were examined on an exotic versus native basis. All non-native plants are exotic forbs and grasses. Interestingly, when comparing the three plot types of the 1998-disturbed corridor, there appear to be more effects due to treatment than due to year. The only monitoring years included were 2018 and 2019. Precipitation or an edaphic factor may have caused these results.

Exotic cover was significantly higher in the reference plots. Restored plots had the least amount of cover among all plot types. About seven typical species occurred across the sites and years. When parsed between grasses and forbs, I found that exotic grass cover was driving the exotic cover effect since annual forbs exhibited no statistical significance. Exotic grass cover exhibited the exact pattern as total exotic cover, although in this case, unrestored and restored plots did not significantly differ from one another.

All native annuals were forbs. While they do have different longevities, they were classified under the same group because the climate of the area kills off most species within the year. Native annuals of any longevity, which includes annual (<1 yr.), annual-biennial, and annual-perennial, significantly different by treatment. Restored plots were intermediate between unrestored and reference plots with reference plots having the highest cover. When considering annuals that will only last one growing season, the same trend is apparent.

Significant Results

Table A9. Effects of year and plot type on the cover, species richness and density of native perennials and the effects of year and plot type on cover and species richness of native and exotic annuals.

Results are following restoration treatments (biotic, outplanting) for the Fish Hatchery, Lake Mead National Recreation Area, NV USA. Plot types include disturbed/unrestored, disturbed/treated, and reference. Significant effects (α≤0.50) are in bold and moderately significant effects (α≤0.10) are italicized. Degrees of freedom for all variables including annuals were: Year (1, 6), Plot type (2, 6), and Year × Plot type (2, 6). Degrees of freedom for all perennials-only variables were: Year (2, 12), Plot type, (2, 6), and Year × Plot type (4, 12). The exception, woody perennial cover, had the degrees of freedom as follows: Year (2, 9), Plot type (2, 6), and Year × Plot type (4, 9). [a]Annual measurements regarding degrees of freedom. [b]Perennial exception regarding degrees of freedom,. *Total annual forbs includes annals, annual-biennials, and annual-perennials.

Fish Hatchery	Effect	Year		Plot type		Year × Plot type	
	Variable	F Value	Pr > F	F Value	Pr > F	F Value	Pr > F
Exotic	Total cover[a]	2.46	0.168	91.50	**<0.001**	0.38	0.699
	Total richness[a]	3.56	0.108	0.02	0.981	0.58	0.588
	Annual forb cover[a]	2.32	0.179	0.34	0.725	0.34	0.725
	Annual forb richness[a]	3.00	0.134	0.00	1.000	0.00	1.000
	Annual grass cover[a]	4.01	*0.092*	26.53	**0.001**	0.71	0.530
	Annual grass richness[a]	1.07	0.341	0.01	0.989	1.05	0.408
Native	Annual forb cover[a]	5.45	*0.058*	7.93	**0.021**	1.22	0.359
	Annual forb richness[a]	2.25	0.184	0.27	0.773	0.59	0.581
	Total annual forb cover[a*]	5.02	*0.066*	8.21	**0.019**	1.43	0.311
	Total annual forb richness[a*]	5.02	*0.066*	8.21	**0.019**	1.43	0.311
	Perennial cover	14.81	**0.001**	1.15	0.377	2.66	*0.085*
	Perennial richness	0.27	0.771	2.30	0.181	1.00	0.443
	Perennial forb cover	1.14	0.351	1.14	0.379	1.14	0.382
	Perennial forb richness	4.00	**0.047**	4.00	*0.079*	4.00	**0.027**
	Perennial grass cover	0.40	0.678	4.32	*0.069*	0.40	0.804
	Perennial grass richness	0.80	0.472	4.55	*0.063*	0.80	0.548
	Woody perennial cover[b]	9.03	**0.007**	2.13	0.200	1.34	0.327
	Woody perennial richness	2.00	0.178	1.35	0.328	3.06	*0.059*
	Subshrub cover	0.86	0.448	3.10	0.119	0.84	0.525
	Subshrub richness	2.02	0.176	1.49	0.299	0.50	0.733
	Subshrub-shrub cover	1.70	0.225	3.14	0.116	1.68	0.219
	Subshrub-shrub richness	4.80	**0.029**	8.73	**0.017**	1.60	0.238
	Total shrub cover	12.21	**0.001**	1.22	0.358	1.72	0.210
	Total shrub richness	0.99	0.400	4.03	*0.078*	1.52	0.257
	Total shrub density	15.34	**<0.001**	5.88	**0.039**	10.77	**<0.001**
	Larrea tridentata density	3.20	*0.077*	9.39	**0.014**	2.00	0.159
	Ambrosia dumosa density	3.08	*0.084*	1.44	0.309	1.95	0.167
	Encelia farinosa density	12.98	**0.001**	6.80	**0.029**	11.03	**<0.001**
	Stephanomeria pauciflora density	6.72	**0.011**	9.37	**0.014**	6.72	**0.005**

Appendix II. Fish Hatchery

Table A2.1 Effects of year and plot type on the cover, species richness and density of native perennials and the effects of year and plot type on cover and richness of native and exotic annuals for the 1998 pipeline corridor.

Results are following restoration treatments (biotic, outplanting) for the Fish Hatchery, Lake Mead National Recreation Area, NV USA. Plot types include disturbed/unrestored, disturbed/treated, and reference. Significant effects (α≤0.50) are in bold and moderately significant effects (α≤0.10) are italicized. Degrees of freedom for all variables including annuals were: Year (1, 6), Plot type (2, 6), and Year × Plot type (2, 6). Degrees of freedom for all perennials-only variables were: Year (2, 12), Plot type, (2, 6), and Year × Plot type (4, 12). The exception, woody perennial cover, had the degrees of freedom as follows: Year (2, 9), Plot type (2, 6), and Year × Plot type (4, 9). [a]Annual measurements regarding degrees of freedom. [b]Perennial exception regarding degrees of freedom,. *Total annual forbs includes annals, annual-biennials, and annual-perennials.

Fish Hatchery	Effect	Year		Plot type		Year × Plot type	
	Variable	F Value	Pr > F	F Value	Pr > F	F Value	Pr > F
Exotic	Total cover[a]	2.46	0.168	91.50	**<0.001**	0.38	0.699
	Total richness[a]	3.56	0.108	0.02	0.981	0.58	0.588
	Annual forb cover[a]	2.32	0.179	0.34	0.725	0.34	0.725
	Annual forb richness[a]	3.00	0.134	0.00	1.000	0.00	1.000
	Annual grass cover[a]	4.01	*0.092*	26.53	**0.001**	0.71	0.530
	Annual grass richness[a]	1.07	0.341	0.01	0.989	1.05	0.408
Native	Annual forb cover[a]	5.45	*0.058*	7.93	**0.021**	1.22	0.359
	Annual forb richness[a]	2.25	0.184	0.27	0.773	0.59	0.581
	Total annual forb cover[a*]	5.02	*0.066*	8.21	**0.019**	1.43	0.311
	Total annual forb richness[a*]	5.02	*0.066*	8.21	**0.019**	1.43	0.311
	Perennial cover	14.81	**0.001**	1.15	0.377	2.66	*0.085*
	Perennial richness	0.27	0.771	2.30	0.181	1.00	0.443
	Perennial forb cover	1.14	0.351	1.14	0.379	1.14	0.382
	Perennial forb richness	4.00	**0.047**	4.00	*0.079*	4.00	**0.027**
	Perennial grass cover	0.40	0.678	4.32	*0.069*	0.40	0.804
	Perennial grass richness	0.80	0.472	4.55	*0.063*	0.80	0.548
	Woody perennial cover[b]	9.03	**0.007**	2.13	0.200	1.34	0.327
	Woody perennial richness	2.00	0.178	1.35	0.328	3.06	*0.059*
	Subshrub cover	0.86	0.448	3.10	0.119	0.84	0.525
	Subshrub richness	2.02	0.176	1.49	0.299	0.50	0.733
	Subshrub-shrub cover	1.70	0.225	3.14	0.116	1.68	0.219
	Subshrub-shrub richness	4.80	**0.029**	8.73	**0.017**	1.60	0.238
	Total shrub cover	12.21	**0.001**	1.22	0.358	1.72	0.210
	Total shrub richness	0.99	0.400	4.03	*0.078*	1.52	0.257
	Total shrub density	15.34	**<0.001**	5.88	**0.039**	10.77	**<0.001**
	Larrea tridentata density	3.20	*0.077*	9.39	**0.014**	2.00	0.159
	Ambrosia dumosa density	3.08	*0.084*	1.44	0.309	1.95	0.167
	Encelia farinosa density	12.98	**0.001**	6.80	**0.029**	11.03	**<0.001**
	Stephanomeria pauciflora density	6.72	**0.011**	9.37	**0.014**	6.72	**0.005**

Table A2.2 Effects of year and plot type on the cover, species richness and density of native perennials and the effects of year and plot type on cover and species richness of native and exotic annuals for the 1968 and 1998 pipeline corridor.

Results are following restoration treatments (biotic, outplanting) for the Fish Hatchery, Lake Mead National Recreation Area, NV USA. Plot types include disturbed/unrestored, disturbed/treated, and reference. Significant effects (α≤0.50) are in bold and moderately significant effects (α≤0.10) are italicized. Degrees of freedom for all variables including annuals were: Year (1, 8), Plot type (3, 8), and Year × Plot type (3, 8). Degrees of freedom for all perennials-only variables were: Year (2, 16), Plot type, (3, 8), and Year × Plot type (6, 16). The exception, woody perennial cover, had the degrees of freedom as follows: Year (2, 11), Plot type (3, 8), and Year × Plot type (6, 11). [a]Annual measurements regarding degrees of freedom. [b]Perennial exception regarding degrees of freedom,. *Total annual forbs includes annals, annual-biennials, and annual-perennials.

Fish Hatchery	Effect	Year		Plot type		Year × Plot type	
	Variable	F Value	Pr > F	F Value	Pr > F	F Value	Pr > F
Exotic	Total cover[a]	45.75	<0.001	17.76	<0.001	3.92	*0.054*
	Total richness[a]	2.34	0.165	1.77	0.231	0.92	0.472
	Annual forb cover[a]	10.59	**0.012**	4.96	**0.031**	2.61	0.123
	Annual forb richness[a]	12.00	**0.009**	3.47	*0.071*	4.00	*0.052*
	Annual grass cover[a]	42.57	<0.001	10.17	**0.004**	1.69	0.245
	Annual grass richness[a]	0.19	0.672	0.82	0.519	0.21	0.883
Native	Annual forb cover[a]	9.16	**0.016**	8.97	**0.006**	0.92	0.473
	Annual forb richness[a]	7.21	**0.028**	0.22	0.880	1.11	0.400
	Total annual forb cover[a*]	7.60	**0.025**	9.08	**0.006**	1.06	0.417
	Total annual forb richness[a*]	0.03	0.868	0.41	0.748	0.60	0.631
	Perennial cover	14.45	<0.001	2.32	0.152	3.65	**0.018**
	Perennial richness	0.05	0.953	11.53	**0.003**	2.43	*0.073*
	Perennial forb cover	5.79	**0.013**	4.04	*0.051*	5.26	**0.004**
	Perennial forb richness	2.19	0.145	2.63	0.122	1.42	0.267
	Perennial grass cover	3.56	*0.052*	3.12	*0.088*	1.79	0.164
	Perennial grass richness	2.80	*0.091*	4.80	**0.034**	2.26	*0.091*
	Woody perennial cover[b]	16.68	<0.001	1.57	0.270	0.76	0.614
	Woody perennial richness	1.45	0.263	2.79	0.109	2.05	0.118
	Subshrub cover	5.90	**0.012**	4.12	**0.049**	5.14	**0.004**
	Subshrub richness	0.73	0.497	9.87	**0.005**	1.04	0.436
	Subshrub-shrub cover	1.84	0.192	1.07	0.414	1.13	0.390
	Subshrub-shrub richness	16.06	<0.001	13.37	**0.002**	5.42	**0.003**
	Total shrub cover	11.99	<0.001	2.08	0.182	2.58	*0.061*
	Total shrub richness	1.10	0.358	17.27	<0.001	2.92	0.041
	Total shrub density	24.83	<0.001	6.36	**0.016**	8.67	<0.001
	Larrea tridentata density	3.02	*0.077*	14.94	**0.001**	1.99	0.127
	Ambrosia dumosa density	7.07	**0.006**	5.73	**0.022**	3.17	0.030
	Encelia farinosa density	14.70	<0.001	6.37	**0.016**	9.66	<0.001
	Bebbia juncea density	2.38	0.124	17.50	<0.001	2.38	*0.077*
	Early colonizer density	23.65	<0.001	7.39	**0.011**	9.74	<0.001
	Stephanomeria pauciflora den	30.04	<0.001	31.85	<0.001	44.41	<0.001

Curriculum Vitae

Mary Amanda Balogh
marybalogh@yahoo.com

Education

University of Nevada, Las Vegas **Jan 2017 – Dec 2019**
Las Vegas, NV USA
Master of Science - Biological Sciences 4.0 GPA

Loyola Marymount University **Aug 2011 – May 2015**
Los Angeles, CA USA
Bachelor of Environmental Sciences 3.45 GPA

Swinburne University of Technology ` **Jan 2013 – Jan 2014**
Melbourne, VIC AUS
Exchange: Public & Environmental Health, Environmental Philosophy 4.0 GPA

Professional Experience

Student Grader – Principles of Ecology ` **Sept 2019 – Dec 2019**
University of Nevada, Las Vegas
Interpreted undergraduate ecology tests, short answer and essay style questions
Assigned proper extra credits to students who provided a write-up on hosted seminars

Biological Science Technician – Revegetation **May 2017 – Dec 2018**
Vegetation and Ecological Restoration Branch
Yosemite National Park, USA (National Park Service)
Vegetation monitoring
Invasive plant control
Restoration fieldwork (front-country)

Arid Lands Field Botanist ` **Jan 2017 – May 2017**
Abella Soil – Plant – Water – Stress Interactions Lab
University of Nevada, Las Vegas
Field and laboratory assistant; greenhouse studies of Biological Soils Crusts, invasive grass survival plots; field outplanting and vertical mulch installations and seed collection; longitudinal measuring of annual and perennial plants in Lake Mead National Recreation Area; measuring of

gypsum plots with and without a biocrust inoculation; long-term monitoring of native annual communities of southern Nevada; data entry, SAC, analyses

National Park Ranger - Visitor Use Assistant (Fees) **Jun 2016– Oct 2018**
Katherine Landing Entrance Station
Lake Mead National Recreation Area, USA
Handled money, accountability of stock, and visitor services
PAP rating: Superior

Buyer/Nursery Professional/Cashier – Patio Plants, Herbs, Vegetables **Dec 2015 – Jun 2016**
Nicholson-Hardier Home & Garden Center
Highland Park, TX USA
Maintained healthy customer and vendor relationships, controlled the dissemination of pertinent oral/written horticultural literature, trouble-shooting customer plant problems, maintained quality of merchandise

Nursery Professional & Green Goods Buyer **Jul 2015 – Dec 2015**
Armstrong Garden Center
Los Angeles, CA USA
Maintained healthy customer and vendor relationships, controlled the dissemination of pertinent oral/written horticultural literature, trouble-shooting customer plant problems, maintained quality of merchandise

Consultant **Sep 2015 – Jan 2016**
Element Environmental Consulting
Los Angeles, CA USA
Created individual site Storm water Pollution Prevention Plans; conducted storm water

Succulents and Cacti Artist **Jan 2014 – Aug 2014**
A Prickly Affair: Arranged Succulents
Melbourne, VIC AUS.
Built terrariums and cactus gardens. Propagated succulents in a greenhouse setting. Operations & Sales

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INTERNSHIPS

Santa Monica Bay Restoration Commission **Aug 2014 – May 2015**
(The Bay Foundation)
Loyola Marymount University, Los Angeles, CA USA
Identified & photographed wetland invertebrates in laboratory
Gathered, entered, analyzed, & QAQC'd data for published reports
Dissected, analyzed data for two species of sea urchin for Palos Verdes kelp forest restoration.
Active restoration (Ballona Wetland Ecological Reserve, Malibu Lagoon, Stone Canyon Creek)

Hosted public outreach events including groundwater & energy conservation expos.
Measured biomass, cover on plant identification transects in the Ballona Wetlands Ecological Reserve using transect materials, percent-cover & laser quadrats, biomass materials, & dichotomous keys.

Center for Urban Resilience (CURes) **Aug 2014 – May 2015**
Loyola Marymount University, Los Angeles, CA USA
Taught High School students wetland water hydrology as part of an Education Docent in the Ballona Wetlands Ecological Reserve
Hosted & participated in public outreach events including the Southern California Academy of Science Annual Meeting & Coastal Cleanup Day
Participated in Culver City Rain Garden research measuring the effectiveness of runoff & pollution control via groundwater infiltration

Wetland Botanist Intern **Aug 2011 – May 2012**
Loyola Marymount University, Los Angeles, CA USA
Subjected the seedlings of a common dune-area wetland species (*Lupinus chamissonis*) to varying levels of salinity in a greenhouse setting in order to determine salt tolerances that may increase in the coastal wetlands during sea level rise

SERVICE

Botanist, Volunteer **Nov 2019 – Present**
Nevada State Museum, Las Vegas
Las Vegas, NV USA
Identify plants to species or subspecies level gathered from across the arid Southwest and catalogued

Preventative Search and Rescue, Volunteer **Nov 2019 – Present**
Gold Strike Trail (and others) in Lake Mead National Recreation Area
AZ / NV USA
Move around popular trails and also stand near main trailheads to ensure that each visitor is properly equipped and is aware of potential dangers on the hike surrounding area.

The Animal Foundation, Volunteer **Jan 2019 – Present**
Trained to work with canines and felines

National Park Service Multi-Department., Volunteer **Nov 2016 – Feb 2017**
Lake Mead National Recreation Area AZ / NV US
Interpretation Department:
Assisted with biology/ecology-related school field trips around the Mojave Desert; presented Every Kid in the Park (EKiP) presentations encouraging park visitation; “Swore in” Junior Rangers

Biological Research Department:

Assisted with biological research in long-term vegetation monitoring (*Psorothamnus spinosus,* using GPS, compass, camera, & height stick; *Ferocactus cylindraceus,* using GPS, caliper, & PIT tag technology)

CERTIFICATIONS, PROFICIENCY/TRAININGS, MEMBERSHIPS

Certifications

Desert tortoise handler `	2016
First Aid / CPR & Blood Pathogens	2016
PADI certified Rescue SCUBA diver	2014

Proficiencies & Training

Outdoor First Aid

Knowledge of marine (Caribbean, west USA coast), chaparral (CA), desert (Mojave, Sonoran, Great Basin, Simpson), & forest (Sierra Nevadas, Rockies) ecosystems

Knowledge of coastal & wetland hydrology, water quality testing, plant propagation, horticulture, and botany (plant restoration)

Knowledge of North American botany, with specialty in arid regions

Edible plants and herbology of America

Microsoft Office Suite (Word, Excel, PowerPoint, Access)

Statistical Programs (SPSS, SPS)

Geography Programs (GPS, ArcMap, Google Earth Pro)

Trained in long-term camping principles, including navigation with GPS and compass, safety, Leave No Trace rules

Memberships

American Association for the Advancement of Science

California Academy of Science

Friends of the Nevada State Museum, Las Vegas

Society for Ecological Restoration