5-1-2019

Effects of Ecological Restoration Techniques in Glen Canyon National Recreation Area

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EFFECTS OF ECOLOGICAL RESTORATION TECHNIQUES IN GLEN CANYON NATIONAL RECREATION AREA

By

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Bachelor of Science – Biology
University of Utah
2015

A thesis submitted in partial fulfillment
of the requirements for the

Master of Science – Biological Sciences

School of Life Sciences
College of Sciences
The Graduate College

University of Nevada, Las Vegas
May 2019

UNLV | GRADUATE COLLEGE

Thesis Approval

The Graduate College
The University of Nevada, Las Vegas

November 26, 2018

This thesis prepared by

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entitled

Effects of Ecological Restoration Techniques in Glen Canyon National Recreation Area

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

A better understanding of key ecological restoration techniques can inform land management in the Southwest on restoration options for areas infested by invasive grasses that can pose threats to ecosystems, from changes in nutrient cycling to altered fire regimes. In the semi-arid desert of Glen Canyon National Recreation Area (GLCA), several exotic grasses pose risks to local ecosystems: *Saccharum ravennae,* a relatively new invasive perennial grass, and *Bromus rubens* and *Bromus tectorum,* widespread annual grasses. In this study, multiple ecological restoration techniques were implemented to assess their effects on native and non-native vegetation on sites invaded by the non-native grasses *S. ravennae*, *B. rubens*, and *B. tectorum*. *S. ravennae* seeds were tested for germinability after periods of water submersion to address how fluctuating water levels of Lake Powell within GLCA may affect the spread *S. ravennae*. Results showed that *S. ravennae* populations declined within three months of herbicide treatment and manual removal treatment, but began to return by eleven months post-treatment, suggesting the need for repeated treatments to maintain low populations. Herbicide treatment on *B. tectorum* and *B. rubens* did not significantly decrease overall plot non-native cover; however, revegetation treatments yielded higher native plant cover than all other treatments. While shelters and catchments did not significantly affect survival of transplants on all revegetated plots, select plant species had higher survival rates than others. *S. ravennae* seeds were able to survive up to 16 months underwater, indicating the possibility for *S. ravennae* to survive periodic flooding and indicating challenges for managing this grass.

ACKNOWLEDGEMENTS

I would like to show my appreciation to everyone who has supported this project, both academically and personally. I would like to thank: my committee chair and mentor, Dr. Scott Abella, for the opportunity to work within the traditional homelands of my tribe and for providing support for this challenging endeavor; my committee members, Dr. Dale Devitt, Dr. Carolee Dodge-Francis, and Dr. Jef Jaeger, for their continual support and feedback during the project; and Lonnie Pilkington and the National Park Service at Glen Canyon National Recreation Area for contributions in all areas of the project, including funding, boat and lake access, and additional assistance in the field; the Southwest Exotic Plant Management Team for herbicide application, help in the field, and excellent camp dinners. I would also like to thank the members of the Abella lab, Mary Balough, Dominic Gentilcore, Tiffany Pereira, Audrey Rader, Vivian Sam, Camille Traylor, and Lindsay Chiquoine, for help both in the field and lab and for being great lab companions; and my awesome undergraduate volunteers, Sylvia Gentilcore, Keenan Holland, Vardan Kirakosyan, Carmen Lee, Jenna Schmidley, and Roland Viernes, for being of tremendous help in the field and lab.

A special thank you to all of my wonderful friends and family, including the Hualapai Tribe, who have supported me through the many ups and downs of this endeavor. Without you all, this would have been impossible. Hank'yu! (Thank you!)

DEDICATION

I dedicate this work to my wonderful and lovely daughter, Evie.

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CHAPTER 1: INTRODUCTION

OVERVIEW

Semi-arid ecosystems of the southwest have limited resources, such as low precipitation and sparse vegetation, and can be significantly impacted by climatic and anthropogenic disturbances (Crawford and Gosz, 1982; Okin et al., 2001). Invasive plant species are a form of disturbance that are often spread through anthropogenic means and can cause degradation in ecosystems through alterations of ecosystem processes (D'Antonio & Vitousek, 1992; Germino et al., 2016). In particular, invasive grasses can often alter: fire regimes, biodiversity, water resources, soil stability, and soil characteristics (D'Antonio and Vitousek, 1992; Brooks et al., 2004; Wolfe and Klironomos, 2005; Henderson et al., 2006; Pejchar and Mooney, 2009).

Restoration and conservation practices are becoming increasingly important to protect ecosystem resources and to slow or reverse non-native plant invasions. However, management options are being forced to adapt with the changing environmental conditions since restoration to historical conditions is not always feasible because of irreversible changes in ecosystem functions (Cairns Jr, 2000; Suding et al., 2004). Ecological restoration goals that target sustainable conditions that are able to adapt to current and future climate change may more likely be successful, rather than those that aim for original conditions (Hobbs et al., 2009). Traditional Ecological Knowledge (TEK), the ecological knowledge of indigenous people gathered over a period of time through extensive interactions with their environments, can be useful in the context of ecological restoration (Berkes, 2000). TEK methods of addressing these issues can potentially be applicable to today's environment, such as plant care and cost-efficient methods of land management (Allen et al., 2010).

Glen Canyon National Recreation Area (GLCA), part of the National Park Service (NPS) and located in northern Arizona and southern Utah, covers 507,523 hectares of semi-arid desert and riparian environments. The completion of Glen Canyon Dam on the Colorado River in 1964 created Lake Powell, now a well-known tourist destination for boaters (Bureau of Reclamation, 2018). GLCA also holds special areas of interest, including hanging garden habitats and historical Native American ruins (U.S. Department of the Interior, 2015). Dam and reservoir development, tourism, and grazing have affected the area and potentially enabled the invasions of many exotic plant species in the park, including grasses such as ravennagrass (*Saccharum ravennae*) and brome species (*Bromus tectorum* and *Bromus rubens*). These invasive species are a distinct cause of concern because of their environmental impacts, including domination of vegetation communities, alteration of fire regimes, and competition for water (Link et al., 2006; Melgoza et al., 1990). The spread and impacts of these invasive plants may also affected native plant species that are of cultural use to local tribes. Since protection of natural and cultural resources are priorities at GLCA and the area is of importance to local communities, the use of ecological restoration to combat invasive plants in the habitats within the park is needed.

As previous research is lacking regarding management of ravennagrass and brome infested areas within GLCA, this restoration project was created to compare the effects of various methods of ecological restoration in areas infested with ravennagrass and brome in the park. Restoration techniques for controlling the spread of the invasive species and repopulation of native plant species, using TEK, were implemented in the field to assess and compare their efficacies. Additionally, a laboratory experiment tested the effects of flooding on ravennagrass seed germination. As water levels of the lake may periodically inundate the seed bank near the lakeshore, information on the full submersion of seeds can be beneficial to land managers. Therefore, three studies were conducted:

1. Restoration of areas with existing populations of *Saccharum ravennae*
2. Restoration of areas invaded with *Bromus tectorum* and *B. rubens*
3. Effects of periodic inundation on *Saccharum ravennae* seeds

In the proceeding chapters of this thesis, I lay out the relevant background information for the project and address each specific study in a manuscript format, with the introduction, methods, results and discussion sections.

ECOLOGICAL RESTORATION IN SEMI-ARID ECOSYSTEMS

Ecological restoration in semi-arid ecosystems of the American Southwest is of key importance as climatic and anthropogenic disturbances affect ecosystem functions (Kade and Warren, 2002; Abella, 2010). Desert ecosystems are fragile, often containing sparse vegetation with low amounts of annual precipitation and nutrient availability (Crawford and Gosz, 1982), and are at risk of becoming seriously degraded from disturbances (Okin et al., 2001). Human activities have focused on riparian ecosystems in semi-arid regions, which has significantly changed the hydrology of many such systems. For example, impoundments of rivers and streams from dams and controlled flooding has caused irreversible degradation to ecosystems, including declines in vegetation and animal biodiversity (Poff et al., 1997; Stomberg, 2001). These effects on the ecosystem are compounded by the projections of increased frequency and severity of weather events, such as floods and droughts, as well as fire associated with lightning strikes (Easterling et al., 2000). In turn, these events may lead to opportunities for disturbance-adapted non-native invasive vegetation to establish (Zedler and Kercher 2004), proliferating a cycle of disturbance, as many of invasive species can increase fire frequencies (Abatzoglou and Kolden, 2011). An increase of such events may challenge the efficiency of common restoration techniques, making it increasingly important to establish project goals to promote resilience and adaptation to potential changes in site conditions (Abella et al., 2011).

Removal of invasive species may open up resources for native plants to establish (Luken et al., 1997), although new invasive plants may take this opportunity to establish as well (Sheilds et al., 2015), which should be taken into consideration when planning ecological restoration. Revegetation techniques can be used to fill newly opened niches once invasive plants are treated. Favorable timing of restoration treatments and selection of plants for revegetation that are more likely to survive changes in environmental conditions can improve the restoration results (Abella et al., 2011).

Ecological restoration plans often include management of invasive species, and non-native invasive plant species have been in the spotlight for land managers in the Southwest (D'Antonio et al., 2004; Abatzoglou and Kolden, 2011). Salt cedar (*Tamarix ramosissima*), Russian olive (*Elaeagnus angustifolia*), cheatgrass (*Bromus tectorum*), and others invasive species have continuously been targeted by eradication efforts and restoration studies (Whitson and Koch, 1998; O'Meara et al., 2010). In riparian areas, the woody salt cedar and Russian olive have certain survival traits, such as drought and salinity tolerance in the former (Vandersande et al., 2001) and nitrogen fixation in the latter (DeCant, 2008), that allow them to occupy and eventually dominate riparian vegetation communities.

In arid and semi-arid ecosystems with altered hydrology, changing water tables, periodic flooding, spread of invasive species, and changes in climate are all variables that can affect the outcome of restoration projects. Adapting to compensate for these factors may increase the probability of success and keep non-native invasive plant species from re-establishing. The sensitivity of arid and semi-arid environments cause challenges, but pushing ecological restoration techniques to a resilience-based approach may prove to be fruitful (Hobbs et al., 2009).

Glen Canyon National Recreation Area

The field research described in this thesis was conducted in GLCA located in northern Arizona and southern Utah and established as a US National Park Service unit in 1972. Over 4.5 million tourists visited GLCA in 2017, ranking the park 9 out of 417 of all NPS units for economic benefits (National Park Service, 2018a). GLCA is in a semi-arid region and receives an annual average of 15.2 cm of rain, with average temperatures spanning from 38°C in the hotter months of June and July, and -16°C in the colder months of December and January (National Park Service, 2018b; see Figure 1). Located in the Colorado Plateau region, the elevation of GLCA ranges from 930 m to 2,319 m, with a wide variety of vegetation communities from cold desert shrub to pinyon-juniper woodland (Anderson et al., 2010). Lake Powell flows through the length of the park, forming unique riparian communities as well. The sandstone, primarily Navajo sandstone, contains the majority of the area's seeps and hanging gardens that are recognized as biodiversity hotspots (Fowler et al., 1995; Anderson et al., 2010). The topography ranges from upland plateaus and mesas to deep, narrow canyons. Sandstone is the primary rock type in the park, with occasional finer-grained sedimentary rocks found as well. Carbonate rocks are found mainly in older outcroppings, with metamorphic and igneous rocks being found in river gravels (Anderson et al., 2010).

When at full capacity, Lake Powell's surface water reaches about 1,128 m and has a maximum depth of 170 m near the dam. As the second largest man-made reservoir in the United States behind Lake Mead, Lake Powell is quite extensive at 196 miles long, with 1,960 miles of shoreline and 96 major side canyons. Pre-dam, the temperatures of the Colorado River through this region fluctuated seasonally from 26°C in the summer and near 0°C in the winter. Now post-dam, the water below the dam is 7°C year-round and Lake Powell above the dam averages 26°C at the surface in the summer and below 7°C in the winter (National Park Service, 2015).

Glen Canyon, and the river corridor through the Grand Canyon region more broadly, have been inhabited by humans for over 13,000 years, with at least 55 archaeological sites found within GLCA (USGS Grand Canyon Monitoring and Research Center, 2011). Eleven federally recognized tribes in the southwestern US are known to have cultural ties to GLCA: Havasupai Tribe, Hopi Tribe, Hualapai Tribe, Kaibab Band of Paiute Indians, Las Vegas Band of Paiute Indians, Moapa Band of Paiute Indians, Navajo Nation, Paiute Indian Tribe of Utah, San Juan Southern Paiute Tribe, Yavapai-Apache Nation, and the Pueblo of Zuni (Zagofsky, 2014). Some of these tribes are engaged in resource monitoring in the area because of the concern about protecting natural and cultural resources, including: archaeological sites, tribal origin locations, historic sites, native plant and animal species, geologic features, springs, mineral deposits, and resource collection areas (USGS Grand Canyon Monitoring and Research Center, 2011). For example, Rainbow Bridge National Monument, located in GLCA and a well-known sacred area to local tribes, is co-managed with the neighboring Navajo Nation.

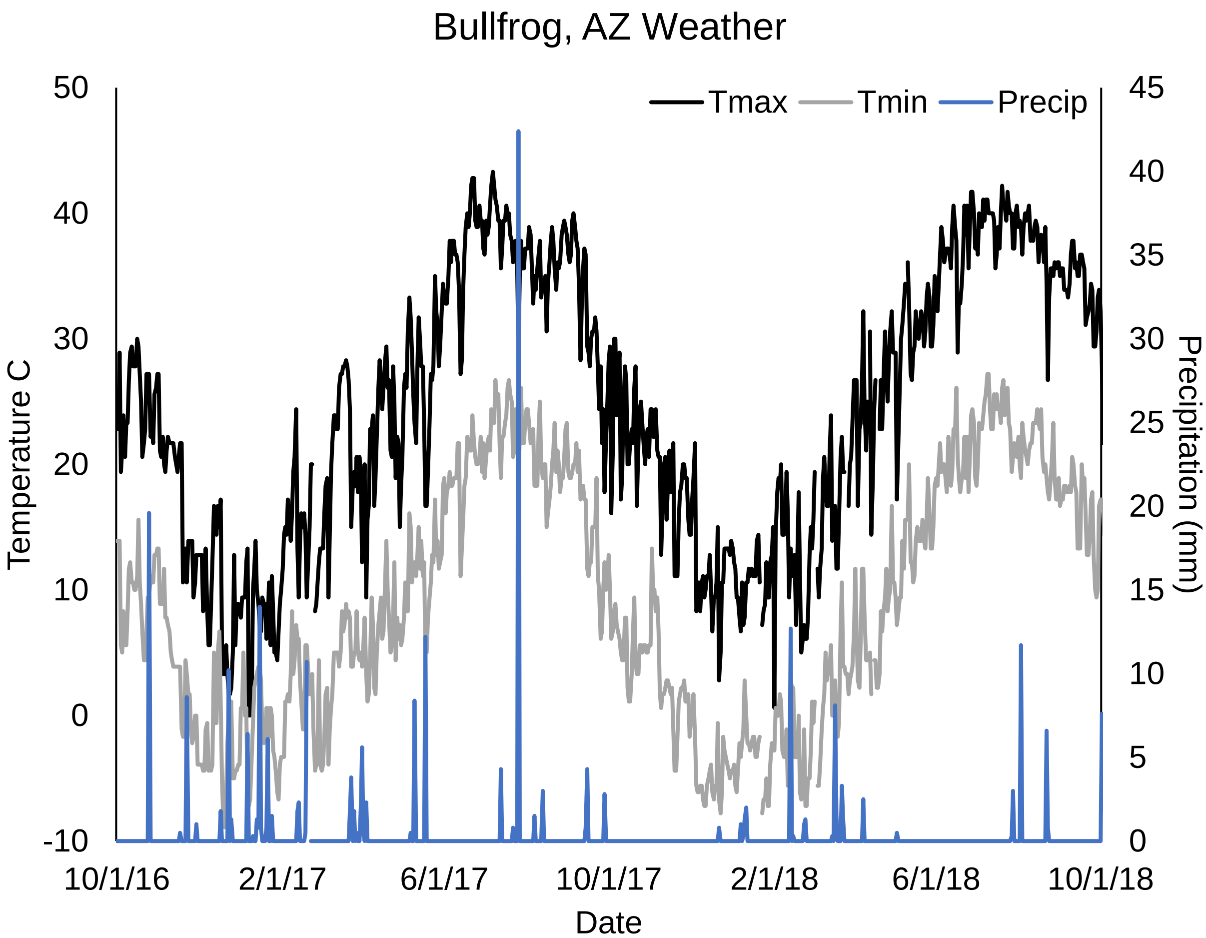

Figure 1. Temperature and precipitation data for Bullfrog, UT. Station located approximately 20 km northwest of closest study site (Slick Rock Canyon) in Glen Canyon National Recreation Area.

SACCHARUM RAVENNAE

Saccharum ravennae is a warm-season, perennial, clumping grass native to the Mediterranean region, Europe, North Africa and central Asia that was introduced into the North America in the 1920s as an ornamental (Pasiecznik, 2015; Firestone, 2007). It is typically found in wet places, such as riparian zones, seeps, and springs, and can also be found in drier areas, such as rocky slopes, grasslands, and roadsides. Ravennagrass produces large amounts of wind-dispersed seeds on plumes at the top of tall stalks, but it can also reproduce from swollen shoot-bases and may re-sprout after damage of the existing root systems (Pasiecznik, 2015). This hardy grass is often noted as being cold tolerant, drought tolerant, and deer tolerant, making it a popular ornamental. However, a drought stress study on the biomass productivity of newly established ravennagrass found the shoot dry weight to be significantly lower under drought versus irrigated conditions, suggesting it does undergo physiological stress in dry conditions (Hattori et al., 2010).

Invasive behavior of ravennagrass was noted in central California in the 2000s (DiTomaso and Healy, 2007) and it has since been listed as a noxious weed or invasive species in several states (e.g., Washington State Noxious Weed Control Board, 2014; Cal-IPC, 2015; New Mexico Department of Agriculture, 2018). Manual and mechanical removal of ravennagrass has previously been effective in similar semi-arid environments (Kearsley and Ayers, 2001), though treatments such as mowing, grazing, and burning are not recommended because of the potential of re-sprouting after damage (Cal-IPC, 2015). This grass has also been shown to be sensitive to chemical treatments that reduce growth (Catanzaro et al., 1993), with glyphosate treatments commonly being used with success (DiTomoso et al., 2013; Cal-IPC, 2015).

Ravennagrass is a particular point of interest and concern for the NPS in GLCA because of the concern for it to impact sensitive habitats and dominate landscapes. While ravennagrass is relatively under control on parts of the San Juan River, a tributary river to Lake Powell, and below the Glen Canyon Dam in Grand Canyon National Park, the plant is still persistent in GLCA, forming monocultures in certain side canyons and overtaking native vegetation. The NPS has implemented chemical and mechanical treatments of ravennagrass within GLCA, though mechanical treatments are difficult and costly and proper research has yet to be conducted in the area regarding effective methods of ravennagrass removal.

BROME SPECIES

Cheatgrass (*Bromus tectorum*) and red brome (*Bromus rubens*) are related winter annual grasses that have spread throughout North America, becoming problematic invasive species. Cheatgrass is prevalent in the Intermountain West of the North America. while red brome is more often found in the arid deserts of the Southwest (Mack, 1981; Salo, 2004). Both species prefer wet winters and dry summers (Novack and Mack, 2001), and germinate in the fall, overwinter with green shoots, flower in the early spring, and senesce in the late spring with seeds surviving until the fall germination. Being able to overwinter and bloom in early spring allows these species to spread seed relatively early in the season and use resources at an earlier time than most native species. Cheatgrass and red brome pose a threat to semi-arid and arid ecosystems, as they tend to establish in open spaces on the soil surface, affecting biodiversity by often creating a homogenous cover of annual grass (Germino et al., 2016).

Since brome grasses are quick invaders and can dominate vegetation communities, other ecosystem functions can be altered (Curtis and Bradley, 2015). Brome can compete with native species for soil water, affect soil nutrient cycling, and increase fire intensity and frequency when dominant on a landscape (Melgoza et al., 1990; Reid et al., 2008; Salo, 2004). Fire frequency is often increased in brome-dominated environments, since the grass can provide greater fuel and coverage for fires to carry across the landscape (Brooks et al., 2004). Following a fire, many desert shrubs and dominant perennial species may be killed, allowing opportunities for brome to invade or expand. Brome can then dominate the landscape post-fire, increasing the frequency of fires and creating a positive feedback loop for its reproduction and spread (Brooks et al., 2004, Abatzoglou and Kolden, 2011).

Treatments on cheatgrass and red brome have been attempted, with some methods being more successful than others. Controlled burning and grazing are often inefficient at controlling the spread of these grasses, which actually may benefit from these activities (Reid et al., 2008). Herbicide treatments are often the most commonly used because of their higher success rate. One study found that herbicide treatments applied on invasive grasses and forbs in shrublands were more effective than raking at reducing non-native plant densities and increasing native plant densities (Steer and Allen, 2010). The timing of herbicide application and type of herbicide used is critical to ensure effectiveness on brome. The pre-emergent herbicide, imapazic, was shown to be more effective than the application of the post-emergent herbicide, glyphosate (Kyser et al., 2007; Munson et al., 2015). Treatment applied early season before emergence occurs is critical; although, mid-season droughts can help the success of herbicide (Steers and Allen, 2010).

While the NPS does not currently treat cheatgrass or red brome in GLCA, treatments may prove to be worthwhile when targeted to assist the re-establishment of native plant species (Melgoza,1990; Salo, 2004). A targeted approach can include uses of both pre-emergent and post-emergent herbicide to reduce the amount of surviving brome that may re-establish.

TRADITIONAL ECOLOGICAL KNOWLEDGE

Traditional Ecological Knowledge is often described as being a cumulative body of indigenous knowledge of an ecosystem gathered through extensive interactions with the environment that is passed down, typically orally, through generations (Berkes et al., 2000). During ecological restoration, TEK can be useful in planning and conducting restoration by providing additional information on local species and historical management practices (Uprety et al., 2012). Native Americans are often viewed as custodians of the environment, with aims to conserve and protect natural and cultural resources (Fowler et al., 2003). Since Native American societies are integrally linked to the biodiversity and natural resource management of the landscape, tribes often have interest in restoration and preservation of areas neighboring or including their current and historical territories.

The NPS has previously worked with Native American tribes to establish agreements that allow the tribes to continue to exercise their cultural practices within parks. Plant collection agreements have been made, such as when Zion National Park and Pipe Spring National Monument worked with the multiple bands of Southern Paiute in Utah, Nevada, and northern Arizona in 1998. Thus allowing the collection of plants and minerals for religious and traditional purposes within park lands by tribal members (Ruppert, 2003). A similar agreement was initiated with Native Hawaiians in Kaloko-Honokohau National Historic Park for traditional and religious uses of plants from specific locations within the park. This plant collection program includes a monitoring program for impacts on native plants from human use and allows both parties to assess when conservation and restoration practices should be implemented. These agreements and methods not only allow cultural practices to continue but contribute to monitoring efforts for conservation and restoration (Ruppert, 2003).

A very similar program has been implemented between federal agencies and local Native American tribes regarding GLCA and the neighboring Grand Canyon National Park (GRCA). The program, Glen Canyon Adaptive Management Program, was created in 2001 over the concern of ecological impacts from the Glen Canyon Dam operations and involved local tribal priorities regarding traditional and cultural resources. This program involves using TEK and scientific knowledge for monitoring of cultural resource conditions to establish a long-term cultural resource monitoring program to assess the effects of the operations of the Glen Canyon Dam, primarily downstream in GRCA (Grand Canyon Monitoring and Research Center, 2011).

In addition to resource monitoring, TEK may have an important application when it comes to restoration ecology in GLCA. TEK co-evolves with the surrounding environment and may be applicable to restoration in an environment with many disturbances, as well as a changing climate, by providing information on reference conditions (Uprety et al., 2012). By providing missing information on species biology and behavior, ecological niches and communities, land management, and farming strategies, TEK has proven to be useful in ecological restoration (Uprety et al., 2012). TEK methods of conserving culturally important species has been practiced many times throughout history and can continue to be applied to modern restoration and conservation efforts (Berkes et al., 2000; Garibaldi and Turner, 2004). Re-establishment of culturally important plant species in GLCA may motivate tribes to assist with restorative efforts that may allow the continuation of cultural practices (Allen et al., 2010).

Other methods of TEK have been demonstrated to be useful in restoration projects, such as reference conditions and irrigation techniques. TEK can contribute to knowledge of reference conditions of ecosystems by providing information on historical species composition and management techniques that may otherwise be unknown (Moller et al., 2004; Uprety et al., 2012). The use of rain catchments as an irrigation technique derived from TEK has previously been shown to be successful at improving the survival of transplants in the Mojave Desert (Edwards et al., 2000). Additionally, other plant management techniques from TEK have been implemented to improve plant populations that serve as cultural food sources on traditional Timbisha Shoshone tribal lands (Fowler et al., 2003).

Involvement of TEK in restoration can also help motivate interest and synergy from Native American groups in future restoration efforts that may lead to more exchanges of information and be beneficial to all parties involved (Berkes, 2000; Uprety et al., 2012). In this study, prioritizing species by ethnobotanical use and implementing irrigation techniques derived from TEK for restoration in GLCA helps set a vegetative target goal, as well as appeal to tribal cultural values (Allen et al., 2010). While this may seem like a very rudimentarily use of TEK, it is none-the-less important to help restore and conserve resources that play critical roles in the culture and lifestyle of Native American groups (Garibaldi and Turner, 2004).

Chapter 2: Restoration in Areas Invaded by *Saccharum ravennae*

Introduction

In Glen Canyon National Recreation Area (GLCA), *Saccharum ravennae*, or ravennagrass, has populated many of the side canyons that stem from Lake Powell. Some of these areas are where unique hanging gardens or cultural and archaeological sites are located. This is a cause of concern as many hanging garden microhabitats contain narrowly endemic species, with GLCA containing at least 82 endemic vascular plant taxa in hanging gardens (Fowler et al., 1995). Additionally, GLCA is considered of high cultural significance to many local Native American tribes, who have interest in the environmental impacts of the area, and conservation and preservation of archaeological sites within GLCA boundaries are high priority to the National Park Service. Since Because the presence of ravennagrass may cause degradation of semi-arid environments through habitat alteration, information on effective strategies to reduce ravennagrass invasions is needed.

Because of the remote nature of the side canyons in GLCA that have been invaded by ravennagrass, it is hard to implement restoration methods that require transportation of big and heavy equipment, numerous plants for revegetation, and numerous personnel needed for field work. Use of on-site materials and limited equipment is often required for these types of remote situations. Such an approach may also be cost efficient. Traditional ecological knowledge (TEK) can be useful in these situations where knowledge of low-cost techniques and local resources can be used in ecological restoration (Moller et al., 2004; Uprety et al., 2012). Future site conditions must also be taken into consideration to increase chances of success in restoration, including selection of plants for methods that involve revegetation (Abella and Newton, 2009). Choosing native plant species that can endure potential changes in the future climate and are of value to local indigenous groups is a benefit for restoration goals and for encouraging cultural interest from local communities. Support from local groups can also benefit the restoration planning process, potentially leading to more knowledge exchange between TEK practitioners and scientists (Uprety et al., 2012). Given the cultural significance of the research area, information on the use of TEK in restoration, such as species selection, and methods of removal of ravennagrass may be useful for local tribes in their own restoration efforts.

This study attempted to address the efficiency of different methods of restoration in areas populated by ravennagrass in GLCA, using herbicide treatment and manual removal of ravennagrass, and revegetation. Ravennagrass has been shown to be sensitive to growth-inhibiting herbicides (Catanzaro et al., 1993) and glyphosate herbicide has been used as a management technique for ravennagrass in California (DiTomoso et al., 2013; Cal-IPC, 2015). However, no formal studies have been conducted on the effectiveness of glyphosate on ravennagrass populations over time. It was hypothesized: 1) herbicide treatment would decrease live ravennagrass cover over time, 2) complete manual removal of ravennagrass would encourage an increase in native plant cover over time, and 3) active revegetation alongside herbicide treatment of ravennagrass would increase native perennial plant cover and decrease live ravennagrass cover over time. We also hypothesized that the use of shelters made from simple natural materials on site would increase survivorship of the transplants. Knowing the results and implications of these methods could inform future restoration project designs by giving insight into potential methods of treating ravennagrass and provide information on the efficacy of using several native plant species that hold cultural value to local tribes for revegetation in these environments.

METHODS

Two experiments were implemented to reduce ravennagrass populations and to encourage native plant species establishment: 1) ravennagrass removal using two separate treatments methods with no active revegetation, and 2) herbicide treatment of ravennagrass alongside active revegetation with additional post-installation treatments on transplants. Plots were selected using the data from an Exotic Plant Management Team (EPMT) mapping effort done in GLCA in September 2015 provided by the National Park Service. Using locations of stands of *Saccharum ravennae* from that effort, five canyons stemming from the main channel of Lake Powell were chosen (according to parameters listed below): Cottonwood Canyon, Llewellyn Gulch, Cottonwood Canyon, Pollywog Bench, and Slickrock Canyon located at river miles 50, 56, 60, 71, and 81, respectively (Figure 2). All of these canyons have seasonal and perennial streams, containing riparian vegetation where ravennagrass was known to occur.

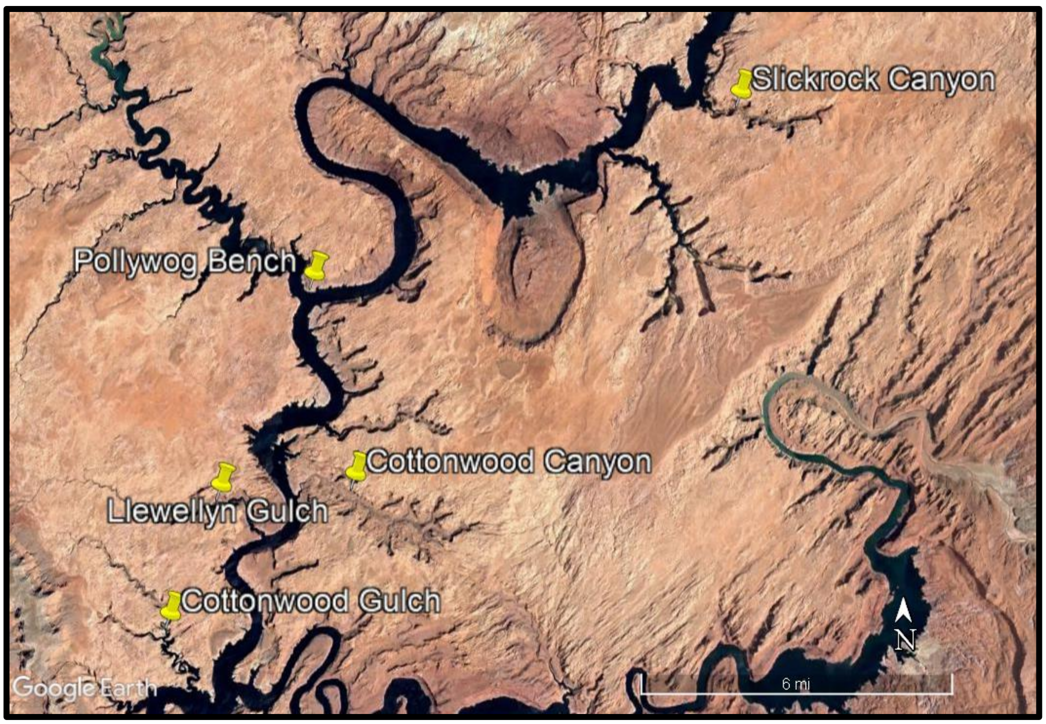

Figure 2. Map of research canyon locations in Glen Canyon National Recreation Area.

Several issues were taken into consideration when finding the placement of the plots within the chosen canyons: accessibility, water level of Lake Powell, and proximity to protected archaeological sites. Some logistical issues that affected accessibility were: boat-friendly access to canyons, and the length and difficulty of hikes to stands of ravennagrass within canyons. Additionally, the water level of Lake Powell fluctuates throughout the year, which affects beach availability and minimum water depth to operate boats, but also may put plots at risk of flooding when water levels are high. Thus, plots were placed in areas that were least likely to be inundated at predicted maximum water level elevation, which was approximately 1119 m in 2017. Lastly, plots were to not be located within a 60 m radius of documented and protected archaeological sites. Plots were placed in an elevation range of 1097 m to 1156 m, with the lowest elevations being unavoidably low due to restrictions of accessibility and ravennagrass locations. Each plot measured 100 m^2 with at least a 5 m buffer space between plots.

Study design

For the first experiment with no active revegetation, live ravennagrass was manually removed from plots using shovels and picks, or the plants were treated with herbicide. The manual removal was done in April and May 2017 in three plots, one in each of three canyons: Slick Rock Canyon, Llewellyn Gulch, and Cottonwood Gulch. The herbicide treatment was implemented in May 2017 on a total of 15 plots, with three plots in each of five different canyons: Slick Rock Canyon, Pollywog Bench, Cottonwood Canyon, Llewellyn Gulch, and Cottonwood Gulch. One reference plot was selected for each treatment plot in similar areas that contained no ravennagrass, excluding Slick Rock Canyon plots, totaling twelve reference plots. The reference plots were used to test if treatment plots differ from non-invaded plots post-treatment. The herbicide mixture used containing 1% imazapyr herbicide, 3% glyphosate herbicide, 0.5% kinetic nonionic surfactant, and 0.5% blue dye, and was applied to the herbicide plots in May 2017 during the start of the ravennagrass growing season and before flowers produce seeds (DiTomoso et al., 2013).

The second experiment with active revegetation had a total of three plots, with one in each of three canyons: Pollywog Bench, Llewellyn Gulch, and Cottonwood Gulch. Revegetation was implemented using transplanting of native species from the areas surrounding the revegetation plots, using plant species from TEK that are of cultural value to local Native American tribes (Table 1). The number of species transplanted per plot was limited to three, although in Pollywog Bench, only two species were transplanted because of limited number of transplantable species available. Cuttings were taken from adult *Baccharis salicifolia*, *Pluchea sericea*, and *Opuntia* individuals (Glenn et al., 1998; García-Saucedo et al., 2005), and entire healthy, vigorous juvenile individuals were transplanted for the other plant species (See Table 1 for complete species list). The plants were placed randomly throughout plots, in areas that were suitable for the plants, based on the variation of internal plot characteristics such as sun exposure, soil texture, and water availability. For the plots with three species planted, two post-installation treatments were implemented: shelters or no treatment. Shelters were created using on-site materials of sticks and rocks to help protect plants from herbivory and create a microhabitat for shade and wind protection (Biggins et al., 1985; Bainbridge, 2001). While the shelters were not strong enough to stop an animal from knocking them over to eat the plants, they were aimed at deterring animals and somewhat disguise the plants visually. Shelters were maintained when necessary during field visits. Six plants of each of the three plant species received a treatment for a total of 12 individuals of each species planted, resulting in 36 plants total on these plots (Figure 3). Pollywog Bench only received two species planted because of limited number of species available that would be able to withstand seep characteristics of the plot, with near constant presence of water and limited amount of plantable soil space. Herbicide treatment for this experiment occurred in May 2017 using the herbicide mixture described above.

Canyon	Species Transplanted	Cultural Value	Source
Pollywog Bench	*Andropogon glomeratus*	Cooking utensil, thatch	Castetter and Opler, 1936
	Baccharis salicifolia	Starvation food source, basket making, building material, medicine	Castetter and Bell, 1951 Powskey and Bender, 1982
Llewellyn Gulch	*Baccharis salicifolia*	--	--
	Opuntia spp.	Food source, dye	Elmore, 1944 Powskey and Bender, 1982
	Pluchea sericea	Food source, crafts, tools, building material, medicine	Powskey and Bender, 1982 Weber and Seaman, 1985
Cottonwood Gulch	*Aristida purpurea*	Ceremonial item, brushes and brooms, doll decoration	Colton, 1974 Vestal, 1952
	Baccharis salicifolia	--	--
	Sporobolus cryptandrus	Food source	Colton, 1974 Elmore, 1944

Table 1. List of native species transplanted onto plots by canyon with their cultural value in Glen Canyon National Recreation Area.

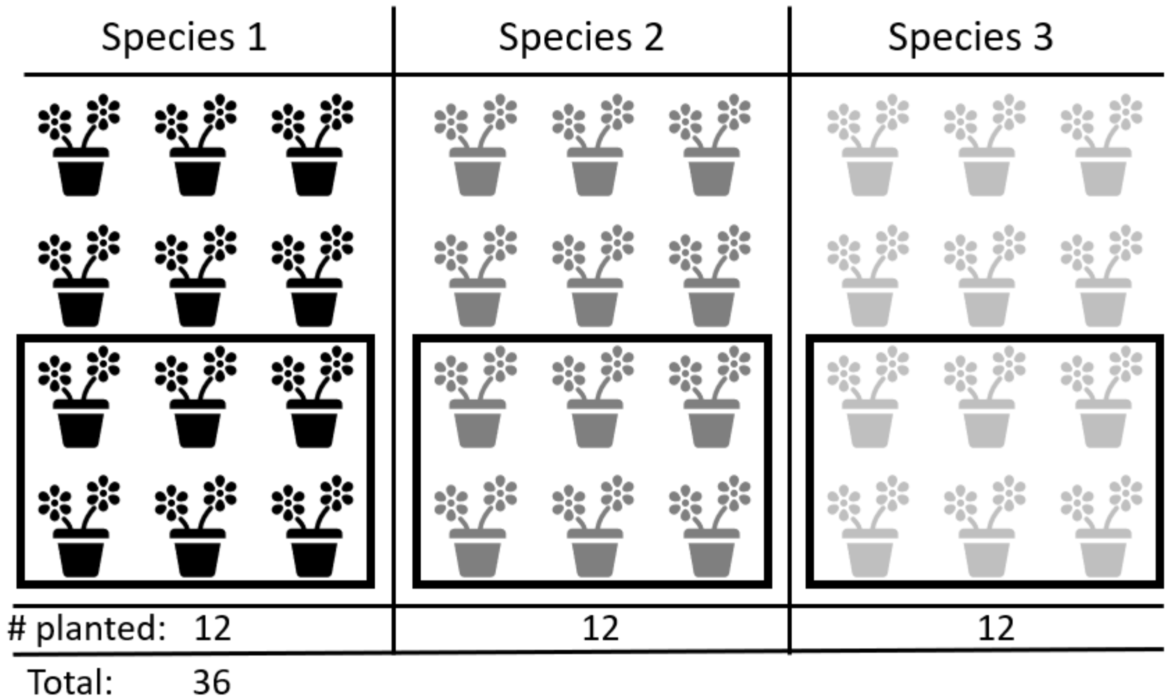

Figure 3. Revegetation design, excluding random planting placement within plots. Twelve individuals of each species were planted, of which half received shelter (shown in square outline), resulting in 36 individuals transplanted in each plot for Cottonwood Gulch and Llewellyn Gulch. Pollywog Bench had two species transplanted, resulting in 24 individuals in the plot.

Plot monitoring occurred in March 2017 to collect pre-treatment data, and in August 2017, April 2018, and October 2018 to collect post-treatment data. Data included repeat plot photography, perennial plant cover by percent adapted from Peet et al., (1998), species richness, and survival monitoring of transplants. Reference plots were not monitored in August 2017 because of time constraints. Additionally during August 2017 and April 2018, varying water levels of Lake Powell caused flooding and inaccessibility to one plot in Llewellyn Gulch and one plot in Pollywog Bench. In October 2018, low water levels caused inaccessibility to Llewellyn Gulch and a majority of plots in Cottonwood Gulch. The data collected that month was incomplete and inappropriate for statistical analysis and was therefore excluded.

Statistical analysis

Response variables of available data were statistically analyzed within years using a mixed-model analysis of variance including canyons as block, plot as a random variable, and treatment using SAS software (v. 9.3). Native and non-native plant cover and richness were statistically analyzed comparing treatment plots to reference plots using data gathered in March 2017 and April 2018. Other data points were not analyzed due to lack of complete data (low or high water affecting access; see results). Data were analyzed for the main fixed effects of plot type and canyon, with plots blocked by canyon and interactions between plot type and canyon using a two-way ANOVA. March 2017 and April 2018 data were analyzed separately. Following the same statistical model, repeated measures analyses were run to assess if ravennagrass cover varied among treatment plots, by canyon and by time.

Results

Herbicide treatment

Native and non-native plant cover and richness

No significant differences were found between plot types or canyons for native and non-native species richness and cover before treatments ($p > 0.05$). There were also no differences between reference and treatment plots for non-native plant richness and cover after treatments ($p > 0.05$). Among all plots, non-native plant cover significantly declined ($p = 0.007$) from spring 2017 to spring 2018 (Figure 4).

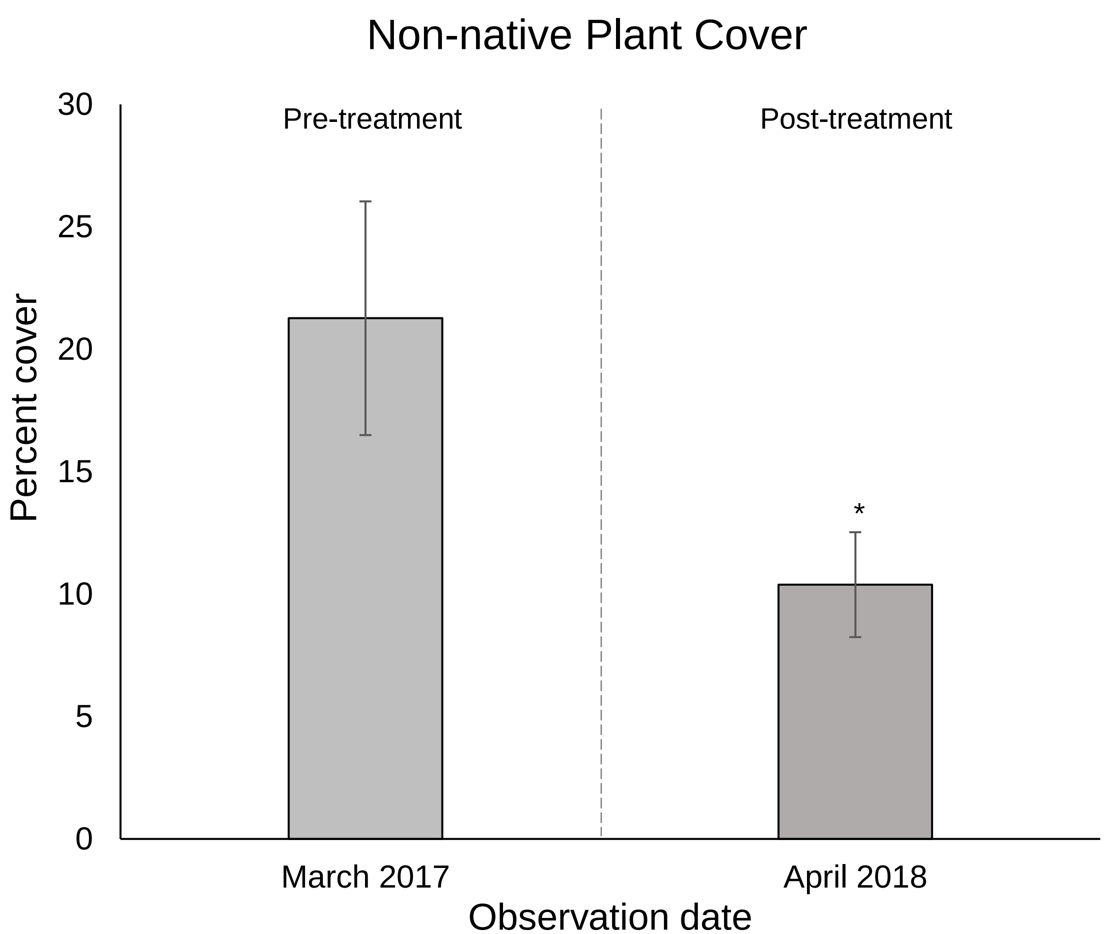

Figure 4. Mean non-native plant percent cover among all ravennagrass plots between years. Cover significantly declined following treatments in April 2018, denoted by asterisk (p = 0.007). Error bars are ±1 SE.

For native plant richness, no significant differences were found between plot type or canyon, nor between years. However, there was a significant difference between canyons in spring 2018, with a significantly higher native plant cover in Llewellyn Gulch than in other canyons ($p = 0.0316$). Treatment plots and reference plots did not differ in 2018.

Ravennagrass cover and density

Ravennagrass density varied among canyons for both years, although differed between years. A significant difference ($p = 0.001$) in Slick Rock Canyon was found among all plots in March 2017 (pre-treatment), with both Slick Rock Canyon and Pollywog Bench having highest mean percent cover (Figure 4). In April 2018, mean ravennagrass cover in Cottonwood Canyon and Cottonwood Gulch was zero, while Pollywog Bench had significantly higher cover ($p < 0.05$) (Figure 5). Mean ravennagrass density among all plots in canyons declined a year after herbicide treatment; however, an increase occurred the following growing season of spring 2018 (Figure 6).

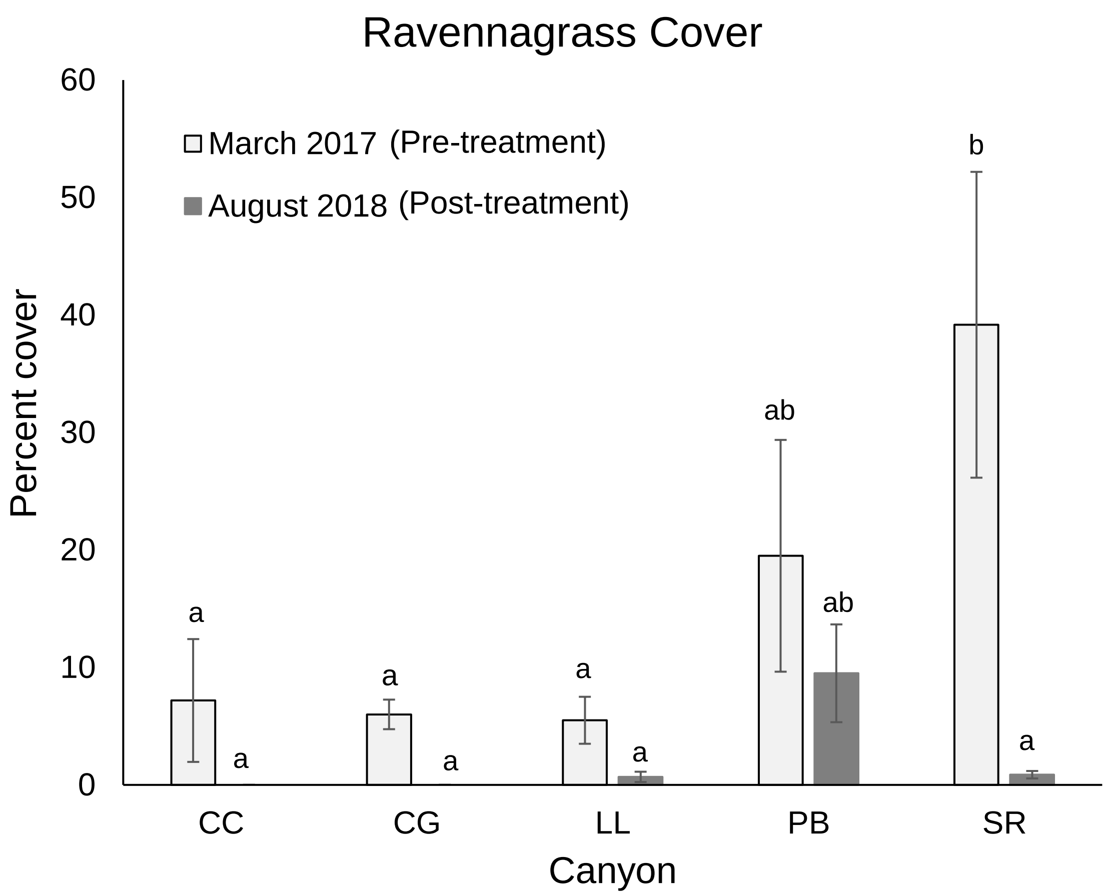

Figure 5. Mean percent cover of ravennagrass among all canyons between spring 2017 and spring 2018. CC = Cottonwood Canyon, CG = Cottonwood Gulch, LL = Llewellyn Gulch, PB = Pollywog Bench, SR = Slick Rock Canyon. Letter denote homogenous groups. SR had significantly higher cover in March 2017 ($p = 0.001$). PB showed significantly higher mean in April 2018 ($p < 0.05$). Error bars are ±1 SE.

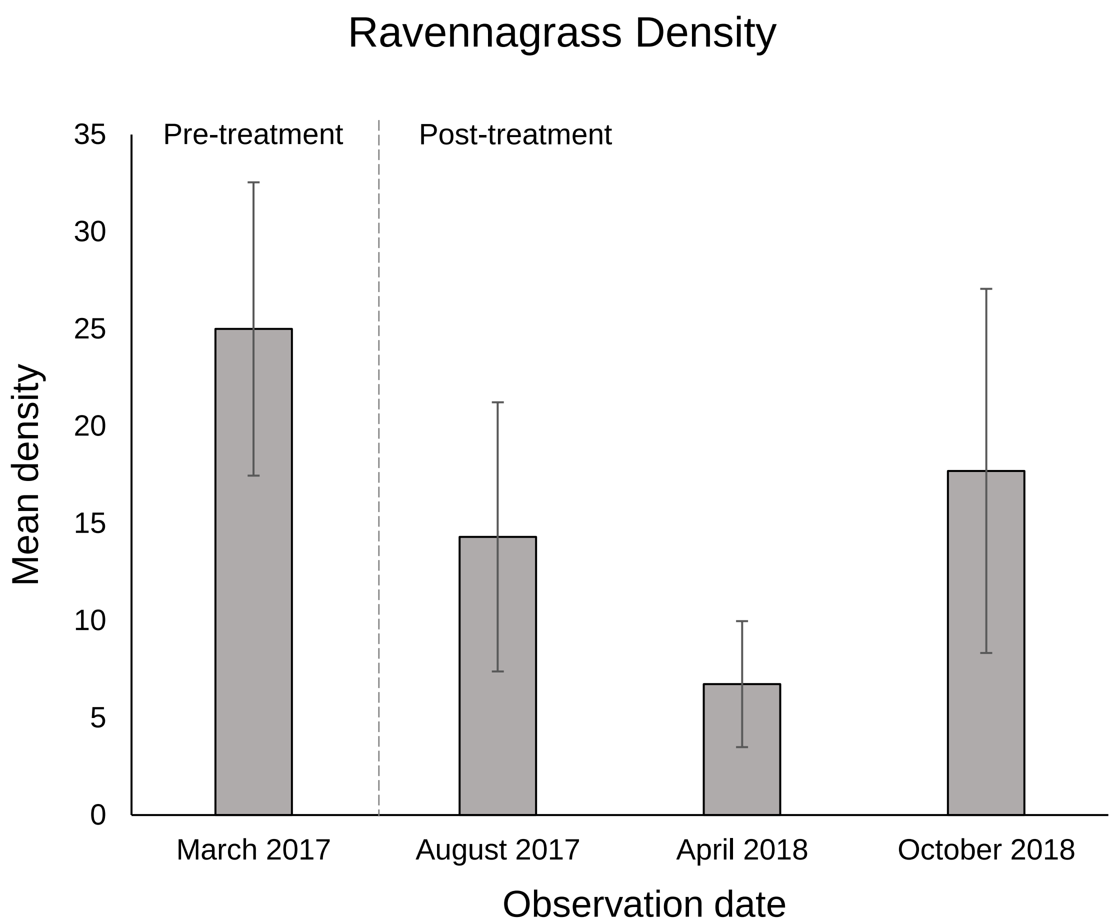

Figure 6. Mean density of ravennagrass after herbicide treatment over time. Density decreased from March 2017 (pre-treatment) to April 2018 then increased from April 2018 to October 2018. Error bars are ±1 SE.

Revegetation and herbicide treatments

Ravennagrass cover and density

Mean percent cover and density for ravennagrass declined from pre-treatment observations in March 2017 to post-treatment observations in August 2017 and April 2018, with significant (Figures 7 and 8). Ravennagrass cover declined from about 13% in August to 0.55% by April 2018, while density declined from 18% to 1.7% during the same time period.

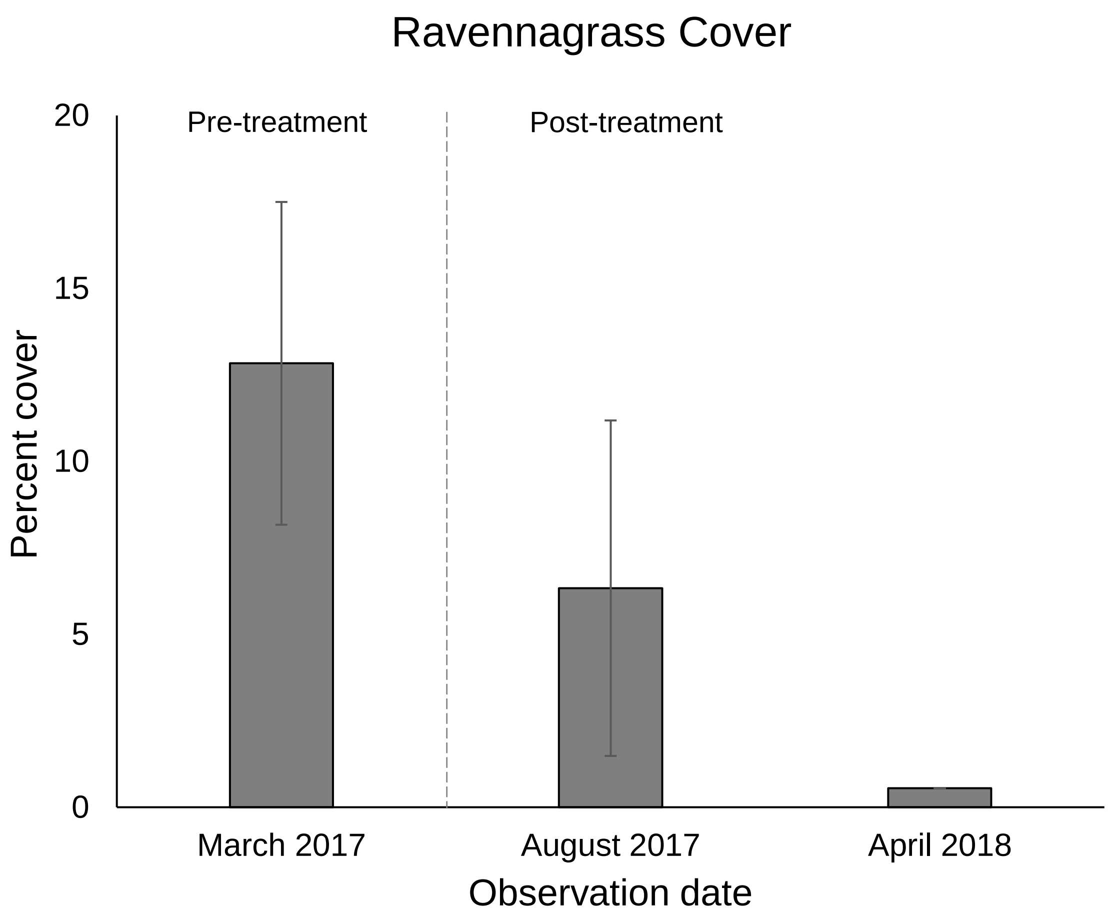

Figure 7. Mean percent cover of ravennagrass on revegetation and herbicide plots over months after treatment. Cover decreased from March 2017 (pre-treatment) to April 2018. Error bars are ±1 SE.

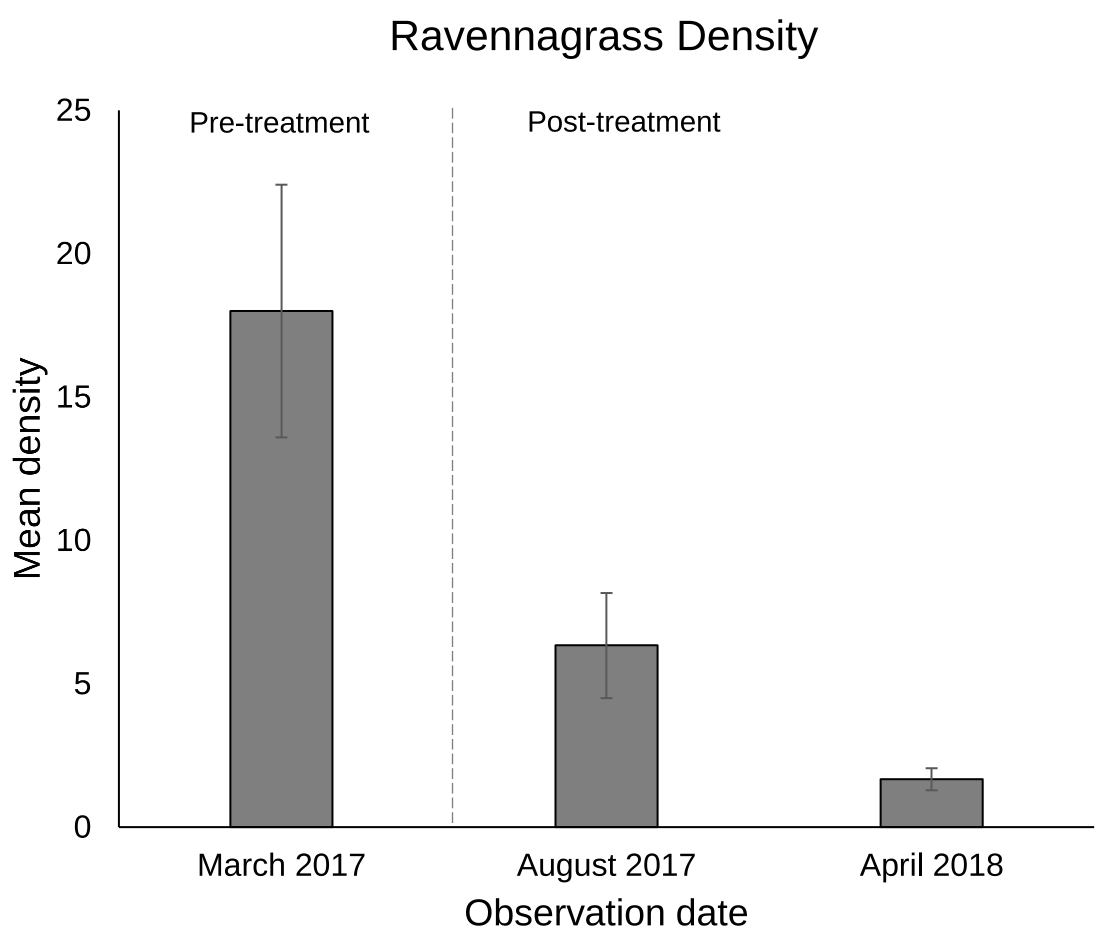

Figure 8. Mean ravennagrass density on revegetation and herbicide plots over months after treatment. Density decreased from March 2017 (pre-treatment) to April 2018. Error bars are ±1 SE.

Native and non-native plant cover and density

Native plant cover increased in plots from pre-treatments observations from March 2017 to August 2018, while cover of non-native plants showed a downward trend over time (Figure 9A). Mean percent native and non-native species richness varied over time (Figure 9B).

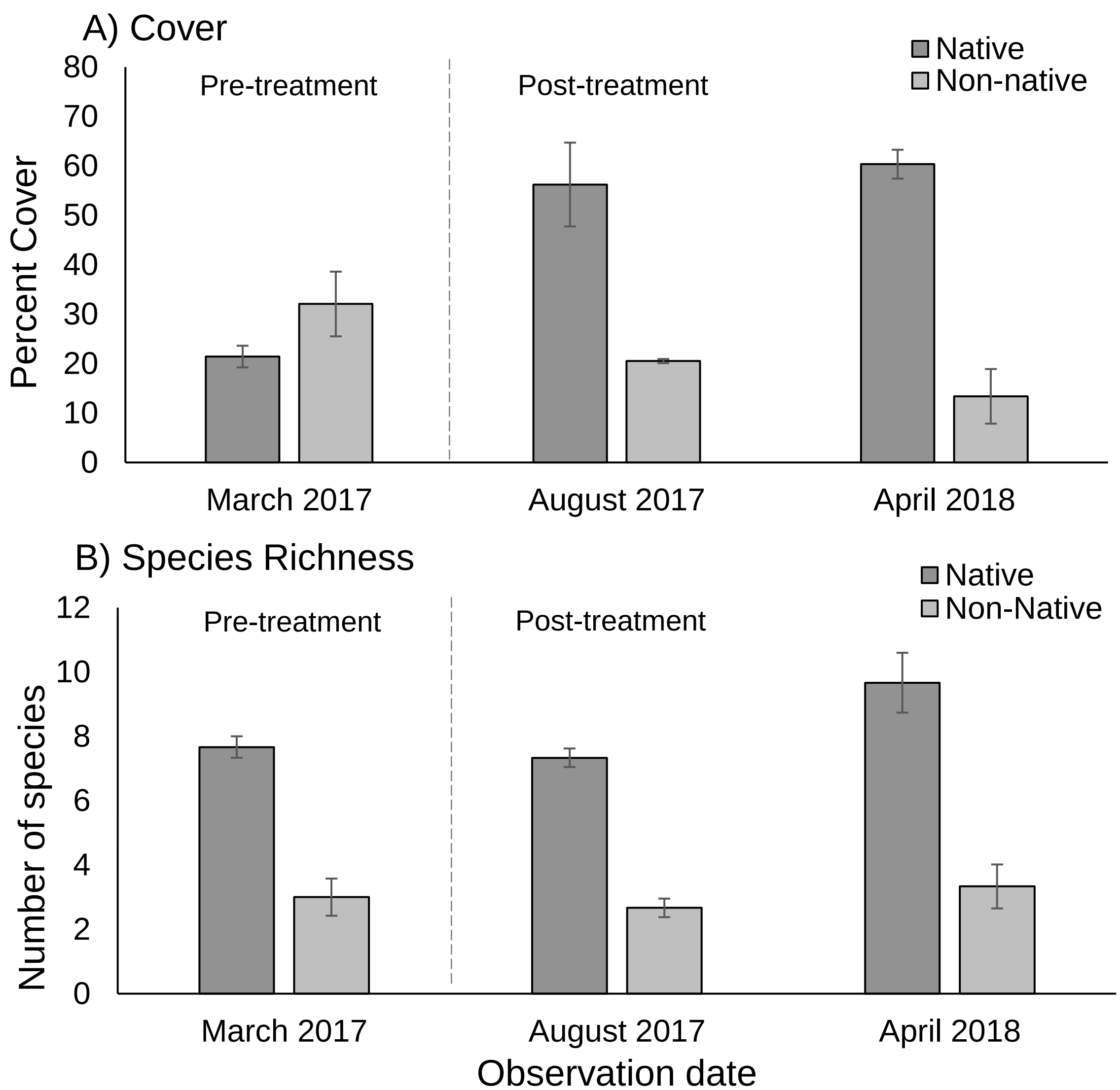

Figure 9. A) Mean percent cover of native and non-native plants before and after revegetation and herbicide treatments on ravennagrass plots. Native plant cover increased, and non-native plant cover decreased over time. B) Native and non-native plant species richness on ravennagrass revegetation plots. Both native and non-native species richness appeared stable over time. Error bars are ±1 SE.

Transplant survival

Mean percent survival of transplant species showed that *Andropogon glomeratus* and *Opuntia* had the highest survival among species (Figure 10). *A. glomeratus* had a 100% survival rate in both years, while *Opuntia* had 75% survival by April 2018. *Baccharis salicifolia* had a percent survival of about 36%. *Aristida purpurea, Pluchea sericea,* and *Sporobolus* had zero percent survival in both years. Rain catchments did not significantly help transplant survival ($p > 0$).

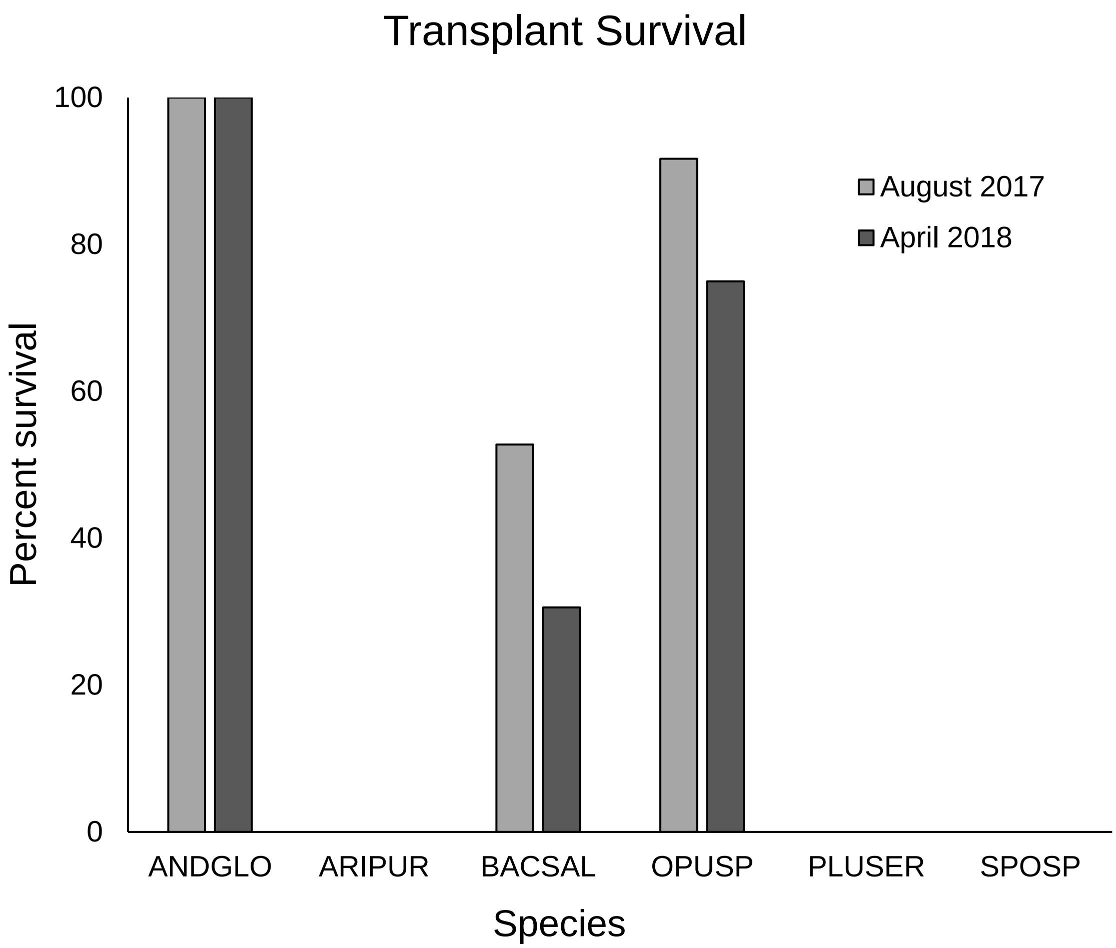

Figure 10. Percent survival of transplant species on ravennagrass plots. ANDGLO = *Andropogon glomeratus*, ARIPUR = *Aristida purpurea*, BACSAL = *Baccharis salicifolia*, OPUSP = *Opuntia* spp*.* PLUSER = *Pluchea sericea*, SPOSP = *Sporobolus* spp. ARIPUR, PLUSER, and SPOSP had zero percent survival in both years.

Manual removal treatment

Ravennagrass cover

Mean percent cover for ravennagrass declined from about 26% to 7.5% within the first five months following manual removal of all ravennagrass on plots (Figure 11). Mean cover of ravennagrass, however, rose until the final observation date in October 2018, 18 months post-treatment, returning to levels similar to pre-treatment data.

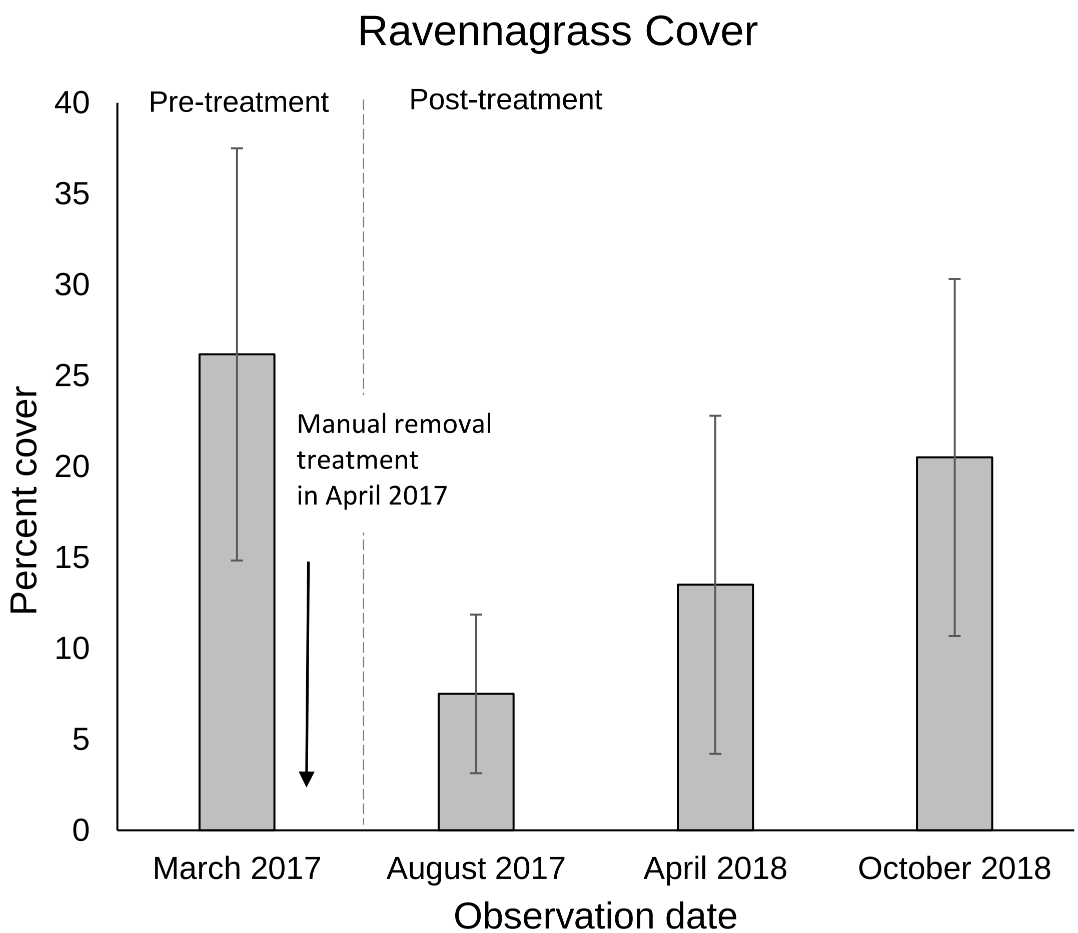

Figure 11. Mean percent cover of ravennagrass on manual removal plots over time. Percent cover continually increased after manual removal treatment in April 2017. Error bars are ±1 SE.

Native and non-native plant cover and richness

Native plant cover remained relatively stable over time. Non-native plant cover appeared to initially decline after manual treatments in August 2017, then began to increase from August 2017 to October 2018 back to levels similar to that of the pre-treatment observation (Figure 12A). Both native and non-native plant species richness remained relatively stable with no significant differences over time (Figure 12B).

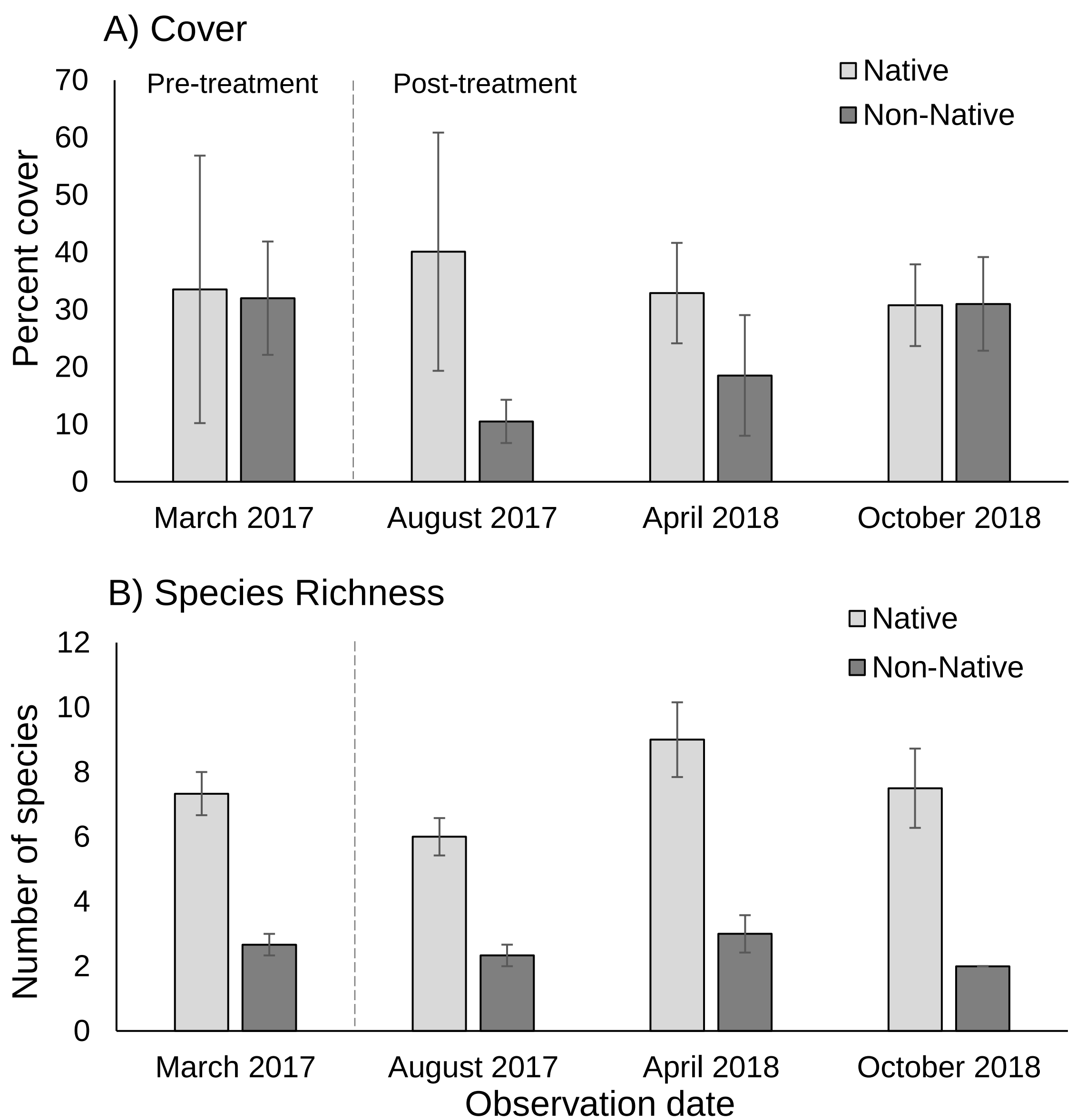

Figure 12. A) Native and non-native plant species cover on manual ravennagrass plots over time. Native cover remained relatively stable over time, while non-native cover had an initial decrease, then increase from August 2017 to October 2018. B) Native and non-native plant species richness on manual ravennagrass plots over time. Both native and non-native species richness appeared stable over all observation dates. Error bars are ±1 SE.

Discussion

Herbicide effects

Herbicide treatment did not appear have negative effects on the native vegetation; however, the lack of significant differences in non-native richness and cover between treatment and reference plots indicates herbicide treatment did not appear to be successful in decreasing non-native species on plots based on the chemical treatments imposed. However, other chemical treatments may prove to be effective. The overall lower non-native cover in April 2018 than March 2017 could have been caused by differences in weather conditions between these years and not treatment effects. The lack of effect the herbicide had on the April 2018 non-native plant cover provides evidence for this perspective. Weather data for the nearest climate station to the study sites in Bullfrog, Utah showed a decrease in precipitation during the summer months from approximately 50 mm in 2017 to approximately 15 mm, potentially creating water-stress conditions for many of the plants (Figure 1). The significant difference in mean native cover between canyons in April 2018 could be attributed to the differences in site conditions, as Llewellyn Gulch is a narrow canyon that has dense thickets of riparian vegetation along perennial streams compared to Pollywog Bench, for example, that is on the shoreline of the main arm of Lake Powell, with little dense vegetation on rocky substrate.

Cover of ravennagrass varied among canyons for both years and was the lowest in April 2018 (Figure 13). Cottonwood Canyon, Cottonwood Gulch, and Slick Rock Canyon had significantly lower cover in 2018 than 2017, indicating impacts from treatment effects. However, the decline and subsequent recovery of ravennagrass densities indicates that while herbicide treatments may have been successful initially, ravennagrass may have survived treatment, growing toward pre-treatment levels in less than two years (Figure 13). Two explanations are that ravennagrass shoot or root material may have survived treatment and produced new growth, or existing ravennagrass seeds in the soil seed bank could have sprouted, established, and grown quickly. Repeat photography showed that both explanations were likely, with apparent regrowth of ravennagrass bunches and many juveniles noted during data collection in October 2018 (Figure 14). The ravennagrass recovery seen in this experiment suggests that repeated herbicide application to ravennagrass populations would most likely be needed to sustain low cover and densities.

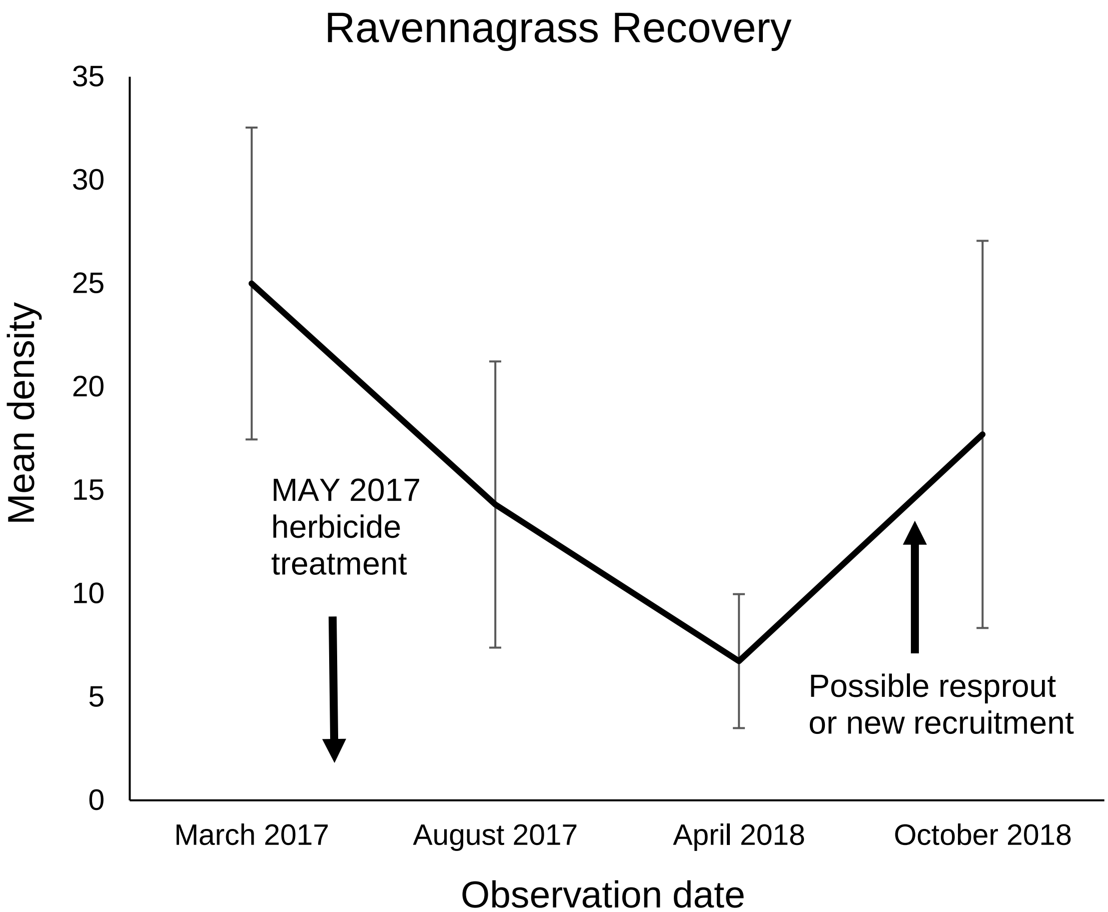

Figure 13. Decrease and following recovery in total mean ravennagrass densities over time. Error bars are ±1 SE.

Figure 14. Repeat photography of Pollywog Bench plot in August 2017 and October 2018.

Herbicide and revegetation effects

Cover of non-native plant species, including ravennagrass, declined while native plant cover increased over time in response to herbicide treatment of nonnative species and revegetation of native plants. This upward trend in native plant cover may have been indicative of successful revegetation efforts, potentially leading to new recruitment of the transplanted plants. The decrease of ravennagrass cover on the plots, could be attributed to the active revegetation and herbicide application. Transplanted plants may have successfully outcompeted surviving or young juvenile ravennagrass for resources, although ravennagrass recovery from herbicide may necessitate the need for additional treatments and monitoring.

Andropogon glomeratus and *Opuntia* species performed the best among all transplants, having the highest survival rates, with *Baccharis salicifolia* showing some success. The zero survival rate for *Aristida purpurea, Pluchea sericea,* and *Sporobolus* species may be due to the specific site conditions of the plots where they were planted. These species were planted on upland slopes, above the riparian corridor within the canyon, and the transplanted plants may not have had easy access to water for establishment. The lack of influence from shelters on transplant survival may be attributed to the quality of the hand-made shelters. Future exploration of shelters could try using pre-fabricated shelters or regularly maintenance of the shelters to potentially improve their effectiveness.

Manual removal effects

Saccharum ravennae cover rebounded over 18 months after manual removal treatment, with the highest recovery in Slick Rock Canyon, returning to pre-treatment cover. This recovery is potentially due to the location of the plot in the riparian corridor where surviving ravennagrass could have had easy access to water. As manual removal is time consuming and physically demanding, the lack of effective results indicates that manual removal is not an effective method for treatment of ravennagrass at sites where soil water is readily accessible or when water is not an issue.

Conclusions

Both herbicide and manual removal of *Saccharum ravennae* resulted in heavy declines in its cover immediately after treatment, but plants began to increase in cover by the following growing season. Treatments were ineffective over time and would likely need to be repeated, annually or bi-annually, to maintain control of ravennagrass populations over time. Active revegetation treatments were shown to yield higher native plant cover, and selection of transplant species that are quick to establish and survive drought conditions, such as *Andropogon glomeratus, Baccharis salicifolia,* and *Opuntia* species, fare best within this semi-arid ecosystem. Since the transplanted species are of interest and value to local Native American tribes, this information can be useful for restoration projects that focus on cultural values. However, the lack of influence on survival from the simple shelters that we observed may indicate that using on-site materials is not sufficient or that other factors, such as water availability, play a more critical role than shelters.

The information from this study can be used to inform land managers and local tribes on the efficacy of these techniques to control invasions of ravennagrass in similar ecosystems, and which plant species are practical for transplantation. Additionally, the involvement of Native American TEK for species selection in this project will hopefully open up opportunities for exchanges of ecological and cultural knowledge between tribes and scientists, supporting and benefiting both groups.

Chapter 3: Restoration Techniques in Areas Invaded by Brome

Introduction

The exotic, invasive grasses *Bromus tectorum* and *Bromus rubens* have long been a problem for land managers in the arid and semi-arid regions of the southwestern US, causing ecosystem alterations, from nutrient cycling to fire processes (D'Antonio and Vitousek, 1992; Germino et al., 2016). According to model predictions, climates in the American Southwest will become warmer in the future (IPCC, 2013; McDowell et al., 2016) with earlier snowmelt and warmer spring weather that can lead to an increase in wildfires (Westerling et al., 2006). Since brome species are quick to establish and spread throughout landscapes through fire disturbances, land managers require efficient methods of restoration and invasive plant management in order to control the presence and spread of these grasses.

In Glen Canyon National Recreation Area (GLCA), both *B. tectorum* and *B. rubens* are prevalent in the landscape. The widespread presence of brome species presents a problem in GLCA by potentially affecting biodiversity and degrading habitats through altered ecosystem processes (Germino et al., 2016) that have special value, such as areas containing archaeological artifacts or cultural resources. GLCA is within or borders traditional territories of Native American tribes in the area, with several more tribes claiming cultural ties to the area (Zagofsky, 2014). Ecological restoration projects in or nearby traditional native homelands can inform tribes on efficient land management practices and may gain the interest and support of the tribes for future projects. This connection between Native American tribes and restoration ecology can be further solidified by use of Traditional Ecological Knowledge (TEK), that can integrate cultural values within restoration projects (Chapter 1).

In this study, various methods of ecological restoration incorporating TEK were tested in areas infested with *B. tectortum* and *B. rubens* in GLCA. The objective was to increase native plant cover and reduce non-native plant cover using various combinations of herbicide treatment and revegetation. Simple and low-cost restoration techniques involving TEK were used in the field, including using culturally significant plants for revegetation and irrigation techniques. Post-installation treatments combinations of handmade shelters and rain catchments, adapted from TEK methods, were applied on transplanted native species to assist with survival on revegetation plots, using materials from on-site. It was hypothesized: 1) the combined use revegetation and herbicide treatment of brome species would result in an increased cover in native perennial vegetation and 2) the combination of the post-installation methods of shelters and rain catchments would increase survivorship amongst transplants.

METHODS

Study design

To assess the effects of using herbicide and active revegetation on areas affected by brome species for ecological restoration, we used a complete randomized block design: presence or absence of targeted plants in both herbicide and revegetation plots. Additional treatments of the presence or absence of shelters and rain catchments at the transplant level were assessed for effects on transplant survival. Sets of four plots were placed in upland slopes that had extensive *B. tectorum* and/or *B. rubens* cover in three canyons: Slick Rock Canyon, Llewellyn Gulch, and Cottonwood Gulch (Figure 2). Each of the 12 plots were 100 m^2 with a minimum 5 m buffer between plots.

Four plot-wide treatments were implemented in each canyon: Herbicide, Herbicide/Revegetation, Revegetation, and Control. Herbicide treatments were implemented by the National Park Service Exotic Plant Management Team (EPMT) using backpack sprayers in October 2017 with a pre-emergent herbicide application mixture of 5 oz/acre imazapic (Plateau® herbicide), 3% glyphosate (Roundup herbicide), 1% methylated seed oil (surfactant), and 0.5% blue dye (for visibility during application). Herbicide was directed at only the brome species, to minimize unwanted effects on native vegetation.

Revegetation occurred in April 2017 using native plants from the surrounding areas for transplanting. If possible, cuttings or translocation of entire healthy juvenile plants were used depending on species. Juveniles of *Achnatherum hymenoides*, *Sporobolus cryptandrus, Sporobolus contractus*, *Artemesia ludoviciana*, *Heterotheca villosa*, and *Atriplex canescens* were translocated. Cuttings of *Opuntia* species pads were planted to a depth of approximately half the height of the pads (García-Saucedo et al., 2005). Since not every canyon held the same plant species for use in revegetation, species planted were not completely consistent among plots (Table 2). Three species were chosen for each revegetation plot, with 24 of each of those species were planted, resulting in 72 total transplants. Species were chosen based on presence of cultural significance (Table 2) (Chapter 2). Immediately after installation, all plants were irrigated with water from the closest running water source. For each species planted, 6 out of 24 individuals received one of the following post-installment treatment combinations: rain catchments, shelters, a combination of both, or no treatment (Figure 15). Simple shelters were built using on-site materials such as dead sticks and rocks to help protect and shade the plants (Chapter 2). Handmade rain catchments, an irrigation technique derived from Native American TEK, were used on select transplants (Edwards et al., 2000). These catchments were built by using shovels, measuring from 0.35 m to 0.5 m in diameter depending on the size of the transplant, to help collect and hold any precipitation that occurred during establishment.

Plot monitoring occurred in March 2017 prior to treatments, and after treatments in August 2017, April 2018, and October 2018 using repeat plot photography, perennial plant percent cover adapted from Peet et al., 1998, and survival monitoring of transplants.

Statistical analysis

Data were analyzed using proc ANOVA in SAS (v. 9.4). Data collected during October 2018 were incomplete due to the inaccessibility of plots in Llewellyn Gulch and Cottonwood Gulch and were therefore excluded from statistical analysis. To first test if native and non-native cover differed among canyons before treatments, we tested the effect of canyon, where plots were blocked by canyon.

Canyon	Species Transplanted	Cultural Value	Source
Cottonwood Gulch	*Achnatherum hymenoides*	Main food source	Elmore, 1944 Murphy, 1990 Stevenson, 1915 Weber and Seaman, 1
	Opuntia spp.	Food source and dye	Elmore, 1944 Powskey and Bender,
	Sporobolus cryptandrus *Sporobolus contractus*	Ceremonial item, brushes and brooms, doll decoration	Colton, 1974 Vestal, 1952
Llewellyn Gulch	*Artemisia ludoviciana*	Topical and oral medicine, ceremonial item	Murphy, 1990 Elmore, 1944
	Heterotheca villosa	Topical and oral medicine	Vestal, 1952 Whiting, 1939
	Opuntia spp.	--	--
	Sporobolus spp	--	--
Slick Rock Canyon	*Atriplex canescens*	Ceremonial item, topical medicine, dye, food source	Colton, 1974 Elmore, 1944 Weber and Seaman, 1985
	Sporobolus spp.	--	--
	Opuntia spp.	--	--

Table 2. List of native plant species selected for transplantation by canyon and their cultural value.

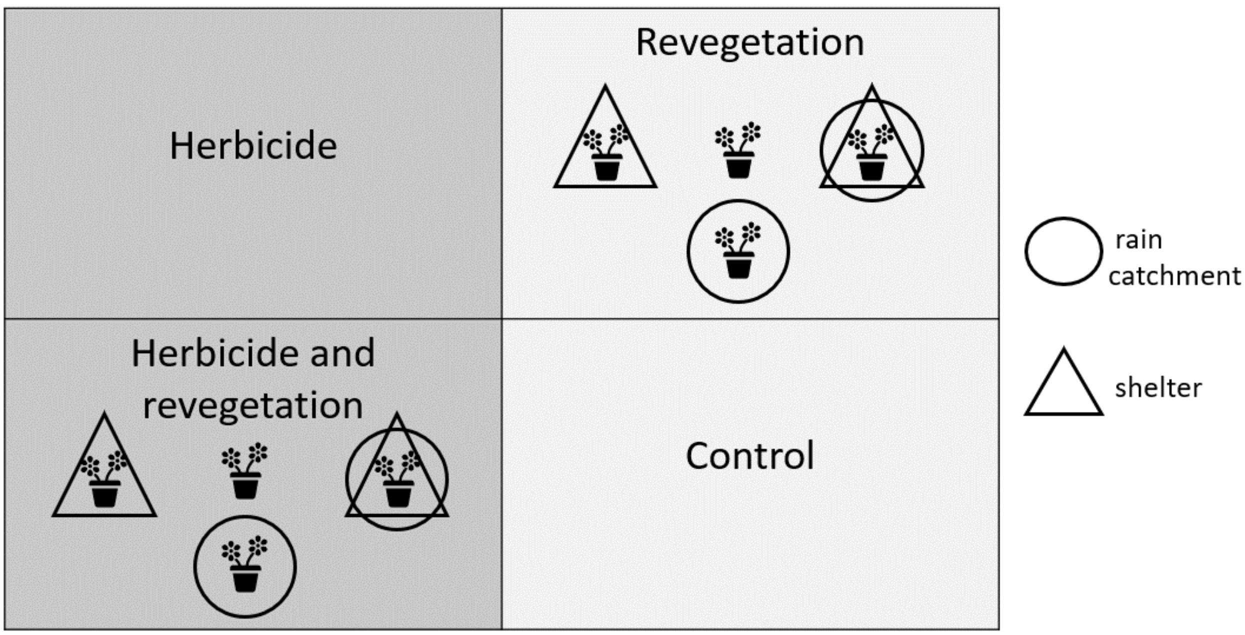

Figure 15. Plot design showing four plot types and transplant treatment combinations. Plot types: herbicide, revegetation, herbicide and revegetation, and control; and treatment combinations on transplants: rain catchment (circle), shelter (triangle), rain catchment + shelter (circle and triangle, and control.

RESULTS

Native and non-native plant cover

Prior to treatment in March 2017, non-native cover, mostly consisting of *Bromus tectorum* and *Bromus rubens,* did not differ among canyons ($F_{1,2} = 2.28$, $p = 0.158$), while native cover was significantly lower in Cottonwood Gulch ($F_{1,2} = 7.27$, $p = 0.013$) (Figure 16).

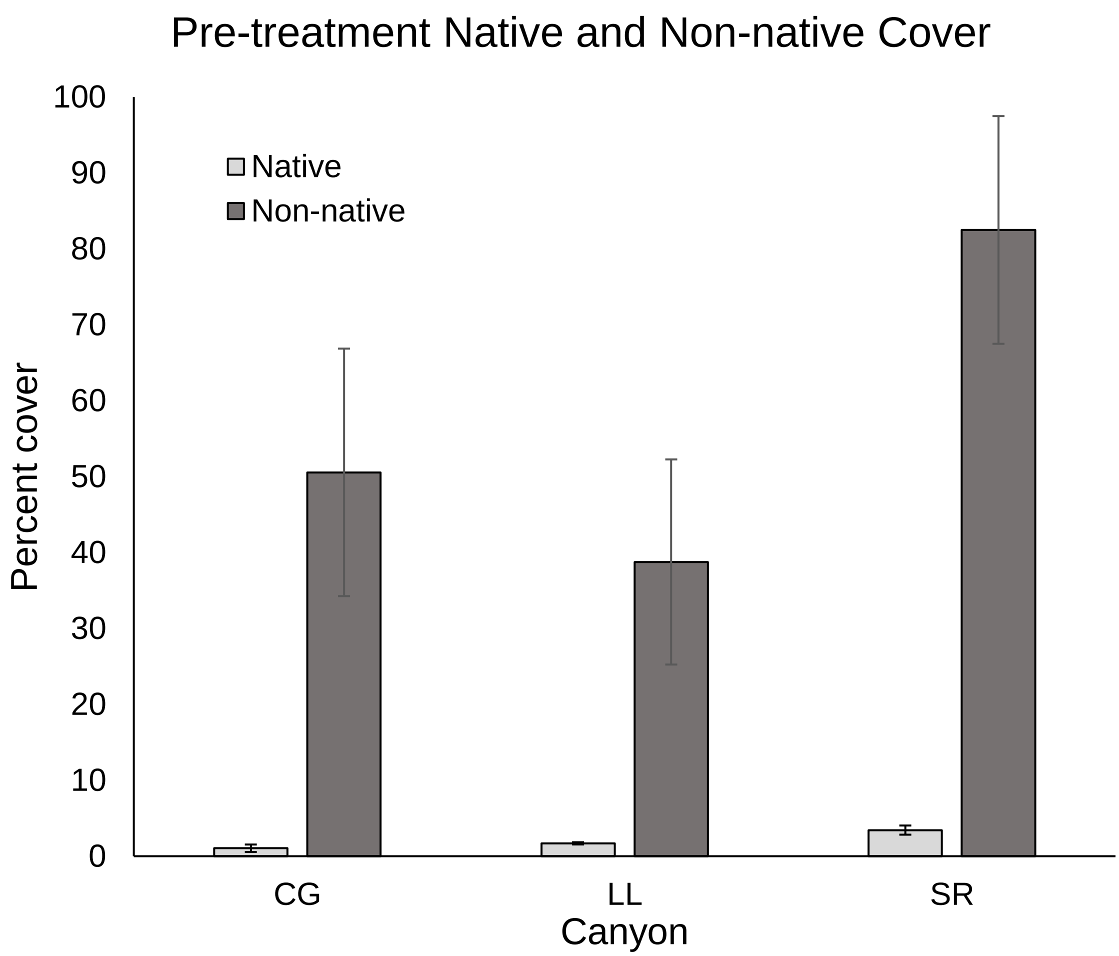

Figure 16. Mean pre-treatment native and non-native plant cover among canyons on brome plots. CG = Cottonwood Gulch, LL = Llewellyn Gulch, SR = Slick Rock Canyon. Cottonwood Gulch had significantly lower native cover than other canyons (p = 0.013). Error bars are ±1 SE.

Among all plots, post-treatment non-native cover did not differ among treatments. However, revegetation treatments significantly affected native plant cover compared to all other treatments in April 2018 ($F_{1,3} = 3.54$, $p = 0.068$, $\alpha = 0.10$) (Figure 17). No other plot treatments had significant effect on native plant cover.

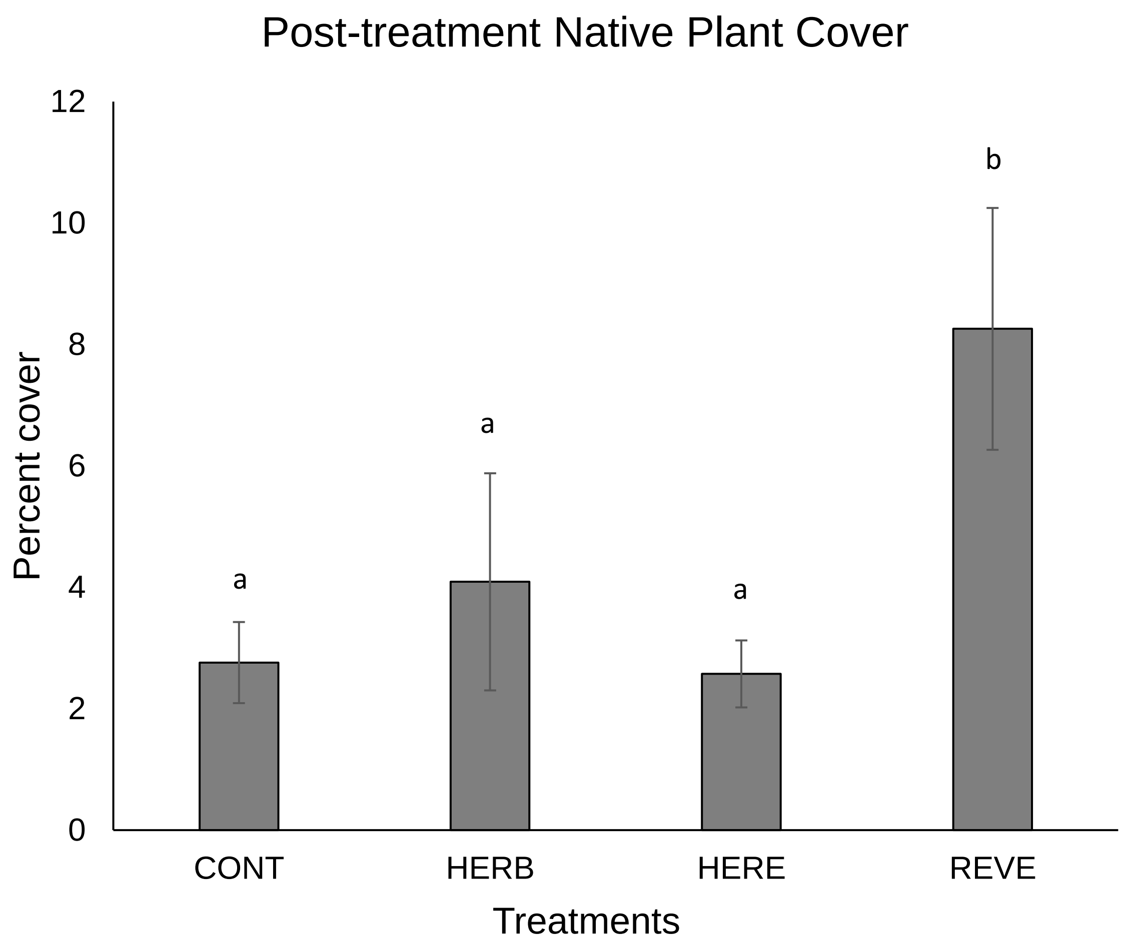

Figure 17. Mean post-treatment native cover among canyons on brome plots by treatment. Revegetation had significantly higher mean cover than other treatments ($p = 0.068$). Letters denote homogenous groups. CONT = control, HERB = herbicide, HERE = herbicide and revegetation, and REVE = revegetation. Error bars are ±1 SE.

Transplant survival

Among species, *Opuntia* and *Sporobolus cryptandrus* had the highest overall survival in April 2017 and August 2018, regardless of plot or treatments (Figure 18). In April 2018, *Opuntia* had a survival rate of 69%, while *Sporobolus* was about 38%.

Shelters and rain catchments did not significantly affect transplant survival ($p > 0.10$). However, there was a difference in survival over time: transplant survival declined between August 2017 and April 2018 ($p = 0.012$). In August 2017, approximately 50% of transplants had survived since initial planting, but an additional 11% decline in survival occurred between summer 2017 and spring 2018 (Figure 19) resulting in 39% overall survival by the end of the study.

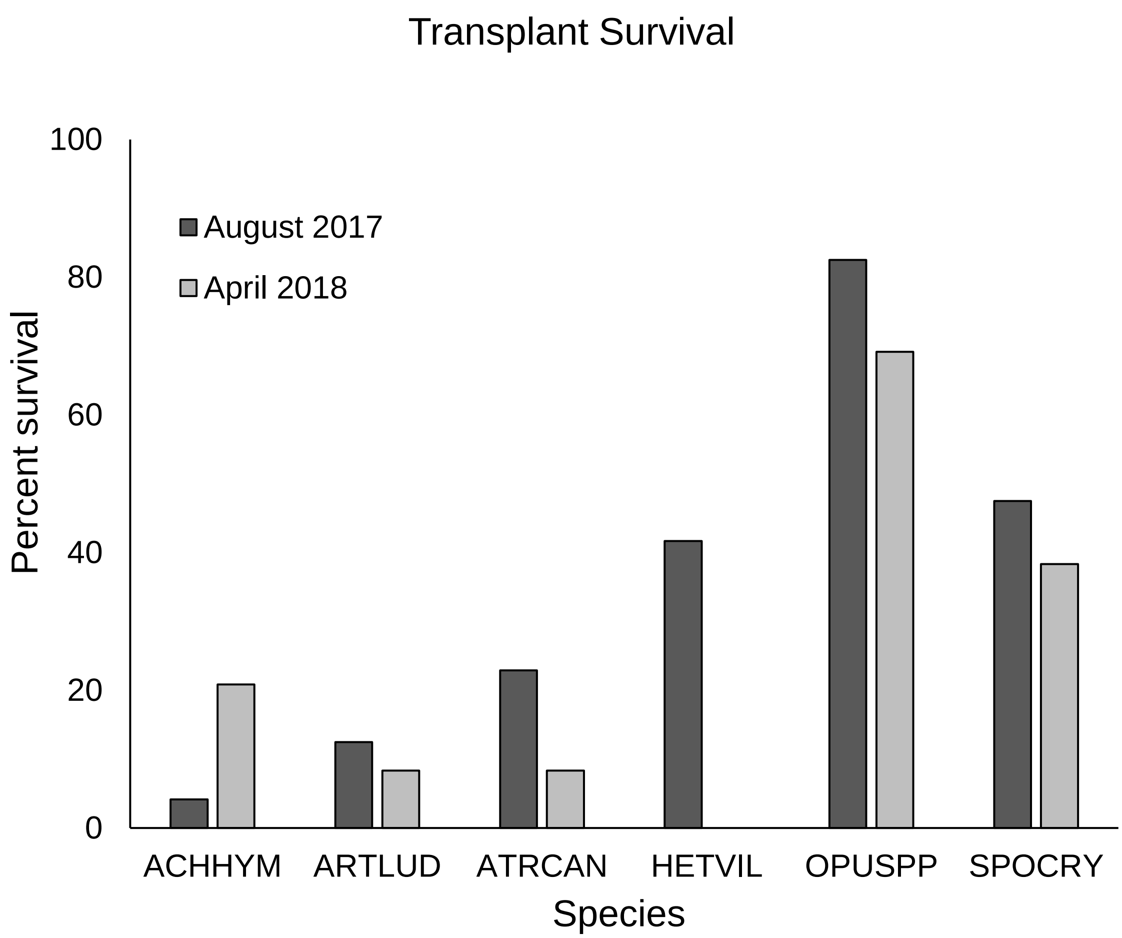

Figure 18. Total percent survival of transplant species used on brome plots. ACHHYM = *Achnatherum hymenoides*, ARTLUD = *Artemisia ludoviciana*, ATRCAN = *Atriplex canescens*, HETVIL = *Heterotheca villosa*, OPUSPP = *Opuntia spp*., SPOCRY = *Sporobolus cryptandrus.* OPUSPP and SPOCRY had highest overall survival in both years.

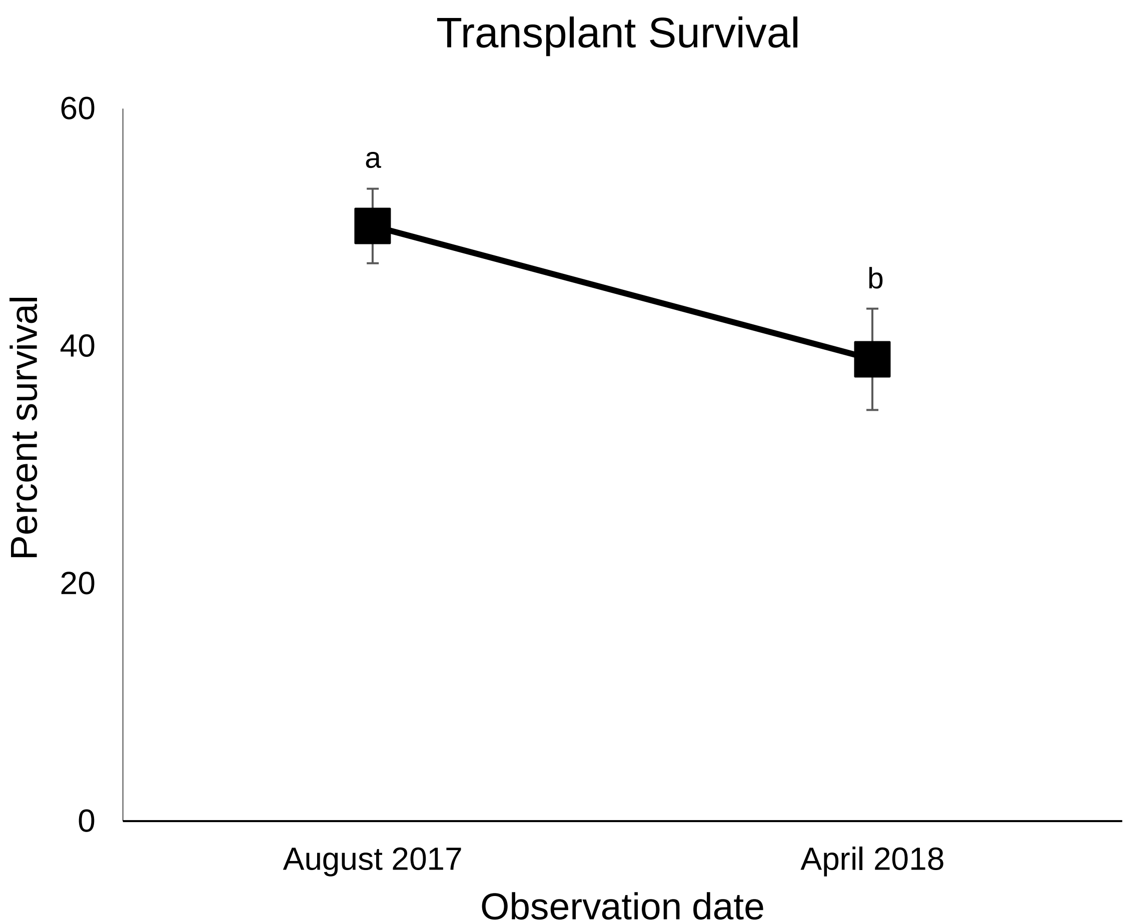

Figure 19. Transplant survival decline between years on brome plots. Survival significantly decreased ($p = 0.012$) from 50% in August 2017 to 39% in April 2018, an 11% difference. Error bars are ±1 SE.

Discussion

Effect of treatments on plant cover

The significant effect of revegetation without herbicide on native plant cover may indicate that this method of revegetation can be a successful way to increase native plant populations in the semi-arid landscape in GLCA. However, it was shown that herbicide application may not be useful in reducing non-native plant cover based on the treatments imposed. The use of a combination treatment of herbicide and revegetation on the plots did not significantly affect native plant cover. This difference between the revegetation and the combination of herbicide and revegetation may be due to possible unwanted herbicide contamination onto nearby native plants or soil during application. Imazapic herbicide has been shown to reduce non-target native forb populations up to an average of 84% when applied in fall for pre-emergence of *B. tectorum* (Baker et al., 2009; Morris et al., 2009). A separate study showed that native plants can recover after spring application post-emergence application (Barnes, 2007), suggesting that spring application may be a better approach.

Transplant survival

The use of rain catchments and shelters had no significant effect on the survival of the transplants during the duration of our study. This may be due to multiple factors: quality of materials used, lack of maintenance on structures, or the use of rain catchments during transplant establishment. The significant decline in survival in transplants over time may indicate that transplants need additional care and irrigation, as precipitation may not have been sufficient post-transplanting. Alternatively, the use of on-site materials may not always be efficient to protect establishing individuals.

The species chosen for transplantation may have played a part in the overall transplant survival. The high survival rate of *Opuntia* and *Sporobolus* species indicates that species selection is important to the overall success of revegetation efforts, as seen in other studies (Dreesen et al., 1998; Abella and Newton, 2009). *Opuntia* and *Sporobolus* are species of drought tolerance, as well as having cultural value to local Native American tribes. Such plants can be useful when revegetating with cultural goals in mind.

Conclusions

This study showed that one-time application of herbicide did not significantly affect the cover of exotic plant species, potentially indicating the need for additional application of herbicide (Morris et al., 2009) or perhaps other chemical treatment. The possible contamination of herbicide onto non-target native species indicates that future projects must take this initial native cover decline into account, although recovery of native plant cover may still be possible in the future (Barnes, 2007). While shelters and rain catchments did not significantly help survival in this project, other studies have shown their success (Grantz et al., 1998; Edwards et al., 2000), and precipitation may not have been sufficient for transplant survival in this study. Additional irrigation and use of prefabricated structures, such as plastic cones, may be of more value in enhancing survival (McAuliffe, 1986). The species chosen for revegetation played a major part in whether the transplants were able to persist in the semi-arid environment of GLCA, demonstrating how species selection can be a critical component in project design. Native American tribes and other land managers can successfully use the culturally important plants, *Opuntia* and *Sporobolus* species, for revegetation in areas infested with brome, without supplying additional irrigation.

CHAPTER 4: EFFECTS OF PERIODIC INUNDATION ON *SACCHARUM RAVENNAE* SEEDS

INTRODUCTION

Non-native invasive species are a prevalent problem in arid and semi-arid environments, yet continuing disturbances and changes in climate are creating challenges for land managers who seek to manage using restoration strategies. These riparian ecosystems historically experienced seasonal flooding and drought (Poff et al., 1997; Bendix and Hupp, 2000), but now many river systems have been developed for human use, thus creating controlled flow regimes and altered ecosystems (Williams and Wolman, 1984; Jansson et al., 2000; Tonkin et al., 2018). Dam operations that result in reservoirs can affect the vegetation communities of the riparian ecosystem (Beauchamp and Stromberg, 2008), and may lead to new plant invasions (Mortenson and Weisberg, 2010). Flooding conditions can be suitable for certain species' seeds to recover from dormancy (Coops and van der Velde, 1995), particularly if floods occur in spring (Insausti et al., 1995; Hölzel and Otte, 2004). Invasive grasses are known to be able to withstand flooding events, potentially resulting in higher annual exotic grass cover (Greet et al., 2013). The presence of invasive exotic grass species in riparian habitats that exhibit unnatural flow regimes from dams may indicate a level of flood tolerance in the seeds that enable the plant to establish in these conditions.

In the semi-arid deserts of the southwestern US, *Saccharum ravennae* is invading riparian corridors (Cal-IPC, 2015), and the plant has spread to the areas of Glen Canyon National Recreation Area (GLCA). The flow of five tributary rivers to Lake Powell and the managed flow from dam operation, result in seasonal fluctuations of the water level of Lake Powell. High water levels occur in mid-summer and low levels in winter (National Park Service, 2015). These fluctuations can result in months of completely inundated conditions in low elevation areas near the shore of the lake, with periods of exposure during low water levels. It is currently unknown if these flooding events throughout the year are affecting the germination and establishment of ravennagrass within GLCA. However, due to ravennagrass presence within seeps and springs in the park, its seeds may possess a high tolerance to long durations of water inundation, which may facilitate its spread.

As land managers become more aware of ravennagrass, understanding how its seeds may be able to survive amongst changing environments will give more insight into how to manage for this species. Anthropogenic influences, such as dams that create periodic flooding events, may influence how this invasive plant species is able to reproduce in these environments. A previous study has shown successful germination rates of ravennagrass seeds (greater than 80%) within 14 days in laboratory growth chamber experiments (Springer and Goldman, 2016). No research has been done, however, on the viability of ravennagrass seeds after periods of water submersion. The aim of this study was to assess the effect of complete water submersion on ravennagrass seed germination at varying lengths of time. The work was conducted in a laboratory setting but was intended to understand how ravennagrass may be surviving period of inundation from Lake Powell. This information can then be used to inform land managers and scientists about life history traits of ravennagrass, specifically on how its seeds may survive and persist in riparian environments.

METHODS

To assess the effects of water submersion on the germinability of ravennagrass seeds, samples were taken from mature seed heads in the field and transported to the lab. Treatment group of 30 seeds underwent complete submersion in lake water and a control group did not. Ravennagrass seeds and water samples from Lake Powell were taken from GLCA in December 2016. These seeds were transported in dry sacks along with lake water in a sealed container to the lab. The seeds were kept in dry storage at room temperature and the lake water was stored in a cold room at 4°C.

All seeds underwent a pre-treatment to help prevent fungus and mold growth that may inhibit germination during the experiment. This entailed using UV exposure on the seeds for two hours with seeds rotated every 30 minutes, followed by a 3% hydrogen peroxide soak for 30 minutes. Seeds were then stored until used for the experiment in a sealed, sterile container at room temperature to prevent contamination.

The treatment group consisted of 30 seeds that were placed on sterilized filter papers in 3.5 cm clear petri dishes. The dishes were completely filled with Lake Powell water and sealed with parafilm then placed inside specimen cups. The cups were filled with 5 mL of Lake Powell water, sealed closed, and placed within a diurnal germination chamber with a daytime temperature of 19°C and a nighttime temperature of 8°C. For the control group, the exact procedure for the treatment group was followed, excluding the addition of Lake Powell water, resulting in non-submersed seeds.

Treatments were repeated monthly for 15 months. After month 16, the cups were then removed from the germination chamber and the seeds taken out of the petri dishes and placed upon new, sterilized filter papers dampened with purified water. Any seeds that showed signs of germination or emergence were counted and taken out of the experiment. Contaminated seeds that were overtaken by fungus or mold were counted if germinated then removed. Remaining seeds were then sealed within the petri dish and placed back into the germination chamber. Germination was recorded every two to three days over four weeks. After week two, 500 ppm Gibbeleric acid was applied to the seeds to stimulate germination, and seeds were placed back into the germination chamber for an additional two weeks.

RESULTS

Using the mean percentages of germination for each treatment month, it was revealed that ravennagrass seeds were still germinable after being submerged in Lake Powell water up to 16 months, up to an average of 7.5% (Figure 20). Lowest germination was seen in seeds that were submerged the least amount of time. No differences were seen between the treatment and control groups.

Mean germination rates of the seeds spent in dry storage before study treatment, it was observed that seeds in storage for longer than 15 months had lower rates of germination than seeds that spent the least amount of time in storage (Figure 21).

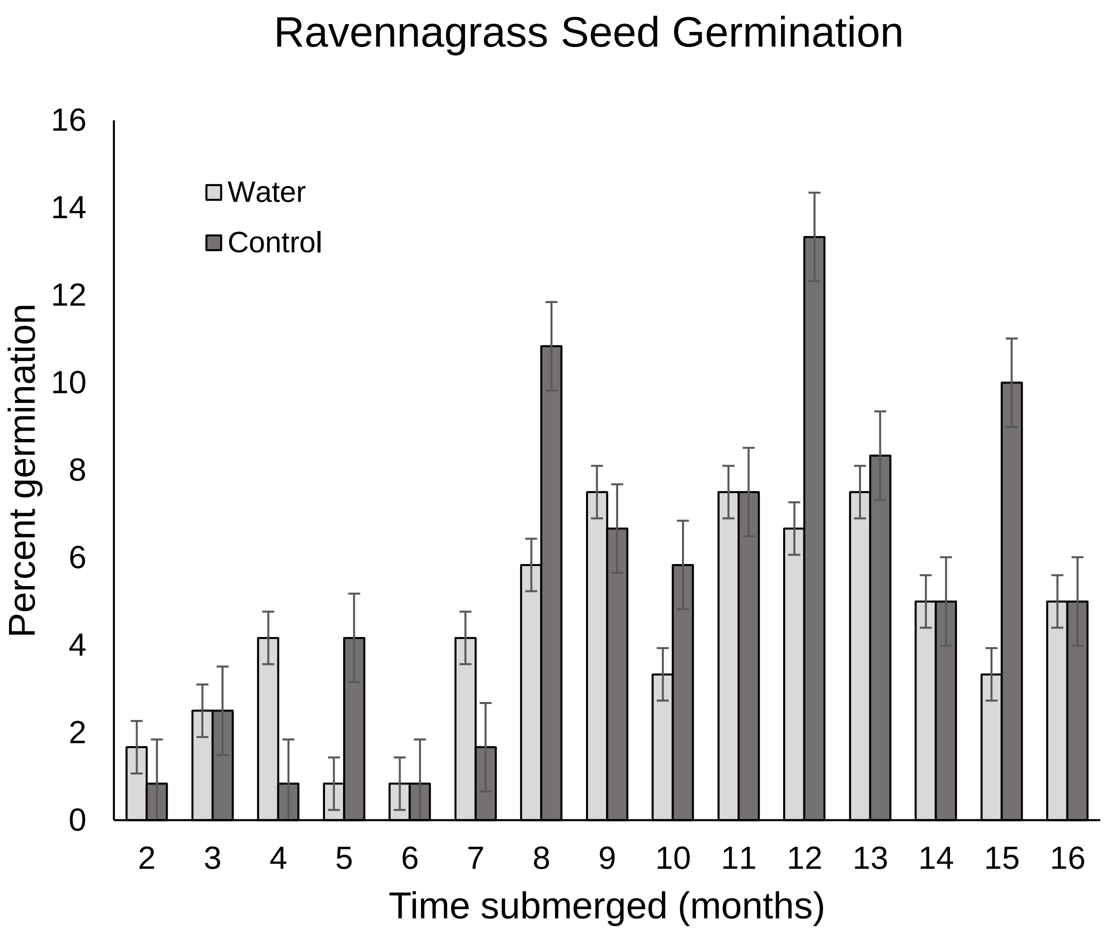

Figure 20. Percent germination of ravennagrass seeds over the time submerged. Controls were not submerged. Error bars ±1 SE.

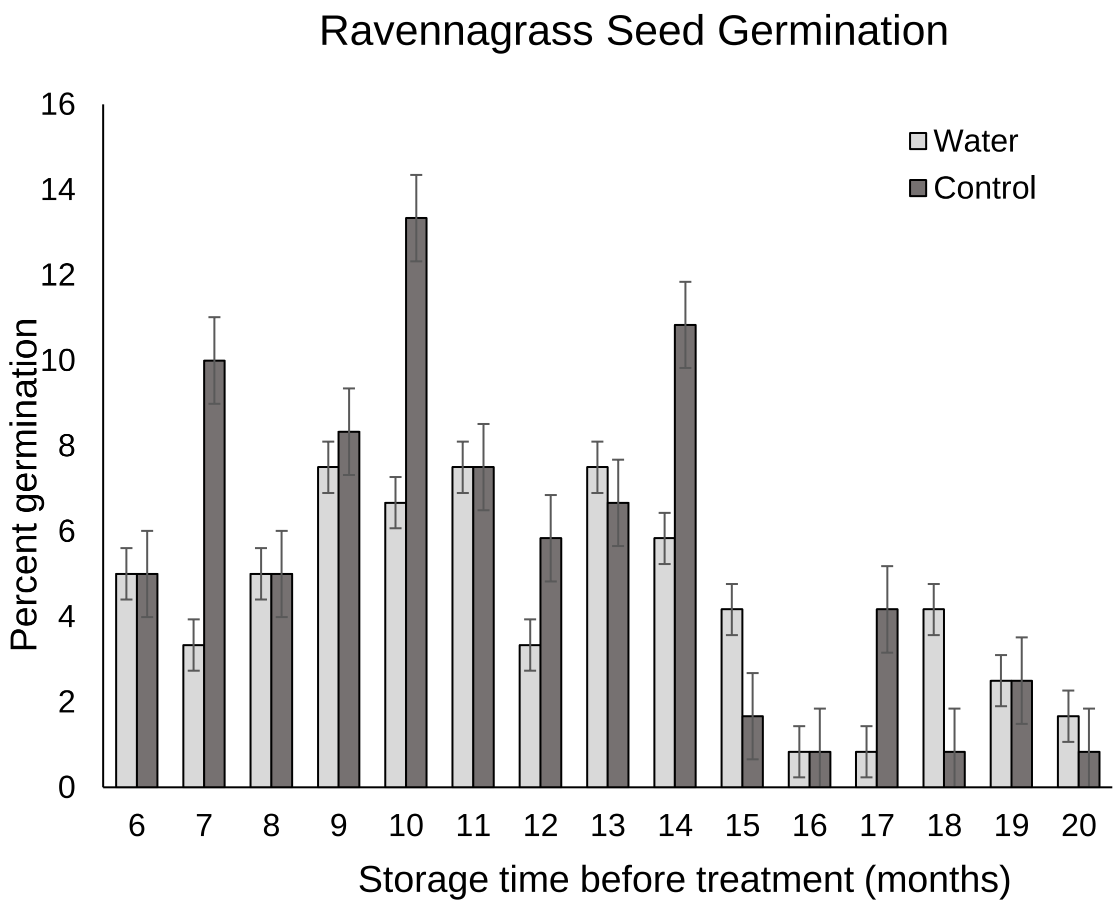

Figure 21. Percent germination of ravennagrass seeds over the months spent in storage before treatment. Error bars are ±1 SE.

DISCUSSION

Ravennagrass seeds were able to germinate after being submerged in Lake Powell water for up to 16 months. With no differences observed between the treatment and control groups, full submersion in water did not be appear to affect the rate of germination. However, treatments that were submerged for shorter amounts of time had lower germination rates that may be due to the amount of time the seeds spent in dry storage before being transferred to the germination chamber. Time spent in dry storage may negatively affect the germination of seeds, given that seeds with the shortest amount of time spent in dry storage had higher rates of germination, regardless of the submersion treatment.

The viability of the ravennagrass seeds after submersion indicates that seeds may be able to germinate and establish in areas of GLCA that undergo periodic flooding. This also could allow the seeds to travel and survive through water, potentially enabling ravennagrass to spread along neighboring riparian corridors. With this knowledge, care must be taken when dealing with ravennagrass in the field and when considering the effects of natural and controlled flooding on riparian vegetation communities. Seeds that are disturbed and released from seed heads into a riparian ecosystem are able to survive under submerged conditions for periods longer than previously known, making ravennagrass a resilient invader.

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CURRICULUM VITAE

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EDUCATION

2016 - Present	M.S. Biology, School of Life Science, University of Nevada, Las Vegas Master's Thesis: *Effects of Ecological Restoration Techniques in Glen Canyon National Recreation Area*
2013 - 2015	B.S., Biology, University of Utah, Salt Lake City, UT
2011-2013	A.S., Salt Lake Community College, Salt Lake City, UT

PRESENTATIONS AND PUBLICATIONS

2017	"*Integrating tribal values in collaborative restoration efforts.*" Invited guest speaker, EPA-Tribal Wetlands Workshop, Albuquerque, NM.
2017	"*iUTAH: Shaping the future one scientist at a time*." Invited guest speaker, iUTAH Annual Summer Symposium, Logan, UT.
2016	*Patterns in nitrogen stable isotope ratios in algae and macroinvertebrates along montane to urban stream gradients.* Poster presenter, Society for Freshwater Science Annual Meeting. Sacramento, CA.
2015	*Stable isotopes of algae and mosses reflect nitrogen sources along montane to urban stream gradients*. Poster presenter, Society for Freshwater Science Annual Meeting. Milwaukee, WI.

ACADEMIC RESEARCH EXPERIENCE

2018	Botanist – School of Life Sciences, University of Nevada, Las Vegas Performed vegetations surveys and compiled plant species lists in Gold Butte National Monument. Identified plants and created herbarium specimens. Surveyed endangered and rare plant species populations in gypsum habitat.
2015 - 2016	Laboratory Technician – Biology Department, University of Utah Prepared samples in the SIRFER isotope facility for isotope analysis on the elemental analyzer and mass spectrometer using various methods. Conducted an independent research project under supervision that assessed

the efficacy of macroinvertebrates and algae as indicators of urban stream pollution through Carbon and Nitrogen stable isotope analysis.

2015 - 2016 Hydrologic Technician Assistant – Biology Department, University of Utah

Assisted with water quality instrumentation in the laboratory and field in all weather conditions, including water sampling and sample management, velocity area stream gauging, water surface elevation surveying, and instrument maintenance and calibration. Routinely performed data quality control on large environmental datasets.

2014 - 2015 Undergraduate Research Assistant – Biology Department, University of Utah

Assisted in ecological research regarding water quality in Salt Lake Valley watersheds as part of the state-wide collaborative project, iUTAH. Collected and prepared water, plant, and soil samples for isotope analysis. Conducted a research project under supervision that assessed the suitability of Nitrogen stable isotopes in algae and moss in local streams as water quality indicators.

GRANTS AND AWARDS

2017	American Indian Graduate Center Scholarship
2011; 2012; 2013; 2017	Hualapai Department of Education and Training Scholarship
2014	Hualapai Department of Natural Resources Scholarship
2014	Freeport-McMoran Merit Scholarship
2013	University of Utah Merit Scholarship
2013	American Indian Services Scholarship
2013	Honors Graduate, Salt Lake Community College
2013	Pride in Academics Award

SERVICE

2017 - 2018	Volunteer speaker/water quality activity leader, UNLV American Indian Research and Education Center College Tour
2013 - 2015	Member, Inter-Tribal Student Association, University of Utah
2012 - 2013	President, American Indian Student Leadership Club, Salt Lake Community College

2011 - 2012 Treasurer, American Indian Student Leadership Club, Salt Lake Community College

ACADEMIC AND PROFESSIONAL MEMBERSHIP

2016 - present Society for Ecological Restoration
2015 - present Ecological Society of America
2015 - present American Indian Science and Engineering Society
2013 - present National Society of Collegiate Scholars
2015 - 2017 Society for Freshwater Science
2013 - 2015 American Chemical Society