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Restoration Strategies to Migrate Shrub Encroachment into Coastal Prairies

Erick Mark Verderber

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Restoration strategies to mitigate shrub encroachment into coastal prairies

by

Eric Mark Verderber

A Thesis Presented to the Graduate Faculty of the College of Science, Mathematics and Technology in Partial Fulfillment of the Requirements for the Degree of

Master of Science

in the Field of Biology

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ABSTRACT

Shrub encroachment into grasslands is a phenomenon facilitated by fire suppression, climate warming, and overgrazing of grasses by cattle, and if left unmanaged, can alter ecosystem structure and function. In this study, I compare the effectiveness and cost/benefit of various restoration strategies (prescribed fire, mechanical, and chemical) used singly and in different combinations at decreasing two shrubs, honey mesquite (*Prosopis glandulosa* Torr.) and huisache (*Acacia farnesiana* (L.) Willd.), and promoting growth and survival of gulf cordgrass (*Spartina spartinae* (Trin.) Merr. ex Hitchc.), the dominant coastal prairie grass within the Laguna Atascosa National Wildlife Refuge-Bahia Grande Unit in deep south Texas. The U.S. Fish and Wildlife Service (USFWS) currently manages this unit for the Federally–endangered aplomado falcon (*Falco femoralis septentrionalis*), a grassland-dependent species. This study is unique as it evaluates shrub reduction, shrub resprouting, and grass recovery in response to eight treatment regimes using up to three restoration strategies in various orders. These responses were assessed at 4-mo intervals over a 12-mo post-treatment period and then ranked using a dimensionless scale to create a ‘success index’ comparable across treatments. By 12-mo post-treatment, all treatments caused a similar reduction in original stem basal area and density. This response was immediate when treatments used roller-chopping, but more gradual following a single prescribed fire. Treatments where herbicide followed roller-chopping most effectively killed adult shrubs and limited resprouting. Gulf cordgrass recovery was slowest when roller-chopping was the only or last treatment, likely because roller-chopping created leaf litter and woody debris that hindered grass seed germination. Overall, index results suggest that roller-chopping followed by herbicide then fire was the most effective treatment for reducing woody shrubs while facilitating gulf cordgrass recovery, while a single prescribed fire was least effective. Though more costly than other treatments, roller-chopping followed by herbicide then fire may provide land managers with a restoration strategy for both removing woody shrubs and restoring gulf cordgrass cover in coastal prairie ecosystems.
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I. INTRODUCTION

Shrub encroachment, which is the increase in density, cover, and biomass of woody or shrubby plants into grassland ecosystems, is an increasingly common global phenomenon with numerous causes and consequences (Van Auken 2009). In Australia, displacement of grasslands by native shrubs is occurring at higher rates than thought to exist prior to European settlement (Eldridge et al. 2011; Noble 1997). In South Africa, shrub encroachment affects over 13 million hectares of grass savannas (Trollope et al. 1989). Currently, in the United States, shrub encroachment threatens over 300 million hectares of grasslands (Knapp et al. 2008; Pacala et al. 2001). Active fire suppression (Box and White 1969; Lehmann 1965; Allred and Mitchell 1955), overgrazing of palatable grasses and coincident fuel removal and spread of shrub seeds by cattle (Coetzee et al. 2008; Angassa and Obia 2007; Scholes and Archer 1997), land fragmentation via agriculture and land development (Eldridge et al. 2011; Grace 1998), and long-term climate change (D’Odorico et al. 2010) all may contribute to the spread of woody shrubs into grasslands.

Shrub encroachment poses a variety of ecological consequences. Dense shrub cover can provide habitat for many undesirable and exotic animal species, such as feral hogs (Sus scrofa) (Stevens 2010). High shrub density may cause habitat loss and increased hunting competition for many raptors that require open grasslands for hunting and breeding (Mutch et al. 2005). For example, Federally-endangered northern aplomado falcons (Falco femoralis septentrionalis) are succumbing to direct predation by great horned owls (Bubo virginianus), which commonly reside in areas of dense shrub cover (Perez, Zwank, and Smith 1996). Also, a shift from herbaceous grasses to woody shrubs increases carbon recalcitrance, storage, and allocation (from aboveground and belowground biomass) (Knapp et al. 2008). Greenhouse gases and tropospheric ozone reduction may increase due to stimulation of volatile organic carbon emissions from increased density and presence of woody shrubs (Archer 2010). Reduced grass cover and fibrous roots decrease soil stability, potentially increasing erosion from wind and runoff.
This evidence suggests that problems related to shrub encroachment may amplify over time if left unmanaged.

This study focuses on coastal prairies of south Texas, where shrub encroachment combined with land use changes, has reduced cover of this ecosystem. By 1937, over 2.4 million hectares (93%) of coastal prairies in Texas had been lost due to intensive agriculture, ranching, and urbanization (Ortego and Kalmbach 2010; Lehmann 1941). As the loss of coastal prairies continued through the remainder of the 20th century, less than one percent remains in original condition (Ortego and Kalmbach 2010; Smeins, Diamond, and Hanselka 1991). Today, substantially less than one tenth of a percent of coastal prairies remain in a relatively undisturbed condition; the remaining 99.9% has been converted to other land uses such as agriculture and urban development (Grace 1998; Smeins et al. 1991). These prairies provide critical habitat for numerous fauna, including the aplomado falcon (Mutch et al. 2005), and buffer inland areas from storm surges, floodwaters, and dust storms. Reductions in Texas’ coastal prairies, instigated by fragmentation and conversion to other land uses, have created smaller, non-contiguous prairie landscapes which are more susceptible to rapid shrub encroachment and consequently creating less desirable habitat for many wildlife species (Grace 1998).

Two woody shrub species — mesquite (*Prosopis glandulosa* Torr.) and huisache (*Acacia farnesiana* (L.) Willd.) — may infest unmanaged areas due to adaptations that promote their growth in low moisture conditions, high-saline soils (Felker 1981), and the ability to re-sprout when top-killed (Lyons and Rector 2009). Seedlings of both species have a tap root system that rapidly extends deeply into the soil soon after germination, assuring the seedling an adequate supply of water (Ansley, Huddle, and Kamp 1997). Within their second year of growth, mesquite and huisache plants form a meristematic bud zone just below the soil surface allowing for prolific resprouting if the tops are removed mechanically or by fire (Clayton, Lyons, and McGinty 2014). Many studies have shown that grass production can increase following control of mesquite (Bedunah and Sosebee 1984; Dahl et al. 1978). High densities of mesquite (>25% canopy cover) reduce available sunlight, thus suppressing grass growth and understory species diversity (Ansley et al. 1997).
Rapid encroachment of mesquite and huisache on unmanaged lands can displace populations of native grasses along the south Texas coast. Gulf cordgrass (*Spartina spartinae* (Trin.) Merr. ex Hitchc.) dominates saline soils along the Texas coast and grows in large dense clumps (Scifres, McAtee, and Drawe 1980). In addition to preventing soil erosion and stabilizing shorelines, gulf cordgrass clumps also provide good bird nesting habitat and cover for wetland margin species such as geese, sandhill cranes, and mottled ducks (USDA 2006). Additionally, aplomado falcons require open grassland habitat with low woody plant densities ranging from <1 shrub/ha (P. Juergens, The Peregrine Fund, personal communication; Perez et al. 1996) in south Texas to 73 shrubs/ha in northern Chihuahua, Mexico (Montoya 1995). Consequently, widespread shrub encroachment into grasslands is causing both a loss of habitat and decline in population of aplomado falcons (Cade, Jenny, and Walton 1991). The consequences of shrub encroachment on coastal prairies may negatively impact both coastal grasses and many bird species.

In general, fire suppression and overgrazing of palatable grasses (non-gulf cordgrass) are major causes of shrub encroachment in south Texas (Box and White 1969; Lehmann 1965; Allred and Mitchell 1955). Most grasses tolerate low to moderate grazing; however, intense grazing can negatively affect growth and reproduction caused from excessive biomass removal (Van Auken 2000; Gardener et al. 1998; Heitschmidt and Stuth 1996; Archer 1994; Louda et al. 1990; Belsky 1986; Harper 1977). Additionally, overgrazing of existing grasses by cattle removes much of the aboveground biomass that serves as carrying fuels necessary to sustain frequent, intense fires that can kill young woody shrubs (Eldridge et al. 2011; Roques, O’Connor, and Watkinson 2001; Oba et al. 2000; Scholes and Archer 1997; Hanselka et al. 1997). One potential result from the lack of fire in a grassland ecosystem is high leaf litter accumulation, which may hinder grass seed germination and seedling development (Heisler, Briggs, and Knapp 2003; Briggs and Knapp, 2001; Knapp and Seastedt 1998; Abrams, Knapp, and Hulbert 1986; Knapp and Seastedt, 1986; Bragg and Hulbert, 1976).

Controlling shrub expansion in prairies requires the use of strategies that target the woody species’ ability to regenerate following disturbances. Mesquite and huisache
are prolific sprouters following top-kill, and their ability to re-sprout increases with shrub size (Lyons and Rector 2009). Seedlings are considered the weakest growth stage and treatments applied to seedlings will cost less and prevent/reduce reestablishment and seed production (Meyer and Bovey 1982). Effective restoration strategies must inhibit the re-sprouting response from mesquite and huisache by depleting or killing belowground biomass (nutrient stores) and hinder seedling establishment (Bontrager, Scifres, and Drawe 1979).

The use of prescribed fire is a common treatment strategy; however, fire may not be an effective option for woody shrub mitigation unless used repeatedly (Wright, Bunting, and Neuenschwander 1976). Fire-only treatments often kill small, young woody shrubs interspersed in fine fuels, but not larger individuals within denser stands (Van Auken 2000; Glendening and Paulsen 1955). Though a single fire can reduce woody cover during the first year following a burn, shrub mortality is generally < 15% (Texas Natural Wildlife 2013). The belowground bud zone is the target for control of most woody shrubs, and unless the buds can be killed with repeated burning, woody shrubs will continue to re-sprout (Clayton et al. 2014). After shrub density is reduced to desired levels in a grassland following mechanical or chemical treatments, a three year burn rotation can be effective at controlling shrub re-growth as the lapse in time between burns allows for higher fuel loading and prevents shrubs from reaching maturity (Hanselka, Drawe, and Ruthven 2007). Repeated burning may not be effective on mature trees as the taller, wider stature of these trees may not allow fire to burn under or around the trunks as carrying fuels (i.e., grass) may be sparse from canopy shading. Repeated burning is unable to prevent re-sprouting, and in these instances, fire alone is unable to reduce existing shrub cover (Teague et al. 2001).

Another strategy is the removal of woody vegetation via mechanical means. Mechanical shrub management methods can be classified into two categories: 1) removal of only the aerial (aboveground) portions of the plant and 2) removal of the entire plant. Shredding (i.e. mowing or masticating) and roller-chopping (spike-toothed drum rolled over vegetation) are the two methods commonly used for simple top removal (Reuschel et al. 2003). Mechanical-only treatments effectively remove aboveground biomass, but
often produce only short-term results due to the rapid re-sprouting ability of many shrubs (Bontrager, Scifres, and Drawe 1979). For example, the effectiveness of shredding and roller-chopping treatments generally last less than five years for mesquite and huisache in coastal parries, and in some cases, huisache re-sprouts grow to nearly half their original height in five months (Bontreger et al. 1979; Mutz et al. 1978; Powell, Box, and Baker 1972). Though effective at removing standing biomass, mechanical treatments often fail to provide long-term lethal results.

Individual plant treatments (IPT) of herbicides to shrubs are often an effective option. Herbicide treatments are generally only effective on mesquite and huisache for 5-7 years; follow-up ‘maintenance’ herbicide treatments and/or prescribed burning may be required to prevent re-sprouting (Fulbright and Taylor 2001). Though effective at killing the bud zones on most woody shrubs (Meyer and Bovey 1980), the individual application of herbicides may demand higher associated costs than other treatments (i.e. labor, chemicals, licensing, and application equipment) making this an expensive and time-consuming option for large-area treatments. Also, chemical treatments do not remove dead or standing biomass, which is an important function in restoring an open-canopy ecosystem. Chemical-only strategies can increase shrub mortality, but higher overall costs combined with slower and less dramatic responses make chemical-only treatments impractical and expensive for high-density, large-scale applications (Teague et al. 2001). Furthermore, application of herbicides poses certain inherent risks and liability to not only the applicator, but to the surrounding environment due to wind drift and runoff.

In general, shrub reduction treatments used singularly are not as effective as when used in combination with other treatments. For instance, effective prescribed burning in south Texas is often limited by the amount/distribution of fine fuels (i.e., grasses) which are required to carry fire across a burn area (White and Hanselka 1991). Thus, prior to burning, mechanical, or herbicide shrub control treatments may be necessary to produce adequate amounts and distribution of fine fuels to sustain hotter, shrub-killing fires (Hanselka et al. 2007). In previous studies, greater shrub mortality in south Texas was observed by applying fire treatments after mechanical and/or herbicide treatments due to the increase in available fine fuels (Van Liew et al. 2012; Scifres and Hamilton 1993).
Reductions in canopy cover and greater shrub mortality were also observed after two consecutive burns (once in the fall and once in the winter, over two successive years) following initial mechanical and herbicide treatments (Box and White 1969). Although extensive research and information is available on fire following chemical or mechanical treatments, limited research has been conducted utilizing fire, mechanical, and chemical treatments applied in various combinations.

This study addresses the current knowledge gap and lack of research on the ability of treatments used singly and in multiple combinations (fire, mechanical, and chemical) to reduce shrub encroachment into coastal prairies of South Texas. Though many studies address these treatments in different vegetation types, little research has been conducted to test all three treatments on a coastal prairie ecosystem. The objectives for this research are to: 1) evaluate the effectiveness of each treatment, and 2) determine the cost-benefit of each treatment. My overarching hypothesis is that a prescribed fire regime, when used in conjunction with chemical and mechanical treatments, will reduce woody shrub density and re-growth, suppress encroachment of invasive plant species, and aid in the reestablishment of a grass-dominated ecosystem. This study is unique because it tests the effectiveness of combining restoration strategies, evaluates the order of implementation of the strategies, and tracks vegetation responses over time following each treatment, which allows a relative assessment of change due to the compounded effects of each treatment. Furthermore, a success index was created to rank the relative effectiveness of each treatment. Ultimately, results from this study will assist land managers and ecologists in choosing the most optimal treatment regime to both efficiently suppress woody shrub growth/spread and create a positive response in prairie grasses.
II. METHODS

Study Area

This study took place at the Laguna Atascosa National Wildlife Refuge-Bahia Grande Unit near Laguna Vista, TX, USA (16-km west of the Gulf of Mexico; 3-m elevation). Historically a shallow bay, Bahia Grande (Spanish for ‘big bay’) is a lowland area consisting of high-saline ephemeral basins and low-lying flats while upland areas consist of prairies and woodland habitats (USFWS 2003). Vegetation in upland areas consists primarily of gulf cordgrass interspersed with varying densities of mesquite and huisache. Salt-tolerant plants such as sea oxeye daisy (*Borrichia frutescens* (L.) DC.), Virginia glasswort (*Salicornia depressa* (Standl.)), turtleweed (*Batis maritima* (L.)), and shoregrass (*Monanthochloa littoralis* (Engelm)) are found in relatively low-lying areas.

Under private ownership, the Bahia Grande area supported cattle ranching operations from 1920’s to the 1980’s (Texas Sea Grant 2002). In 1999, the 8,806 ha Bahia Grande Unit was purchased by the USFWS (from private ownership) with goals of restoring and conserving surrounding lands for wildlife and recreational uses (USFWS 2003). In 2005, the 2,630 ha impounded wetland (landlocked from a ship channel constructed in the 1930’s) was re-flooded via newly dug channel, and today, these wetlands have been mostly restored providing habitat for many waterfowl and aquatic animal species (USFWS 2003). A major management goal set by USFWS for upland portions of this Unit include restoring habitat for aplomado falcons by removing dense stands of woody shrubs and increasing gulf cordgrass cover.

Mean annual rainfall for the area is estimated at 66-cm (50-yr average) and mean temperatures range from 16 °C (winter) to 29 °C (summer) (USFWS 2003). During the study period, total precipitation was 48 and 70 cm in 2013 and 2014, respectively, with peaks in September 2013 (15 cm) and 2014 (30 cm) (Figure 1). Maximum daily temperatures were greatest June–August (> 33 °C) in both study years (NOAA National Climatic Data Center, Station USW00012957, Port Isabel, Cameron County Airport, Texas). Soils consisted of Lomalta clay and Sejita silty clay loam (USDA-NRCS 2012) and generally, were not saline (0.14 ± 0.0 dS/m) within gulf cordgrass-shrubland portions.
of the study area (Alexander, unpublished data), although saline soils (> 2 dS/m) were measured within nearby low-lying areas.

**Experimental Design**

A completely-randomized block design was implemented to test the effectiveness of different treatments (prescribed fire, mechanical roller-chopping, and chemical herbicide, used singly and in combination). In February 2013, to achieve true treatment replication, I used satellite imagery and field reconnaissance to locate three blocks (~ 0.1 km²) of relatively homogeneous shrub cover (basal area of 5.7–7.9 cm²/m²) within three non-contiguous areas of Bahia Grande (Figure 2). Within each block, I delineated nine plots (40-m x 40-m) located at least 10-m apart. Plot corners were permanently marked with 2-m tall steel T-posts; GPS coordinates and digital photodocumentation of vegetation cover (to facilitate visual tracking of growth over time) were taken at each corner post (camera pointing towards the center of the plot). Within the study area, mesquite consisted of 11.4% of woody shrubs while huisache made up the remainder at 88.5%. Mean basal diameter for both species was 8.8 cm (mean: 9.7 cm for mesquite; 8.0 cm for huisache) while mean height was 3.5 m (mean: 3.0 m for mesquite; 4.0 m for huisache). Both mesquite and huisache were pooled together in this study as both share similarities in growth patterns and size.

Plots within each block were randomly assigned to one of eight treatments: mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), and fire followed by roller-chopping and herbicide (FRH). Each block also contained an untreated control plot, which was monitored prior to treatment and periodically throughout the study (Feb-Mar 2013, Feb-Mar 2014, Jun-Jul 2014, and Sep-Oct 2014) to determine background variations in response variables. All treatments listed above (except for fire-only) incorporated roller-chopping to immediately remove aboveground shrub biomass, as this is a major management goal set by the USFWS for restoring and conserving aplomado falcon habitat. A single prescribed fire was chosen as a treatment because historically, fire was a natural part of this grassland
ecosystem, and removal of this disturbance is believed to have promoted shrub encroachment; thus, I was interested in understanding whether simply restoring the previously removed disturbance would sufficiently alleviate the shrub encroachment problem. Treatments utilizing herbicide only or herbicide-first were not chosen for this study because USFWS avoids this approach as herbicide use on live adult shrubs is prohibitively expensive and impractical to apply in dense, overgrown vegetation (J. Moczygemba, USFWS, personal communication). In general, treatments and/or treatment combinations that did not meet USFWS management goals, or were deemed cost prohibitive, were excluded from this study.

USFWS personnel conducted all treatment applications. Roller-chopping was conducted in February-March 2013 on all plots where roller-chopping was assigned as the only or initial shrub removal strategy (R, RH, RFH, RHF, and RF) and in November 2013 on all plots where roller-chopping followed treatment with prescribed fire (FRH and FR). Roller-chopping was conducted by rolling a heavy-duty, spike-toothed, pasture renovator ‘roller-chopper’ (Marden Industries, Punta Gorda, FL) pulled by a D6 dozer (Caterpillar Inc., Peoria, IL) over the entire plot. Initially, the front blade of the dozer pushed over standing woody biomass; then, the weight of the roller-chopper (containing numerous 15-cm x 10-cm cutting spikes welded in a spiral pattern to two offset steel drums) forced the spikes to lacerate the woody biomass against the soil surface as it passed over. Under most circumstances, roller-chopping removed aerial portions of grasses it passed over, often leaving grass leaf litter on the ground, and depending on soil softness, sometimes uprooted grass bunches.

Prescribed fire treatments were initially scheduled for March 2013 but were delayed until fall 2013 due to unfavorable weather conditions and burn restrictions. On November 8, 2013, prescribed fires were ignited using both handheld and All-Terrain Vehicle (ATV)-mounted drip torches (3:1 diesel-gasoline mixture). For all blocks, the ignition sequences used to complete the burns started with a backfire (upwind) followed by flanking fires, and finally a head fire (downwind). Each block was burned independently, starting with block 1 and ending with block 3. The prescribed burn was conducted in accordance with USFWS policy and prescribed fire burn plan, which
included air temperatures between 0-37 °C, relative humidity of 30-50%, and wind speed and direction of 6-10 knots out of the northwest. Weather forecasts and on-site weather conditions were evaluated by NOAA-National Weather Service and determined by USFWS ground personnel to be within prescription prior to burning. On the day of burn, maximum air temperature was 23.9 °C; average wind speed was 7.4 knots from the NW; and less than 2 cm of precipitation fell within the two weeks prior to the burn (NOAA National Climatic Data Center, Station USW00012957, Port Isabel, Cameron County Airport, Texas; Figure 1). Throughout the burning process, on-site weather conditions were monitored hourly by USFWS ground personnel.

In July 2013 and February 2014, plots that required chemical treatments following roller-chopping and/or burning (RH, RFH, and FRH) received a 25% solution of triclopyr (Dow AgroSciences LLC., Indianapolis, IN) mixed with 75% basal oil, which was applied circumferentially around trunks/stems (between ground level and 25-cm) of shrubs exhibiting resprouting characteristics (i.e., only top-killed by treatment). The 4-month lag time following roller-chopping/burning allowed resprouts to be easily detected and likely increased their susceptibility to the herbicide as they had more foliage by this time. The 25% herbicide/75% basal oil mixture application rate was chosen based on recommendations made by the Texas AgriLife Research and Extension Service for treating mesquite (McGinty and Ueckert 2001) and huisache (Hanselka and Lyons 2000) stems of all sizes with smooth or rough bark. Other mixtures rates that contained lower herbicide percentages were available; however, many of these rates were only suggested for use on shrubs having smooth bark or shrubs < 3.8 cm in diameter. Though all resprouting shrubs fell within these lower size limits, larger original stems that were still alive (post-roller-chopping and/or post-fire) exceeded these limits thus prompting use of the higher percentage rate. Diesel fuel was recommended as an emulsifying agent; however, basal oil was substituted as USFWS policy prohibits use of diesel fuel as an emulsifying agent when applied on USFWS property.

**Burn Temperature**

Burn temperatures were estimated within each burn plot using pyrometers constructed from aluminum tags painted with six heat-sensitive Tempilac® paints (79 °C,
163 °C, 246 °C, 316 °C, 399 °C, and 510 °C) and wrapped in aluminum foil (melting point: 644 °C) (Rebeck et al. 2006). Each pyrometer was attached to metal pin flags at two points, ground level and 30-cm. Pyrometers were placed at six locations located 5-m apart along the center of three 10-m x 30-m subplots spaced 5-m apart within each plot for a total of 18 pyrometers per plot (Figure 2). To preserve the integrity of burned pyrometers against moisture and wind damage, collection of pyrometers should occur immediately following the burn; however, due to safety concerns of nighttime conditions, collection of burned pyrometers was delayed until the next morning (~12-hrs post burn).

Vegetation Response

Vegetation response to shrub removal techniques was assessed initially in Feb-Mar 2013 (pre-treatment) and following treatment at 4-month intervals over a 2-yr period. Sampling periods include: (1) Feb-Mar 2013, (2) Jun-Jul 2013, (3) Sep-Oct 2013, (4) Feb-Mar 2014, (5) Jun-Jul 2014, (6) Sep-Oct 2014, and (7) Feb-Mar 2015. The fire-only (F), fire followed by roller-chopping (FR), and fire followed by roller-chopping then herbicide (FRH) were initially sampled on Feb-Mar 2013, then again in Feb-Mar 2014 and at every sampling period thereafter; all other treatments were sampled during each sampling period (Table 1). Sampling period one was the initial ‘base-line’ period for which other sampling periods are compared to; sampling periods two and three represent samples taken post roller-chopping and post roller-chopping following by herbicide application; sampling periods four thru seven reflect results from all treatments 1-yr and over (Table 1). Vegetation characteristics were sampled within three subplots (10-m x 30-m; 300-m²) delineated within each plot (Figure 2). Using steel pin flags, the centerline of each subplot was marked longitudinally from 0-m thru 30-m at 5-m increments. A 2-m buffer was placed between subplots and a 2.5-m buffer was placed between subplots and plot perimeter.

To assess shrub mortality and resprouting, density and basal area of all live (presence of visible green foliage) shrubs of mesquite and huisache were quantified within each subplot. Live shrubs included original stems > 2 cm in diameter that remained alive after treatment and resprouts (new growth post-treatment from around/near original trunk). Basal diameter was measured using calipers just above the
point where the stem exited the soil. Each original stump still alive post-treatment and/or resprouting clump received a unique ID tag and pin flag to facilitate long-term tracking; any new sprouts detected during subsequent sampling periods were also tagged and recorded. On the 12-mo sampling date, I also visually estimated what percentage of dead shrubs still remained standing, as one of the primary goals of restoring aplomado falcon habitat is removing standing biomass.

To determine if treatments negatively impacted gulf cordgrass and/or created bare areas increasing susceptibility to erosion and invasive plant establishment, visual percentage estimates of ground cover vegetation (classified by functional type and species), bare areas, leaf litter (dead, non-woody plant material), and woody debris (dead woody stems > 1 cm in diameter) were taken within each subplot at the same locations described above for pyrometers using a 0.25-m² quadrat subdivided into 100 5x5-cm cells. Assessment of leaf litter and woody debris were quantified as burn temperature and grass recovery (via influences on the seedbed and seed germination) may be influenced by their presence.

**Cost-Benefit Analysis**

All quantities and expenses relating to treatment applications (i.e. herbicide, basal oil, etc.), total number of labor hours, and diesel fuel costs were tracked and figured by treatment applications (prescribed fire, roller-chopping, and herbicide application); costs were estimated on a per hectare basis. Costs figured for prescribed burning were estimated at $67.13/ha for 12 personnel per 80-ha per day (USFWS 2013 Prescribed Fire Burn Plan). Mechanical roller-chopping costs were estimated at $100.71/ha and calculated by adding total diesel fuel cost plus total labor cost: [diesel usage/ha (78.88 L/ha) x diesel cost/L ($0.99/L) = diesel cost/ha (78.09/ha)] + [hours/ha (3.12/ha) x minimum wage/hour ($7.25/hr.) = labor cost/ha ($22.62)]. Herbicide application costs were estimated at $238.66/ha and were calculated by adding total herbicide cost plus total labor cost: [herbicide mixture usage/ha (27.59 L/ha) x herbicide mixture cost/L ($7.83/L) = herbicide mixture cost/ha ($216.02/ha)] + [hours/ha (3.12/ha) x minimum wage/hour ($7.25/hr.) = labor cost/ha ($22.62)]. Total costs for multi-treatment regimes were calculated by adding the per-hectare cost of each treatment together: mechanical +
burning = $163.88/ha; mechanical + chemical = $339.37/ha; and mechanical + chemical + burning = $402.54/ha (Table 3). Diesel fuel costs were based on 2014 U.S. annual average costs of $0.99/L (U.S. Department of Energy) and minimum wage rates were based on 2014 standards at $7.25/hr. (U.S. Department of Labor).

**Data Analysis**

Treatment effects on original shrub density and basal area, sprout density, vegetation ground cover, and burn temperature at different time periods post-treatment were analyzed using a two-way analysis of variance (ANOVA) with shrub removal treatment, time period, and their interaction as fixed factors and block as a random factor using JMP statistical software (version 11.0). The treatment plot was considered the experimental unit (n = 3). However, pyrometer readings and on-ground observations indicated that the fire-only treatment plot within block 3 did not burn; therefore, this plot was excluded from analyses (n = 2 for fire-only treatments).

To account for variability and timing of treatment applications, sampled data were standardized (by treatment) to reflect post-treatment changes in 4-mo intervals: 0-month (pre-treatment), 4-month, 8-month, and 12-month (Table 1). Control plots were only assessed by sampling period (over time), as they were not sampled during every sampling period, and so changes over time in these plots were analyzed separately via a one-way ANOVA. Analyses of invasive species presence (post-treatment) were not conducted as preliminary results indicated < 1% was present in the study area. Annual rates of shrub mortality and grass recovery were also calculated based on percent changes in these variables over the 12-mo post-treatment sampling period; treatment effects on these changes were analyzed using a one-way ANOVA. All data were checked for normality using the Shapiro-Wilk test and homoscedascity using Levene’s test. Square-root transformations were used when needed to better meet these underlying assumptions and Tukey’s test (post hoc) were used to determine differences among time effects and treatment effects.

In order to compare the relative 12-mo post-treatment effectiveness of shrub removal methods tested in this study, I constructed a ‘success index’ based on several indicators that most describe whether these treatments met the primary management
objectives outlined by USFWS for restoring aplomado falcon habitat and which were addressed in this study: 1) reduction of shrub density within a range suitable for aplomado falcons, 2) low density of resprouting clumps, as these clumps ultimately become adult shrubs, 3) reduction of standing dead shrubs, and 4) recovery of gulf cordgrass percent cover to a level suitable for the continued use of fire as management tool to hinder shrub regrowth (Table 2). This index was constructed by assigning project indicators for each treatment plot to a categories ranging from 0 to 1 on a dimensionless scale and then averaging across indicators to determine a treatment rank, where values closer to 1 represent the most effective treatments and those closest to 0 are the least effective.

For each of the four indicators used in the index, plot values at the 12-mo post-treatment sampling interval were assigned to three categories based on several criteria. Density of original shrub stems still alive at the 12-mo post-treatment sampling period and density of resprouting clumps calculated by dividing the number of individual sprouts by 25 (based on an estimate of the average number of sprouts per clump) were assigned to the following categories based on documented shrub density ranges suitable for aplomado falcons: 1 (most desirable): 0 to 1 shrub/ha (based on densities specific to south Texas (Perez et al. (1996); P. Juergens (The Peregrine Fund, personal communication); 0.5 (moderately desirable): > 1 to 73 shrubs/ha (based on values reported outside of Texas but considered within preferred habitat limits (Truett 2002; Montoya 1995); 0 (undesirable): > 73 shrubs/ha, (based on values outside the range of recommended shrub densities for aplomado falcons across their habitats). Assuming standing dead shrubs can also provide habitat for predators of aplomado falcons, categories for estimated percent of dead shrubs remaining standing at the 12-mo post-treatment sampling period were assigned as follows: 1 (desired): ≤ 25%; 0.5: 25-74%; 0 (undesired): ≥ 75%. A potential long-term goal of USFWS is to use periodic prescribed burning as a management tool to maintain low shrub density, which requires relatively continuous cover of live gulf cordgrass as it is the dominant fuel in this system. Thus, gulf cordgrass cover at the 12-mo post-treatment sampling period was assigned a value of 1: > 75% (sufficient to carry fire (T. Herzberger, Fire Management Officer, USFWS,
personal communication)); 0.5: 50-74%; 0: < 50% cover. Treatment effects on index rank values were analyzed using a one-way ANOVA; a Fisher’s LSD post-hoc test was used to determine differences among treatments.
III. RESULTS

Shrub Basal Area and Density

All treatments significantly reduced shrub basal area per unit area (time, p < 0.0001), but there were treatment-level differences in the magnitude of this change (treatment, p = 0.0013) (Figure 3A; Table 4). In all treatments except for prescribed fire, basal area declined immediately after treatment (~15-fold) and remained relatively the same over the study (< 0.68 cm$^2$/m$^2$). In fire-treated plots, mortality became increasingly evident over time. Overall, the most pronounced treatment differences were between F and RH and F and RHF (p < 0.01 for both comparisons), as F was least effective at reducing basal area (91% mortality at 12-mo) while these other treatments were most effective (100% mortality at 12-mo) (Table 3). Notably, there was no significant difference in the amount of basal area reduction between treatments containing all three removal methods (p > 0.5 for all comparisons). When expressed relative to the amount of original stem basal area pre-treatment, there were no significant treatment effects on annual percent change in original stem basal area at the 12-mo post-treatment sampling period (p = 0.62; Table 3). Original stem basal area within control plots did not vary over the study period (p = 0.51; Table 5).

All treatments, except F, reduced the average size of an individual shrub (cm$^2$/individual) (time, p < 0.0001; treatment, p = 0.0003) (Figure 3B; Table 4). Individual size of shrubs in fire plots remained similar (p > 0.05 for all comparisons) across sampling periods; the lack of a reduction in average size in these plots, despite a reduction in basal area and density, suggests that a single fire targets smaller individuals. By 12-mo post-treatment, treatments without herbicide (R, F, RF, and FR) were least effective at reducing the size of individual shrubs while those with herbicide were most effective (Table 3). Individual shrub size within control plots remained consistent over the study period (p = 0.49; Table 5).

All treatments significantly reduced density of original shrub stems (time, p < 0.0001), but the degree of reduction varied by treatment (p < 0.0001) (Figure 3C; Table 4). Again, treatments without herbicide were least effective in reducing density, while
those with herbicide were most effective, with RH and RHF, causing a three to four times
greater reduction in shrub density compared to treatments without herbicide (p > 0.02 for
all comparisons). However, RH and RHF were no better at reducing shrub density than
FRH and RFH (p > 0.28 for all comparisons). Thus, treatments using all three shrub
removal techniques were equally effective at reducing shrub density. Again, relative to
the amount of original stem densities present prior to treatment, there were no significant
treatment effects on annual percent change in original stem density at the 12-mo post-
treatment sampling period (p = 0.68; Table 3). Shrub density within control plots did not
vary over the study period (p = 0.55; Table 5).

All treatments caused a prolific sprouting response (Figure 3D), but the degree of
sprouting varied by treatment (p < 0.0001) (Table 4). The greatest sprouting density (0.08
– 0.12 sprouts/m²) was seen in roller-chopped plots that did not receive herbicide (R, FR,
and RF) and lowest (< 0.05 sprouts/m²) in roller-chopped plots with herbicide (RH, RHF,
and RFH). Sprouting was relatively low in fire plots, likely because fire-caused mortality
of shrubs was lowest (i.e., sprouting is low without top-kill). Of the treatments using all
three methods, FRH was least effective at limiting resprouting, but only significantly
lower than RHF (p = 0.03). Sprout density was not assessed in control plots as sprouts
were not present in the absence of treatment.

Groundcover

Across all treatments except R and RH, bare area tended to be highest
immediately after treatment (4-mo post treatment) and then gradually decreased over the
post-treatment interval (time x treatment, p < 0.0001; Figure 4A; Table 4). The highest
bare area cover (4-mo post-treatment) was in F and FR plots (~ 70%). In R and RH plots,
bare area remained consistent (< 28%; p > 0.6 for all comparisons) over the study period.
By 12-mo, bare area cover within each treatment was statistically similar (p < 0.05 for all
comparisons) to pre-treatment cover. Bare area exposure within control plots remained
consistent over the study period (p = 0.05; Table 5).

Prior to treatment, leaf litter substrate was 63-87%, but then declined immediately
post-treatment in plots with fire as part of the treatment regime and remained unchanged
in plots without fire (time x treatment, p = 0.02; Figure 4B; Table 4). Among plots
experiencing a leaf litter reduction post-treatment, the magnitude of reduction was most extreme in plots where fire was used as the first treatment (4-29% cover at 12-mo) and least extreme in plots where roller-chopping was used first (44-50% cover at 12-mo). Leaf litter cover within control plots remained relatively constant during the study (p = 0.71; Table 5).

Woody debris cover tended to increase post-treatment, but the degree of this effect varied by treatment and post-interval time period (time x treatment, p = 0.02; Figure 4C; Table 4). For treatments that did not utilize fire (R and RH), woody debris tended to be highest 12-mo post treatment, while for other treatments, woody debris was greatest at the 4-mo interval. However, the only significant differences were that RF plots had about six times more woody debris cover at the 4-mo interval (35%) than pre-treatment (6%). Woody debris cover in control plots remained constant over the study period (p = 0.20; Table 5).

All treatments reduced gulf cordgrass cover, but the magnitude of this effect varied by treatment (time, p < 0.0001; treatment, p = 0.002) (Figure 4D; Table 4). Across all treatments, gulf cordgrass cover was highest pre-treatment, reduced to the lowest cover 4-mo post treatment, and then began to recover at 8 and 12-mo post treatment. In general, plots without fire in the treatment regime (R and RH) had the lowest gulf cordgrass cover (~30%), which was about 1.5 times lower than plots treated with RHF and FRH (~ 45%; p < 0.05 for all comparisons). Relative to the amount of gulf cordgrass present prior to treatment, there were no significant treatment effects on annual percent change in gulf cordgrass recovery at the 12-mo post-treatment sampling period (p = 0.24; Table 3). Gulf cordgrass cover within control plots also remained consistent over the study period (p = 0.05; Table 5).

**Burn Temperatures**

Burn temperatures were significantly lower in roller-chopped-first plots compared to burned-first plots (p = 0.0008) (Figure 5; Table 6). Treatments that burned first clearly burned hotter (max mean temp = 311 °C) than treatments that were roller-chopped first (max mean temp = 141 °C). Across all treatments, surface levels burned hotter (max mean = 240 °C) than 30-cm levels (max mean = 212 °C) (Figure 5).
Cost Analysis and Success Index

Per hectare costs and biological effects of each treatment differed amongst treatment combinations (Table 2). Treatment costs per hectare ranged from fire being least expensive ($63.18/ha) to combinations of all three treatments (fire, roller-chopping, and herbicide) being the most expensive ($402.54/ha). Based on the index, treatments significantly varied (p = 0.002) in their ability to reduce shrubs, prevent resprouting, and allow gulf cordgrass recovery. Treatments that used herbicide (RH, RHF, RFH, and FRH) were significantly different from treatments that did not use herbicide (R, F, RF, and FR) (p ≤ 0.03 for all comparisons) suggesting that herbicide is effective at reducing stem and sprout densities while promoting gulf cordgrass recovery. Across treatments that used herbicide, RHF yielded the highest score (0.75) but was more costly to apply ($402.54), suggesting that this treatment is best at reducing stem/sprout densities while promoting gulf cordgrass recovery. Compared to RHF, RH yielded the second highest score (0.71) and was least costly to apply when compared to all treatments that used herbicide ($339.97), and did not use fire. Across all treatments, F yielded the lowest score (0.25) suggesting that it was the least effective of all treatments at reducing stem and sprout density, standing dead trees, and increasing gulf cordgrass recovery; however, it was least costly to apply ($63.17).
IV. DISCUSSION

Reducing shrub basal area and density to a range preferred by aplomado falcons was key in this study. Treatments utilizing roller-chopping effectively reduced original stem basal area and density. Fire was also effective, but the results were more gradual over time since this treatment did not physically remove biomass immediately following treatment. Fire also tended to target smaller, younger shrubs as opposed to larger individuals. Field observations indicated that larger shrubs were directly and indirectly protected from fire. Research on fire effects on velvet mesquite (*Prosopis velutina* Woot.) concluded that a larger basal diameter (and corresponding thicker bark) provided added protection from direct flame contact (Uchytil 1990). In addition, a lack of contiguous fine-fuel cover (i.e., gulf cordgrass) underneath shrubs, as a result of high canopy cover, may have prevented high intensity fires from reaching and killing woody plants (Agee et al. 2000). Higher burn temperatures are desired as they have shown to decrease shrub density and potentially increase shrub mortality (Box and White 1969). Compared to all treatments, RH and RHF most effectively removed aboveground shrub biomass.

Assessment of post-treatment resprouting of shrubs was important throughout this study as resprouts, over time, may increase shrub density to unacceptable levels for aplomado falcons. Though all treatments caused resprouting, treatments using roller-chopping without herbicide instigated a higher sprouting response than treatments containing herbicide. Roller-chopping only ‘top-kills’ shrubs and substantial resprouting may occur because belowground meristems remain intact (Wilson, Webb, and Thomas 2001). A reduction in sprout density was highest in treatments that applied herbicide directly following roller-chopping (RHF, RH, and FRH). When fire occurred between roller-chopping and herbicide applications (RFH), resprout density remained high. This may have been attributed to the charring and defoliating effects of fire on sprouts causing them to appear dead when, in actuality, they were still alive. For this reason, many sprouts may not have been detected or sprayed, as herbicide is only applied to visibly live individuals. Shrub mortality from herbicide appeared to be faster in sprouts than in original stems. Herbicide absorption by shrubs is typically greatest on young, smooth-
stemmed growth opposed to thicker, corkier bark of mature trees (Lyons and Rector 2009). Though field observations indicated that herbicide treatments often killed vegetation adjacent to treated shrubs, mortality of non-shrub vegetation was minimal as spot-treatment application of herbicide targets mainly shrubs.

The open canopy and low woody shrub density of gulf cordgrass prairies creates ideal hunting and breeding habitat for reintroduced aplomado falcon breeding pairs (Brown et al. 2006), as low shrub densities minimizing predation by other raptors that may otherwise roost in dense shrub stands (Perez et al. 1996). Despite substantial reductions in gulf cordgrass cover directly following treatments, all treatments showed a trend towards recovery during the study period. However, recovery rates were often lower in the absence of fire, suggesting that increased leaf litter and woody debris may inhibit seed germination and seedling establishment of gulf cordgrass, a phenomenon common in many plant communities (Cavieres et al. 2007). In addition, leaf litter and woody debris created from roller-chopping may increase fuel moisture, thus reducing overall burn temperatures. Hotter fires are desired as they may be more effective at reducing shrub densities and removing grass biomass, which may stimulate higher grass recovery. Fast recovery rates of gulf cordgrass are important because high grass cover provides fire-sustaining biomass for future burns and habitat for prey of aplomado falcons (Keddy-Hector 1998). Through field observations, herbicide use did not appear to negatively impact gulf cordgrass cover or increase size of bare areas.

Intrusion by invasive species into disturbed areas created by roller-chopping and fire treatments were concerns throughout the study. Research conducted by the USFWS found that establishment of invasive species can occur following prescribed burning in grasslands (Brooks and Lusk 2009). Another study in Florida determined that invasive plant germination can occur following roller-chopping disturbances in coastal shrub habitat (Reuschel et al. 2003). Several invasive grasses and/or grasslike species do occur within or near the Bahia Grande Unit and are often found near disturbed areas such as roadways and abandoned railroad beds. These species include Bermudagrass (*Cynodon dactylon* (L.) Pers.), bafflegrass (*Pennisetum ciliare* (L.)), guineagrass (*Urochloa maxima* (Jacq.)), Johnson grass (*Sorghum halepense* (L.) Pers.), King Ranch bluestem
Bothrichloa ischaemum var. songarica (Rupr. ex Fisch. & C. A. Mey.), and nutgrass (Cyperus rotundus (L.)). However, at my study sites, < 1% of Bermudagrass and nutgrass (combined) actually established following treatment. Soil salinity may contribute to a lack of invasive species establishment. Although initial measurements taken across the study area indicate that soils were not saline, bare areas created by fire disturbances may have high evaporation rates (Ohlenbusch and Hartnett 2000), which could increase salinity at the soil surface. Soil salinity was higher in low-lying areas, so salinity could be a limiting factor for invasive species establishment in those areas.

Post-treatment soil erosion was another factor of concern because disturbances from treatments create bare areas. Though the effects of erosion were not tested in my study, roller-chopped plots notably contained high leaf litter cover that may have discouraged soil erosion. In bare areas created from burning, promotion of soil erosion did not appear to occur as field observations indicate that a crust formed over the soil surface (a reaction from intense heat and clay-like properties of the soil), which may have temporarily sealed the soil surfaces, decreasing chances for erosion. Herbicide usage did not appear to increase erosion as shrubs were only spot-treated, which minimized bare areas.

Fuel moisture and bulk density may impact the ability of fire to kill shrubs in coastal prairies. Results of my study indicate that fire-first treatments may sustain higher burn temperatures. Despite similar pre-treatment gulf cordgrass cover (primary carrying fuel), burn temperatures from plots that were burned first exhibited higher burn temperatures than those that were roller-chopped first. The combustible nature of aerial portions of gulf cordgrass may contribute to higher temperatures in plots where fire occurred first. In plots that received roller-chopping treatments first, increased bulk density of leaf litter (and potentially higher fuel moisture) may have lowered burn temperatures, though not measured. In Southern pine forests, for example, leaf litter/duff moisture > 30% does not support active burning (Wade 2013). Field observations also suggest that the compacted nature of roller-chopped debris may decrease air space, thus slowing preheating of fuels prior to ignition and potentially lowering overall burn temperatures and fire sustainability (Anderson 1982). Historical data also indicate that
fires occurred regularly (1-3 years) and naturally across Texas’ coastal prairies and were set primarily by lightning and settlers (Stambaugh et al. 2014; Grace et al. 2005). Active suppression of naturally-occurring fires can be problematic for coastal prairies because without fires, shrubs reach larger sizes. Larger shrubs may, in turn, be more resistant to fires because thicker bark, concomitant with increased size, may protect shrubs from contact with direct flame. Also, as shrubs increase in size and density, heavy canopy cover may inhibit grass growth, thus lowering fuel loads around the trunk and protecting shrubs from fire (Hanselka et al. 2007).

Regimes using all three treatments (fire, roller-chopping, and herbicide) may produce the best results for mitigating shrub encroachment into coastal prairies. A study in Florida suggests that mechanical or chemical treatments should be used as pre-treatments to prescribed burning, citing that both treatments can effectively reduce shrub biomass and increase fuel loads (Long, Godwin, and Kobizar 2014). Similarly, fire, mechanical, and chemical treatments are typically most effective at mitigating shrub encroachment when used together (Van Liew 2012; Hanselka et al. 2007). The combined effects of roller-chopping followed by herbicide (RH) produced results comparable to RHF. However, the addition of fire (RHF) produced better grass recovery rates that in turn ranked this treatment regime higher in the success index (Table 2).

Weather conditions prior to burning may have influenced burn temperatures in this study. Low rainfall and high temperatures during August 2013 followed by high rainfall and slightly lower air temperatures in September 2013 (Figure 1.) may have increased grass cover that can influence the rate and spread of fire. Consequently, increased fuel moisture content (water uptake and retention by vegetation) may decrease overall burn temperatures as preheating of fuels prior to combustion may take longer. Also, burning in the spring or summer, prior to September 2013, may have produced different burn results as slightly higher air temperatures and lower precipitation may favor hotter fires. Fire can be detrimental to shrubs during summer months as most vegetation is water-stressed from low precipitation (White and Hanselka 1991). In contrast, winter burning might not have produced effective shrub mortality. A study in
south Texas ranchlands found that fall burning, compared to winter burning, increased both shrub mortality and grass abundance (Hanselka et al. 2007).

The results of this study support my overarching hypothesis stating that fire, when used with roller-chopping and herbicide treatments, effectively reduces woody shrub density and minimizes invasive species establishment while promoting a positive recovery in gulf cordgrass cover. The use of all three treatments also supports USFWS management goals aimed at restoring aplomado falcon habitat. The use of RHF may provide the best initial option to achieve these goals, although RH appears to be a cheaper and similarly effective option.
V. MANAGEMENT IMPLICATIONS AND CONCLUSION

This study was implemented using available federal resources, equipment, and manpower. Treatment costs calculated in this study were based on actual costs incurred by the federal government and do not necessarily reflect estimates deemed practical for non-federal land managers. For instance, prescribed burning on federal lands is highly regulated to provide not only greater safety for civilians, firefighters and wildlife, but to minimize liability. Costs for prescribed landscape burning of federal lands requires a minimum of 12 qualified personnel, with associated support equipment (i.e. fire engines, water tenders, etc.), to be on location and present throughout the duration of the burn. Replicating similar burns in the non-federal sector may cost substantially less as use of qualified manpower and equipment may not be required, readily available, or cost prohibitive. Similarly, costs to apply roller-chopping and/or herbicide treatments may fluctuate in the non-federal sector as fees for contractual services, equipment rentals, fuel, and bulk purchasing price differences will vary by geographic region.

Results of this study suggest that regimes incorporating all three treatments may likely reduce stem and sprout densities the greatest while promoting positive gulf cordgrass grass recovery. Though initial cost per hectare is high, I recommend utilizing a RHF treatment regime to initially remove standing biomass and increase gulf cordgrass cover. If burning is not practical, RH may produce comparable results with lower costs than RHF (Table 2). After shrub reduction has reached a desired level, I recommend a follow-up maintenance treatment (herbicide and/or fire) to keep shrub densities in-check. If feasible, applying herbicides to resprouting shrubs may provide an effective and practical approach for some land managers. Though fire treatment effects for more than one year were not assessed in my study, previous research suggests that a three-year burning regime (Frost 1998) may be effective at killing smaller resprouting shrubs and beneficial to prairie grasses (Axelrod 1985) in addition to mimicking the natural fire return interval for the region (Strambaugh et al 2014). If fire is chosen as a maintenance option, land managers must continually monitor and determine if fuel loading and continuity are high enough to sustain an effective fire (Fernandes and Botelho 2003).
Other considerations for the use of fire may include changes in seasonality, as this may influence burn temperatures due to variations in fuel density and moisture. Though fuel density/loading and differences in seasonal burning were not assessed in my study, other research has found that post-burn shrub mortality rates may differ substantially by season (wet vs. dry; summer vs. winter) and by species (Ruthven et al. 2003). Also, land managers must be aware that limitations to burning may include local governmental restrictions (i.e., proximity to homes), biological restrictions (i.e., bird nesting season), and weather, which must be taken into consideration prior to burning.

In summary, effective shrub mitigation practices must be implemented to prevent grassland to woody shrub dominated regime shifts, which pose many negative ecological consequences, especially for habitat of grassland-dependent species like the aplomado falcon. Though many treatment strategies that remove standing biomass exist, none tested in this study completely prevented resprouting of woody shrubs. Treatment approaches combining all three strategies (fire, mechanical, and herbicide) were generally most effective at removing existing shrubs and suppressing their spread and reestablishment, although rollerchopping followed by herbicide (RH) appeared to provide an effective alternative at a lower cost. My research concludes that roller-chopping followed by herbicide then fire (RHF), combined with a recommended follow-up herbicide and/or fire maintenance regime, may effectively transition woody shrub infested coastal areas back to functional prairie ecosystems.
VI. REFERENCES


Texas Sea Grant College Program. (2002). *A Second Chance*. Texas Shores Magazine. College Station, TX: Texas A&M University. 35:2, 6-8.


VII. FIGURES

**Figure 1.** Daily and monthly climatic data (Jan-2013 to Jan-2015) during a coastal prairie restoration project located the U.S. Fish and Wildlife Service-Bahia Grade Unit near Laguna Vista, TX. Data were obtained from the National Oceanographic Atmospheric Administration (NOAA) National Climatic Data Center’s Port Isabel/Cameron County weather station (USW00012957; 26.16583N, -97.34583W), located about 6 km southwest of the study area. Values are means ± (SE).
Figure 2. Experimental design for testing eight treatments regimes [mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), fire followed by roller-chopping and herbicide (FRH), and controls (C)] for a coastal prairie restoration project located at the U.S. Fish and Wildlife Service-Bahia Grade Unit near Laguna Vista, TX.
Figure 3. Pre-treatment (pre-trt) and post-treatment (post-trt) changes in original stem basal area (cm²/m²) (A), original stem basal area (cm²/indiv) (B), original stem density (indiv/m²) (C), and sprout density (indiv/m²) (D) across eight treatments regimes [mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), and fire followed by roller-chopping and herbicide (FRH)] over a 12-mo post-treatment period at the U.S. Fish and Wildlife Service-Bahia Grade Unit near Laguna Vista, TX. Values are means ± (SE).
Figure 4. Pre-treatment (pre-trt) and post-treatment (post-trt) changes in percent cover of bare area (A), leaf litter (B), woody debris (C), and gulf cordgrass (*Spartina spartinae* (Trin.) Merr. ex Hitchc.) cover (D) across eight treatments regimes [mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), and fire followed by roller-chopping and herbicide (FRH)] over a 12-mo post-treatment period at the U.S. Fish and Wildlife Service-Bahia Grade Unit near Laguna Vista, TX. Values are means ± (SE).
Figure 5. Mean burn temperatures for treatment regimes utilizing fire (F), fire followed by roller-chopping (FR), fire followed by roller-chopping then herbicide (FRH), roller-chopping followed by fire (RF), roller-chopping followed by fire then herbicide (RFH), and roller-chopping followed by herbicide then fire (RHF) at surface level and 30-cm height for a coastal prairie research project located at the U.S. Fish and Wildlife Service-Bahia Grande Unit near Laguna Vista, TX. Values are means ± (SE).
Table 1. Sampling periods, treatment times, and standardized time intervals used for coastal prairie restoration for eight shrub removal treatment regimes [mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), and fire followed by roller-chopping and herbicide (FRH)] located at the U.S. Fish and Wildlife Service-Bahia Grande Unit near Laguna Vista, TX.

<table>
<thead>
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<tbody>
<tr>
<td>R</td>
<td>0-mo, R</td>
<td>4-mo</td>
<td>8-mo</td>
<td>12-mo</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>RH</td>
<td>0-mo, R</td>
<td>H</td>
<td>4-mo</td>
<td>8-mo</td>
<td>12-mo</td>
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<td>0-mo, R</td>
<td>H</td>
<td>F</td>
<td>4-mo</td>
<td>8-mo</td>
<td>12-mo</td>
</tr>
<tr>
<td>RF</td>
<td>0-mo, R</td>
<td>--</td>
<td>F</td>
<td>4-mo</td>
<td>8-mo</td>
<td>12-mo</td>
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<td>RFH</td>
<td>0-mo</td>
<td>0-mo</td>
<td>F</td>
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<td>4-mo</td>
<td>8-mo</td>
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<td>F</td>
<td>0-mo</td>
<td>--</td>
<td>F</td>
<td>4-mo</td>
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<td>--</td>
<td>F, R</td>
<td>4-mo</td>
<td>8-mo</td>
<td>12-mo</td>
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<tr>
<td>FRH</td>
<td>0-mo</td>
<td>--</td>
<td>F, R</td>
<td>H</td>
<td>4-mo</td>
<td>8-mo</td>
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**Table 2.** Index rankings for original stem density, sprout clump density, percent standing dead trees, gulf cordgrass (*Spartina spartinae* (Trin.) Merr. ex Hitchc.) cover, overall index, and cost per treatment comparison of eight shrub removal treatment regimes [mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), and fire followed by roller-chopping and herbicide (FRH)] used for a coastal prairie restoration project located at the U.S. Fish and Wildlife Service-Bahia Grande Unit near Laguna Vista, TX. Values are means ± (SE).

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<th>Rank (0-1)</th>
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<th>RH</th>
<th>RFH</th>
<th>FRH</th>
<th>RF</th>
<th>R</th>
<th>FR</th>
<th>F</th>
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<td>Stem Density</td>
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<td>0.83</td>
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<td>Sprout Clump Density</td>
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<td>0.67</td>
<td>0.50</td>
<td>0.67</td>
<td>0.50</td>
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<td>0.50</td>
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<td>Standing Dead</td>
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<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Gulf cordgrass</td>
<td></td>
<td>0.33</td>
<td>0.17</td>
<td>0.33</td>
<td>0.29</td>
<td>0.17</td>
<td>0.00</td>
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<td>0.58ab</td>
<td>0.50b</td>
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<td>0.25c</td>
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<td>Cost per Treatment</td>
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<td>$339.37</td>
<td>$402.54</td>
<td>$402.54</td>
<td>$163.88</td>
<td>$100.71</td>
<td>$163.88</td>
<td>$63.17</td>
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Table 3. Rates of annual change (± % change) of original stem basal area/density and gulf cordgrass (*Spartina spartinae* (Trin.) Merr. ex Hitchc.) cover between pre-treatment and 12-mo post-treatment intervals (mean ± (SE)) across eight shrub removal treatment regimes [mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), and fire followed by roller-chopping and herbicide (FRH)] used for a coastal prairie restoration project located at the U.S. Fish and Wildlife Service-Bahia Grande Unit near Laguna Vista, TX. Values are means ± (SE).

<table>
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<th>Response Variable</th>
<th>Treatment</th>
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<th>df</th>
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<th>P-value</th>
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<td><strong>Original Stem Basal Area (cm²/m²)</strong></td>
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<td></td>
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<tr>
<td>R</td>
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<td>(16)</td>
<td>(0)</td>
<td>(0)</td>
<td>(3)</td>
<td>(5)</td>
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<td>-92</td>
<td>-86</td>
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<tr>
<td>RF</td>
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<td>(1)</td>
<td>(0)</td>
<td>(4)</td>
<td>(14)</td>
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<tr>
<td><strong>Original Stem Density (individuals/m²)</strong></td>
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<td></td>
<td></td>
<td></td>
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<tr>
<td>FRH</td>
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<td>-8</td>
<td>-24</td>
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<td><strong>Spartina spartinae Cover (%)</strong></td>
<td></td>
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<td>RHF</td>
<td>-55</td>
<td>-5</td>
<td>-24</td>
<td>-7</td>
<td>-6</td>
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<tr>
<td><em>(Trin.)</em></td>
<td>(2)</td>
<td>(31)</td>
<td>(12)</td>
<td>(8)</td>
<td>(27)</td>
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Table 4. Two-way analysis of variance (actual values; time, treatment, and time x treatment) for eight response variables across eight treatment regimes [mechanical roller-chopping (R), prescribed fire (F), roller-chopping followed by fire (RF), roller-chopping followed by chemical herbicide (RH), roller-chopping followed by fire and herbicide (RFH), roller-chopping followed by herbicide and fire (RHF), fire followed by roller-chopping (FR), and fire followed by roller-chopping and herbicide (FRH)] for a coastal prairie restoration project located at the U.S. Fish and Wildlife Service-Bahia Grande Unit near Laguna Vista, TX.

<table>
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<th>Response Variable (All Treatments)</th>
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<th>Time x Treatment</th>
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<tr>
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<td>df</td>
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<td>P-value</td>
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<td>Original stem basal area (cm²/m²)</td>
<td>3, 58.07</td>
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<td>Original stem basal area (cm²/indiv)</td>
<td>3, 58.11</td>
<td>26.10</td>
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<tr>
<td>Original stem density (indiv/m²)</td>
<td>3, 58.01</td>
<td>155.85</td>
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<td>Sprout density (indiv/m²)</td>
<td>2, 43.23</td>
<td>2.79</td>
<td>0.07</td>
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<td>Bare area (%)</td>
<td>3, 58.21</td>
<td>51.84</td>
<td>&lt;.0001</td>
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<tr>
<td>Leaf litter (%)</td>
<td>3, 58</td>
<td>33.28</td>
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<td>Woody debris (%)</td>
<td>3, 57.19</td>
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<td>Spartina spartinae (%)</td>
<td>3, 58.01</td>
<td>25.29</td>
<td>&lt;.0001</td>
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Table 5. Control plot means ± (SE) and analysis of variance (ANOVA) for eight response variables [original stem basal area (cm²/m²), original stem basal area (cm²/indiv), bare area (%), leaf litter (%), woody debris (%), and gulf cordgrass (*Spartina spartinae* (Trin.) Merr. ex Hitchc.) (%)] across four sampling periods for a coastal prairie restoration project located at the U.S. Fish and Wildlife Service-Bahia Grande Unit near Laguna Vista, TX.

<table>
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<th>Response Variable (Controls)</th>
<th>Sampling Period</th>
<th>Time</th>
<th>F-value</th>
<th>P-value</th>
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<tr>
<td>Original stem basal area (cm²/m²)</td>
<td>5.90 (0.86)</td>
<td>5.53 (1.47)</td>
<td>4.94 (1.29)</td>
<td>4.01 (0.76)</td>
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<td>Original stem basal area (cm²/indiv)</td>
<td>68.08 (8.42)</td>
<td>73.17 (4.85)</td>
<td>74.25 (6.73)</td>
<td>60.98 (8.08)</td>
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<tr>
<td>Original stem basal area (indiv/m²)</td>
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<td>0.08 (0.02)</td>
<td>0.08 (0.03)</td>
<td>0.06 (0.01)</td>
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<td>Bare area (%)</td>
<td>11.57 (3.85)</td>
<td>24.35 (3.72)</td>
<td>12.96 (4.98)</td>
<td>6.94 (2.08)</td>
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<td>Leaf litter (%)</td>
<td>79.63 (1.85)</td>
<td>81.48 (3.70)</td>
<td>83.33 (0.00)</td>
<td>81.48 (1.85)</td>
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<td>Woody debris (%)</td>
<td>1.85 (1.85)</td>
<td>12.96 (8.07)</td>
<td>12.96 (6.68)</td>
<td>5.56 (3.21)</td>
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<tr>
<td><em>Spartina spartinae</em> (%)</td>
<td>63.89 (5.21)</td>
<td>55.37 (3.50)</td>
<td>61.85 (3.07)</td>
<td>48.89 (9.31)</td>
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Table 6. Two-way analysis of variance (ANOVA) results of burn temperatures across six treatment regimes [prescribed fire (F), fire followed by roller-chopping (FR), fire followed by roller-chopping and herbicide (FRH), roller-chopping followed by fire (RF), roller-chopping followed by fire and herbicide (RFH), and roller-chopping followed by herbicide and fire (RHF)] for a coastal prairie restoration project located at the U.S. Fish and Wildlife Service-Bahia Grande Unit near Laguna Vista, TX.

<table>
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<th>Height</th>
<th>F-value</th>
<th>P-value</th>
<th>Treatment</th>
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<th>P-value</th>
<th>Height x Treatment</th>
<th>F-value</th>
<th>P-value</th>
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<tr>
<td></td>
<td>df</td>
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<td>df</td>
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<tr>
<td>Burn Temperatures</td>
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<td>0.0008</td>
<td>5, 19.99</td>
<td>0.29</td>
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