THE EFFECTS OF PRESCRIBED FIRE ON WETLAND ECOLOGICAL COMMUNITIES

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Abstract:

1. Wetland Reserve Enhancement Program restorations in Oklahoma have become overrun by a native invasive species, *Carex hyalinolepis*, creating a structurally homogeneous condition with decreased plant diversity. We investigated the response of *C. hyalinolepis* to winter and summer prescribed burns in a field study. We also conducted greenhouse experiments examining the response of *C. hyalinolepis* shoot production and aboveground biomass to fire, simulated grazing, and flooding. Results of the field study indicated a short-term response of *C. hyalinolepis* to fire with *C. hyalinolepis* cover returning to pre-fire levels within several months of the prescribed burns. The greenhouse study results suggest a combination of fire and flooding can reduce *C. hyalinolepis* aboveground biomass and shoot growth. Implications of results from both the greenhouse and field study indicate that a combination of management methods may be most successful at reducing the impact of invasive species.

2. The accuracy and historical accounts of wildfire and prescribed fire on the Texas southeastern coast may provide perspective and context on the role fire plays in these ecosystems. Using remote sensing techniques, we mapped prescribed fires on the Aransas National Wildlife Refuge between 1985 and 2013. Results indicate that the refuge is maintaining the fire dependent ecosystems with a prescribed burn program that includes a mean fire return interval between 2 and 10 years on a majority of the refuge. Quantifying the current fire regime will be useful for future management efforts on Aransas National Wildlife Refuge.

3. The endangered Aransas-Wood Buffalo whooping crane (*Grus americana*) population is growing and will eventually need additional suitable habitat outside of the current refuge boundaries in order to achieve the goal of down-listing the species from endangered to threatened. Sixty years of crane location data coupled with GIS analyses were used to develop predictive models that forecast crane habitat use based on landcover and refuge management activities as well as to determine spatial patterns of cranes on the refuge. Results indicated that the amount of wetland and tidal flat habitat and the distance required for cranes to reach water and wetlands influence habitat selection. Cranes are significantly clustered along the southern coast of Aransas National Wildlife Refuge (NWR), the Lamar Unit of Aransas NWR, Matagorda and Sand Jose Islands and to the east of Aransas across the bay. The areas of high clustering correspond to crane habitat preferences. Our results indicate locations that can be protected in the future and what habitat types can be increased on and around the refuge.

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CHAPTER I

RESPONSE OF A DOMINANT NATIVE *CAREX* SPECIES TO NATURAL DISTURBANCE ON RESTORED WETLANDS

ABSTRACT

Alteration of natural disturbance processes in wetlands can lead to changes in the plant community such as decreased species richness and increased invasive species. Wetlands are dynamic ecosystems adapted to natural disturbances such as floods and these disturbances are important in maintaining ecosystem services provided by wetlands. Wetland Reserve Enhancement Partnership (WREP) restorations in central Oklahoma have become overrun by a monoculture-forming sedge, *Carex hyalinolepis* Steud. (shoreline sedge), creating a structurally homogeneous condition with decreased plant diversity. The loss of both plant community and structural diversity in wetlands reduces available wildlife habitat, limits ecosystem services such as nutrient cycling, and can impede recreational opportunities for landowners. Returning disturbance to restored wetlands can increase the ecological benefits these habitats provide, but it is important to understand the effects of disturbance on invasive species occurring on WREP restorations. We investigated the response of *C. hyalinolepis* cover to winter and summer prescribed burns in a field study. We also conducted controlled greenhouse experiments examining the response of *C. hyalinolepis* shoot production and aboveground biomass to fire, simulated grazing, and flooding. Results of the field study indicated a short-term response of *C. hyalinolepis* to fire with *C. hyalinolepis* cover returning to pre-fire levels within several months following prescribed burns. The greenhouse study results suggest a combination of fire and flooding can reduce *C. hyalinolepis* aboveground biomass ($p<0.05$) and shoot growth ($P <0.0001$). *C. hyalinolepis* is a native wetland obligate that is well adapted to both flooding and fire. Simply returning one disturbance mechanism to the restoration may not be enough to reduce the cover of *C. hyalinolepis* and lessen its negative impacts on plant diversity. Future research should focus on the interactions of other disturbance mechanisms to help reduce the impacts of invasive species. One solution is to leave patches of sedge habitat intermixed in a mosaic of wetland habitat types, which could be beneficial for species such as bitterns. Our results suggest that prescribed fire alone is not enough to reduce the cover of *C. hyalinolepis*, but a combination of disturbances may be more effective at increasing plant diversity on wetland restorations.

KEY WORDS: *Carex hyalinolepis* Steud., flooding, grazing, management, native invasive species, prescribed fire, Oklahoma, shoreline sedge

INTRODUCTION

Wetlands provide many ecosystem services such as wildlife habitat, flood control, and groundwater recharge. Despite the importance of wetlands, over half of the wetlands in the conterminous United States have been lost or severely degraded (Dahl 1990). As the value of wetlands to society has been realized, the focus on mitigating wetland loss and restoring degraded wetlands has become a primary goal for some federal and state agencies. To help combat wetland losses, the United States Department of Agriculture (USDA) created the Wetland Reserve Enhancement Partnership (WREP). The WREP, formerly called the Wetlands Reserve Program, has been instrumental in helping mitigate wetland loss through restoration of wetland habitats and establishment of conservation easements on private lands throughout the United States (Gray and Teels 2006). The WREP program currently has over one million wetland acres enrolled in conservation easements (USDA 2019). Restoration of degraded wetlands is important for safeguarding the ecosystem services provided by these habitats. While many WREP sites have records of past ditching, channelization, and presence of water control installments, many have been successfully restored with functional hydrology (De Steven and Gramling 2012). In general, restored WREP wetlands provide more ecosystem services and have a higher socio-economic value than agricultural lands (De Steven and Gramling 2012, Jenkins et al. 2010). Wetland restoration on WREP properties has proven to be successful and beneficial for amphibians, birds, and other wetland dependent wildlife at both broad and fine scales (Waddle et al. 2013, King et al. 2006). However, the disruption of natural disturbance regimes in wetlands can hinder restoration success.

Inherent heterogeneity in ecosystems is often a product of natural disturbance (Fuhlendorf and Engle 2001), and natural disturbance mechanisms such as fire, herbivory, and flooding are common in wetlands. However, changes to natural disturbance mechanisms in human-modified landscapes that include wetlands have resulted in less diverse ecological communities (Marty 2015, Nielsen et al. 2013, Catford et al. 2011, Chambers et al. 1999). For example, the change in hydrologic regime or fire frequency and severity resulting from the loss of natural disturbance regimes in wetlands can lead to undesirable shifts in vegetation communities or foster the introduction of invasive species. Returning natural disturbances to wetlands is a potential management tool to help improve wetland function and enhance ecosystem services. In particular, this approach may be important for improving functions and services in wetland restorations.

Alteration of natural disturbances in wetlands can precede the development of plant communities with very low species richness or can promote the spread of invasive species. Wetlands are complex ecological systems that are highly susceptible to developing monotypic stands of individual species due to the natural buildup of nutrients and organic materials (Zedler and Kercher 2004). Native species with invasive characteristics can easily dominate wetlands with the same implications as non-native invasive species by creating monocultures (Rojas and Zedler 2015, Levin et al. 2006, Adams and Galatowitsch 2005) and by altering disturbance regimes (Gaertner et al. 2014, Keeley et al. 2005, Brooks et al. 2004, Mack and D'Antonio 1998). For example, invasive wetland plants have been shown to alter hydrology by changing surface flow and water table depth (Gordon 1998) and alter fire regimes by changing fuel loads (Davies and Nafus 2013, Berry et al. 2011).

One management tool that has been used to restore the natural disturbance regime in wetlands is prescribed fire. Fire is a beneficial ecological disturbance that has occurred on the landscape long before human arrival (Bowman et al. 2009) and is a natural disturbance mechanism that helps maintain ecosystem states (Cissel et al. 1999). Fire can create gaps in canopies, reduce fuel loads, influence nitrogen and phosphorous cycling, improve forage, create mosaics of habitat types (DeBano et al. 1998), and improve species richness (Thom and Seidl 2016). Scientific literature provides many examples of invasive plant species being controlled by the use of prescribed fire (Strong et al. 2013, Ayala-A. et al. 2012, DiTomaso et al. 2006, Kyser and DiTomaso 2002). While fire history varies with location, documented records attest to the presence and regularity of fire occurring throughout the Great Plains (Engle and Bidwell 2001, Hart and Hart 1997, Ford and McPherson 1996, Stewart 1951), which likely burned wetlands, particularly in dry seasons. Fire in wetlands often affect inorganic compound composition, macronutrient presence, plant composition, and increase or decrease nutrient cycling abilities depending upon the fire frequency and intensity (Liao et al. 2013, Gu et al. 2008, Smith et al. 2001, DeBano et al. 1998, Johnson and Knapp 1995, Laubhan 1995).

Returning natural disturbance regimes such as fire to wetlands has potential management applications, but vegetation response to prescribed fire in wetlands can be variable and equivocal (Flores et al. 2011, McWilliams et al. 2007, Clark and Wilson 2001, Kost and De Steven 2000, Johnson and Knapp 1995). Studies have shown increases in biomass and changes in vegetation composition post-fire (Flores et al. 2011, Ford and Grace 1998, Bowles et al. 1996, Johnson and Knapp 1995), while others have shown neither a change in annual production by plant species nor compositional changes (Hogenbirk and Wein 1991, Smith and Kadlec 1985a, Smith and Kadlec 1985b). Previous research suggests that most responses to prescribed fire in wetlands are site and species specific (Wang et al. 2017, Flores et al. 2011, McWilliams et al. 2007, Pendergrass et al. 1999, Johnson and Knapp 1995, Auclair et al. 1976). Given the complexity of relationships among water levels, nutrient availability, precipitation, timing of fire, and fire intensity, it is clear why different patterns of fire-vegetation relationships have been observed among different wetland systems. However, returning fire to wetlands is critical to restoring the structure and function of wetlands by preventing woody encroachment (Luvuno et al. 2016), increasing sprouting density of seedlings (Wang et al. 2017), maintaining habitat structure (Fuhlendorf et al.

2006), reducing existing fuel loads (North et al. 2012), and improving the nutritive quality of plants for wildlife (Boyd and Bidwell 2001, Carlson et al. 1993, Smith et al. 1984).

Another important disturbance factor in wetlands is herbivory. Bison (*Bison bison*) and cattle herbivory have historically influenced vegetation structure and heterogeneity of grassland communities in the Great Plains (Derner et al. 2009, Fuhlendorf et al. 2008, Adler et al. 2001, Hartnett et al. 1996). Sedge-dominated wetlands have been described as wetter versions of tallgrass prairies (Warners 1997) and as such, herbivory likely played a role in these types of wetlands. Furthermore, sedges have been documented as bison forage (Jung et al. 2015), indicating sedge wetlands likely experienced both trampling and grazing disturbance by bison. This indicates the presence of another disturbance mechanism that historically affected wetlands and could be re-established to help wetland restorations succeed. Disturbance from grazing herbivores can impact wetlands in several ways. Large grazers can displace soil and create bare ground, which affects microclimate and can facilitate germination of species that require exposed soil (Touzard et al. 2002). Grazers have the potential to transport seeds (either native or non-native) (Morris and Reich 2013), which can increase diversity (Mester et al. 2015, Marion et al. 2010). Grazing also has the potential to maintain non-native species in lower abundance (Marty 2005), which can alter the structure of wetland vegetation (Jones et al. 2011). In addition to trampling soil/vegetation and erosion (Morris and Reich 2013), grazing by larger ungulate species can decrease soil density and increase salinity (Teuber et al. 2013).

Flooding is one of the most common disturbances associated with wetlands. Although it is common to expect wetlands to be wet at all times, many riverine floodplain wetlands experience cycles of flooding and drying with floods supplying important pulses of nutrients.

Plant nutrient availability often comes from sediment deposition from these flooding events (Olde Venterink et al. 2006). The hydrological impacts of flooding can be grouped into five classes: magnitude, timing, frequency, duration, and rate of change of water (van der Valk 2012). Each of these characteristics can affect plant communities and species present (Zhiqiang et al. 2016, Webb et al. 2012). Water levels can affect plant community composition (Wilcox and Nichols 2008) based on species specific environmental requirements and adaptations (e.g., emergent versus submergent species). For example, duration of flooding can influence changes such as obligate wetland species becoming more prevalent because they are able to withstand long periods of inundation (Nielsen et al. 2013, Raulings et al. 2010, Barrett et al. 2010) versus species adapted to drier conditions that will likely perish if wetlands are flooded for long periods of time. Water level manipulation in wetlands is a frequently used management tool, particularly in WREP wetlands that have water control structures installed during restoration.

Several WREP wetland restorations in central Oklahoma have become overrun by a monoculture forming sedge, *Carex hyalinolepis* Steud. (shoreline sedge), creating a structurally homogeneous condition with decreased plant diversity. Disturbance regimes that have been successfully re-established have been able to return restored wetlands to conditions similar to natural wetlands (Bortolotti et al. 2016), improve wildlife habitat for vulnerable species (Walls et al. 2014, Conway et al. 2010), and restore native obligate wetland communities (Martin and Kirkman 2009). Returning disturbance to the landscape through management activities has potential to reduce dominant species cover. Therefore, we implemented a dual approach to study the response of *C. hyalinolepis* to fire and other natural disturbance mechanisms (e.g., grazing and flooding) with a field study and under controlled greenhouse conditions. The aims of the study were to: 1. Determine if a reintroduction of fire as a disturbance mechanism can reduce the cover of *C. hyalinolepis* and improve species richness in restored wetlands; 2. Determine the effect of fire, grazing, and flooding on shoot growth and mortality in *C. hyalinolepis*; 3. Determine the response of aboveground biomass of *C. hyalinolepis* to disturbance mechanisms of fire, grazing, and flooding; and 4. Provide management recommendations to reduce the cover of *C. hyalinolepis*.

METHODS

We implemented a field study and two greenhouse studies to elucidate the impacts of disturbance on *C. hyalinolepis*-dominated wetlands and on *C. hyalinolepis* directly. We chose this approach to assess the impacts in the field where it is more ecologically relevant, but also in a controlled environment where variables could be accounted for and monitored.

Wetland Field Experiment

Site Description

We selected four restored riverine floodplain wetlands (Dvorett et al. 2012, Brinson 1993) adjacent to the Deep Fork River in Lincoln County, Oklahoma that were enrolled in WREP and were predominantly covered by *C. hyalinolepis*. The restored floodplain wetlands that abut the Deep Fork River have dissimilar hydrologic characteristics, soil properties, and higher percentages of disturbance tolerant invasive species than were historically present (Hough 2011, Nugent 2011, Hartzell et al. 2007).

All four research sites selected were previously under agricultural cultivation or abandoned during the 1980s and early 1990s until wetland restoration was completed between 2003 and 2014 under the Wetland Reserve Program (WRP), which is now WREP

(D. Fowler, USDA-NRCS, personal communication). Site 1 is approximately 78 hectares and was placed under easement at the beginning of 2013. Restoration included constructing new embankments and water control structures. Work was completed in January 2014 (Fig.1). The property was contracted under the United States Fish and Wildlife Service (USFWS) as a part of the Partners for Fish and Wildlife Program, which occurred more than a decade prior to enrollment. Site 2, a 98 hectare wetland, was enrolled in the WREP program in 2007 and restoration work was completed in 2012 (Fig. 2). Prior to restoration, the land was planted in sorghum (*Sorghum* spp.). Restoration work included building new embankments as well as rebuilding existing embankments, installing water control structures, and planting trees around pond areas. The current management regime is to hold water during the fall and winter for waterfowl and to drawdown the water during the spring. Restoration of Site 3, a 79 hectare wetland, began in 2003 and was completed later that year with construction of new embankments and installation of water control structures (Fig. 2). Since completion of the restoration, little to no management has occurred on this unit. Site 4 (Fig. 2) was enrolled in the WREP in 2003. Restoration included repairing existing embankments, removing an old embankment, creating a new embankment, as well as installing water control structures. Restoration was completed in 2005. No management has been conducted on the property since completion. Unit 4 was previously enrolled in the USFWS Partners for Fish and Wildlife Program, who completed the original restoration of the wetlands on the unit in 1992 (D. Fowler, USDA-NRCS, personal communication).

Each site had burned and unburned units assigned to it, with location of units delineated based on the amount and location of large patches of *C. hyalinolepis*. This resulted in seven burn units and seven unburned units across the four properties, totaling fourteen research units. Each unburned research unit, where the vegetation monitoring took place, was approximately 0.72 hectares in size (range: 0.42 – 1.15 ha), while burn units were approximately 1.01 hectares each (range: 0.48-1.64 ha). The yearly total precipitation near the research locations was 75.16 cm in 2014, 126.72 cm in 2015, 84.05 cm in 2016, and 129.36 cm in 2017. 2015 and 2017 were much wetter years compared to the long term average of 87.12 cm of precipitation for central Oklahoma (Oklahoma Climatological Survey 2019).

Prescribed Burns

Prescribed burns were conducted in mid-February 2015 (Sites 3 and 4) and late summer (late August/early September) 2015 (Sites 1 and 2). A wildfire burned site 2 (all vegetation on the burned and unburned units was combusted) during early winter 2017. The winter burns on sites 3 and 4 were burned with a backfire, but other fire-related data were unavailable for the winter burns and wildfire. During the late summer burns, winds were 5.6 km/h, temperature was between 34.5ºC and 36.1ºC with relative humidity dropping from 70.4 to 44%. Site 1 had two prescribed fires, one on each of the burn units. The first unit was burned with a slow moving headfire with 75% consumption of dry matter and 10% consumption of green living matter. The second unit was also burned with a headfire and had 75% consumption of dry matter and 60% of green material. Research site 2 had two burn units as well that were both burned with a headfire. The burn on unit one resulted in 75% consumption of both dry and green matter. The prescribed fire on unit two resulted in a very spotty burn with only 25% dry matter combusted and 10% green vegetation burned. Re-evaluating the burn the next day indicated that few plots burned due to the wetness of the plant material, particularly *Eleocharis* spp.; therefore, it was reclassified as an unburned control unit. Inherent variability and unpredictability common in ecological studies resulted in fewer burns than planned and uncontrolled flooding on several sites.

Vegetation Sampling

Each burned and unburned unit on a research site had ten nested plots placed on them for vegetation sampling, resulting in 140 vegetation plots. Each vegetation plot used a nested plot design where the large plot was 5 meters long by 2 meters wide, allowing for the establishment of 10, 1-meter x 1-meter plots (Fig. 3). T-posts were used to mark the location of the plot. Prior to data collection, a randomly selected cardinal direction was used to determine the orientation of the plot and location from the t-post marker. The plots were located 1 meter away from the t-post to prevent potential influence of disturbance on the vegetation where the post was inserted into the ground. Each vegetation sampling plot was placed a minimum of 5 meters from water, trees greater than 1 meter tall, and edge habitat (e.g., roads and dikes). Each sampling plot was at least 11 meters from all other plots in any given direction.

Vegetation data were collected twice during the growing season from 2014-2017 in late May/early June and late July/early August. Data were collected from every 1-meter x 1-meter plot. We estimated cover using Daubenmire cover classes for functional groups, including graminoids, forbs, litter, bare ground, and *Carex* spp. We also estimated cover for each species found within the 1m x 1m plots.

Feral Hog Damage

During spring of 2015, feral hogs (*Sus scrofa* Linnaeus) damaged several sampling units. Photos were taken of every 1m x 1m vegetation plot for the duration of the research. Therefore, we were able to document the damage done by the hogs with the photographs. A digital grid was placed over the 1m x 1m plot in the photo and percent damage was estimated based on how many grids showed vegetation damage and evidence of rooting and trampling.

Greenhouse Experiments

The greenhouse experiments were designed to test the effects of disturbance on *C. hyalinolepis* shoot production and aboveground biomass. A minimum of 270 *C. hyalinolepis* plants were extracted from the field research wetlands prior to the beginning of the greenhouse experiments. Plants were collected and maintained in native clay soil in 4-L pots. Litter and dead material were removed from the pots and *C. hyalinolepis* plants were allowed to grow for several months in the greenhouse before treatments were applied. When removing the plants from the field, a group of shoots was selected and a circle was dug around the shoots in the approximate diameter of the pots used to contain the plants. We attempted to remove the entire rhizome when possible to prevent the sedge from being killed. Growing conditions were maintained similar to field conditions by using ambient light from the greenhouse and maintaining the temperature at a minimum of 21°C. Plants were watered two to three times a week to field capacity. The greenhouse research was divided into two experiments. The first experiment tested the impacts of burning, flooding, a combination of burning and flooding, and grazing on shoot production of *C. hyalinolepis* and was conducted from June through December 2015. The second experiment focused on the effect of treatments on aboveground biomass of *C. hyalinolepis* and was conducted February through August 2016. Fifteen individual 4-L pots with *C. hyalinolepis* were used for testing each treatment. The treatments were chosen because they resembled potential management options that could be used on WREP properties by private landowners.

Greenhouse Experiment 1

A randomized block design was implemented to test the efficacy of various management treatments that mimicked natural disturbance mechanisms on reducing shoot production or inducing mortality of *C. hyalinolepis*. The number of shoots per pot was counted bi-weekly for the extent of the experiment. Each experimental treatment was monitored for one month after the treatment to determine if the treatment induced mortality. The treatments included flooding, clipping (to mimic grazing), burning, and a combination of burning and flooding. Plants were exposed to flooded conditions for one, two, four, and six months. Pots were placed in large plastic containers and filled with tap water until the depth was 10 cm above the soil level (Ewing 1996), and this depth was maintained for the duration of the treatment. Plants were clipped to various levels above the soil surface to mimic grazing. Leaves were clipped to the soil surface (0 cm), 1 cm, 5 cm, and 10 cm above the soil surface at the beginning of the experiment and again two weeks after the initial clipping.

Plants were burned once or twice, with the second burn occurring two weeks after the initial burn. To conduct the burns, individual *C. hyalinolepis* plants in pots were placed under a 0.9 x 1.8 m metal table with a 20 cm circle removed from the center that was level with the soil in the pot in order to burn one plant at a time (Limb et al. 2011). All fires were conducted with a headfire and 0.5 kg of dry native prairie hay, which was equivalent to field litter levels, placed around the top of the plant to carry the fire through. The first set of burns were conducted with an average air temperature of 27.69°C, relative humidity of 53.92%, and a wind speed of 5.9 km/h. Average litter depth was 0.14 m and mean flame height was 0.42 m. It took an average of 87 seconds for the fire to carry through and burn individual pots of *C. hyalinolepis.* The second set of burns was conducted with an average air temperature of 20.84°C, relative humidity of 77.8% and a wind speed of 5.15 km/h. Mean litter depth was

0.121 m, average flame height was 0.3 m, and it took an average of 86 seconds for each plant to burn. All living biomass was consumed on each pot. Plants that were exposed to the burn and flood treatment were first burned with the aforementioned method and flooded immediately after the burn, following the same treatment as flooded plants.

Greenhouse Experiment 2

For the second experiment, plants were exposed to similar treatments used in experiment 1 and followed the same methods for burning, clipping, and flooding. However, we did not have as many treatment levels in the second experiment as in the first experiment. Aboveground biomass of *C. hyalinolepis* was monitored to determine effect of the treatments. Plants were clipped to the soil level, burned once, flooded for one month, and burned and flooded for one month. The pots for the burn and flood and flood treatments were removed from the flooded conditions at week 4. Plants were burned using the method aforementioned in experiment 1. The burns were conducted in conditions with an average temperature of 14.57°C, an average relative humidity of 44.52%, and a mean wind speed of 7.02 km/h. The average litter depth was 0.141 m, mean flame height was 0.3 m, and it took an average of 55 seconds for each pot to burn. All living biomass was consumed in each pot.

The pots were monitored once a month for six months. Plant aboveground biomass was determined using remote sensing techniques similar to field methods of Limb et al. (2007) and Boyd and Svejcar (2005). Photographic images were taken of plants, and pixel numbers of each plant were used to determine aboveground biomass. A photo booth was constructed of white cloth to reduce background noise and to allow ambient light to enter. Ambient light from the greenhouse was used as artificial light would have created shadows. Pots were placed on the center of a table marked by an ‘x’ so that the same location could be replicated for all plants. The soil was covered with white cloth and a white cloth backdrop was placed around the pot and on the table to prevent excess noise in the photo. The plastic rim on the pots was trimmed to soil level so that the edge of the pot did not obstruct view of the sedge leaves. A tripod mounted camera was placed 112 cm from the edge of the table and photos were taken with a Nikon Coolpix AW100 with an ISO of 124, f/3.9 on the portrait setting, and 4608 x 3456 pixels in size. Photos were edited to remove the background and converted to monochromatic black and white photos (Fig. 4). Pixel number associated with each plant was determined using ENVI 5.3 software (Exelis Visual Information Solutions, Inc., Harris Corporation, 2017).

Data Analyses

Wetland Field Experiment

We analyzed the impacts of summer and winter prescribed burns on several variables using a generalized linear mixed model for repeated measures, which models the covariance structure. The variables were species richness and percent cover of *Carex* spp. (the dominant species was *C. hyalinolepis*, but one site had approximately 25% coverage of *Carex crus-corvi* Shuttlew ex. Kunze (ravenfoot sedge)), litter, bare ground, and forbs. All percent cover responses were analyzed using a beta distribution and species richness was analyzed using a Poisson distribution. All tests were done to the 0.05 significance level. Analyses were conducted separately for winter and summer burns. Burn treatment (summer or winter burn) was a fixed effect, and the combination of year and data season were the repeated measures. We also included interactions between years, season, and burn. If an interaction occurred, we used Tukey-Kramer pairwise comparisons since repeated measures are likely not independent. Data were not collected for both spring and summer seasons for all four years due to substantial flooding in 2015 and 2016 that prevented access to the research sites.

Water depth was not recorded, but coverage of water was monitored and collected throughout the experiment and any plot that had three percent or more water present was given a value of one, and those that had no water present were given a value of zero. The presence of water in the vegetation sample units indicated that the site was currently flooded or had been recently flooded. Although flooding occurred on all sites, plots with standing water when data were collected were removed for the analyses in order to attempt to elucidate the effects of the prescribed fires on the variables in question.

Greenhouse Experiments

For the first experiment, we analyzed the impacts of various treatments on shoot production of *C. hyalinolepis* plants using a generalized linear mixed model for repeated measures. The treatments were clipping to mimic grazing at 0, 1, 5, and 10 centimeters above the soil; burning once or twice; flooding for 1, 2, 4, or 6 months; or burn and flood for 1 or 2 months. All tests were evaluated at the 0.05 significance level. Treatments (clip, flood, burn, burn and flood) were fixed effects, and weeks were the repeated measures. We included interactions between treatments and weeks. If an interaction occurred, we used Tukey-Kramer pairwise comparisons.

For the second experiment, and in order to verify that aboveground biomass was represented accurately by the number of pixels of each plant in the photos, a linear regression was run using aboveground biomass of 43 *C. hyalinolepis* plants. Plants were dried for two weeks and weighed to the gram to model the relationship between aboveground biomass and number of pixels in each image of individual plants (adjusted $R^2 = 0.93$) (Fig. 5). A generalized linear mixed model for repeated measures was used to determine if an interaction occurred between weeks and treatments. The treatments were burn, flood for 1 month, burn and flood for 1 month, and clip to 0 cm. Treatments (clip, flood, burn, burn and flood) were fixed effects and the weeks were the repeated measures.

RESULTS

Wetland Field Experiment

We found that the summer burns had no positive or negative statistically significant effect among burn, season and year on the amount of cover present for *C. hyalinolepis* ($F_{7,34} = 0.43$, $p = 0.874$), litter ($F_{7,34} = 1.92$, $p = 0.096$), bare ground ($F_{7,30} = 2.10$, $p = 0.075$), forbs ($F_{7,34} = 0.52$, $p = 0.813$) or species richness ($F_{7,34} = 0.91$, $p = 0.509$) (Appendix B, figures 1, 3, 5, 7, 9; Appendix A, Table 1 shows effects of summer and winter prescribed burns on individual plant species; Appendix C lists all species observed on the research sites). While there was no significant interaction among seasons, years, and burn treatments, there was an interaction between years and seasons, indicating that some environmental variables, likely the extensive flooding on the WREP properties during 2015 and 2017, affected the changes in percent cover of the aforementioned variables. We found that winter burns did have an effect on the variables measured. An interaction occurred between burn, season, and year for the percent cover of litter ($F_{5,22} = 12.46$, $p < 0.0001$), bare ground ($F_{5,16} = 6.60$, $p = 0.002$), forb ($F_{5,21} = 3.70$, $p = 0.015$), and *C. hyalinolepis* ($F_{5,22} = 9.03$, $p < 0.0001$), and therefore we examined the response of each variable to prescribed fire by burn season and compared results among years. Winter burns had no effect on species richness (Appendix B, Figure 10).

At the beginning of the experiment, control sites and the sites to be burned had similar amounts of litter during the spring ($F_{1,22} = 0.38$, $p = 1.00$) and summer ($F_{1,22} = 0.44$, p

= 1.00) of 2014 (Appendix B, Figure 2). After the prescribed fire was conducted during the winter of 2015, the amount of litter returned to levels comparable to the control sites by summer of 2015 ($F_{1,22} = 4.07$, $p = 0.712$). We were unable to access the sites during the spring of 2015. Litter continued to accumulate on both the burned and control sites and there continued to be no difference between the control and post-burn sites 1.5 years after the burn during the summer of 2016 ($F_{1,22} = 0.34$, $p = 1.00$). A wildfire occurred during the winter of 2017. As expected, fire removed litter from the research plots. The amount of litter was significantly less on the post-burn sites during the spring ($p = 0.012$) and summer ($p = 0.022$) when compared to the levels of litter on the same site the summer before the burn. The post-burn sites also had considerably less litter during the spring ($F_{1,22} = 50.53$, $p < 0.0001$) and summer ($F_{1,22} = 64.29$, $p < 0.0001$) than the control sites in 2017.

At the start of the research, both control and treatment sites had similar amounts of bare ground present in both the spring ($F_{1,16} = 0.01$, $p = 1.00$) and summer ($F_{1,16} = 0.14$, $p = 1.00$) of 2014 (Appendix B, Figure 4). As expected, there was a change and more bare ground was present on post burn sites during the summer of 2015 when compared to pre-burn levels during the summer of 2014 ($p = 0.003$). However, this increase was short lived, and the levels of bare ground decreased quickly. The percent of bare ground present was not statistically different during the summer of 2016 when compared to pre-burn levels during the summer of 2014 ($p = 0.747$). While we observed a slight increase in bare ground when comparing pre-and post-burn sites, we did not observe any difference from post burn sites and the control during summer 2015 ($F_{1,16} = 13.50$, p = 0.061) or summer 2016 ($F_{1,16} = 10.83$, $p = 0.120$). After the wildfire during the winter of 2017, we observed an immediate increase in the amount of bare ground exposed, which was greater than bare ground on the control sites during the spring ($F_{1,16}$ = 30.11, *p* = 0.002). While bare ground began to decrease in the summer as expected because of plant growth and litter accumulation, the amount of bare ground present remained statistically higher on the post-burn sites as compared to the controls ($F_{1,16}$ = 30.73, *p* = 0.002).

During the spring ($F_{1,21}$ = 3.56, *p* = 0.786) and summer ($F_{1,21}$ = 5.79, *p* = 0.476) of 2014, the forb cover on the treatment sites prior to burning and control sites was similar (Appendix B, Figure 6). After the prescribed fire during the winter of 2015, the amount of forb cover remained similar between the control and burn sites the following summer ($F_{1,21}$ = 0.21, *p* = 1.000). The percent cover continued to remain similar between the sites through the summer of 2016 ($F_{1,21}$ = 0.18, *p* = 1.000). After the wildfire during the winter of 2017, there was a spike in spring forb cover, although it was not significantly different from the amount of forb cover on the burn site the previous summer (*p* = 0.305) and remained similar to control sites during both the spring ($F_{1,21}$ = 6.18, *p* = 0.429) and summer ($F_{1,21}$ = 3.18, *p* = 0.837) of 2017. The only significant difference that occurred was that more forb cover existed on post-burn sites during the spring of 2017 when compared to pre-burn sites during the spring of 2014 (*p* = 0.039). However, this was a very short term result as summer 2017 post-burn forb levels were the same as those on pre-burn sites during the summer of 2014 (*p* = 0.890).

The percent cover of *C. hyalinolepis* was not significantly different on the pre-burn sites during the spring of 2014 ($F_{1,22}$ = 9.77, *p* = 0.154) or summer ($F_{1,22}$ = 6.92, *p* = 0.349) when compared to the percent cover on control sites at the same time (Appendix B, Figure 8). Several months after the winter burn in 2015, post-burn sites were not different from the control sites during the summer of 2015 ($F_{1,22}$ = 2.83, *p* = 0.0882). The prescribed burn also had no discernable effect on *C. hyalinolepis* as post-burn levels of sedge cover during the summer of 2015 were similar to those during the summer of 2014 prior to burning (p=1.0000). The trend of no impacts from the prescribed fire continued as no differences were detected in the summer of 2016 between the control and post-burn sites ($F_{1,22} = 9.30$, $p = 0.176$). After the wildfire during the winter of 2017, there was a reduction in the cover of sedge on the post burn site when compared to summer 2016, but it was not significant ($p = 0.299$). There was a significant change on the post-burn sites between the spring and summer of 2017 ($p = 0.002$) where the sedge dramatically recovered. However, the percent cover of sedge was not different from the control sites during the spring ($F_{1,22} = 11.51$, $p = 0.092$) and summer ($F_{1,22} = 6.59$, $p = 0.383$) of 2017.

Feral Hog Damage

The damage done to the vegetation plots from feral hog resulted in no changes to the cover of forbs ($F_{1, 2} = 0.01$, $p = 0.06$), but it did have an impact on litter ($F_{1, 2} = 32.62$, $p = 0.03$), bare ground ($F_{1, 2} = 57.16$, $p = 0.02$), species richness ($F_{1, 77.23} = 7.85$, $p = 0.006$), and sedge cover ($F_{1, 2} = 20.30$, $p = 0.05$).

Greenhouse Experiments

Greenhouse Experiment 1

An interaction between treatments and weeks occurred at the beginning of the initial analyses ($F_{60, 840} = 12.46$, $p < 0.0001$), therefore we compared treatments within weeks. For clarity, the results are discussed by treatment type and are represented in separate figures (Figures 6-12). At the beginning of the experiment there were no differences in shoot number among treatment groups and the control or within treatments ($F_{12, 158} = 1.04$, $p = 0.41$).

However, differences were present among treatments for the other weeks (F > 3.93, $p <$ 0.042).

Clip Treatments

C. hyalinolepis shoot production was not impacted after the first treatment. Shoot production at week two, prior to the second clip treatment, was not different from the control plants at all levels. After the second clip treatment, shoot production was affected by the clip treatment at the soil level (0 cm) when compared to the control, with plants clipped to the soil level having significantly less shoots than the control plants ($p = 0.03$) at the end of the experiment (6 weeks). The 0 cm clip treatment resulted in fewer shoots produced than the control, having 37% less than the average number of shoots compared to the control. Clipping *C. hyalinolepis* to the 1 cm, 5 cm, and 10 cm levels resulted in no significant differences from the control at the end of the experiment (Fig. 7). Clipping to the soil level induced mortality in 6.7% of the samples, while clipping shoots to 1 cm above the soil resulted in 20% of the pots with no shoots post treatment. Clipping shoots to the 5 cm and 10 cm level above the soil resulted in no mortality to any of the plants.

Burn Treatment

Burning *C. hyalinolepis* plants once resulted in no significant difference in shoot number one-month post treatment when compared to the control. Burning once resulted in an average of 6.53 $SE \pm 1.17$ shoots per pot at the end of the treatment with the control having an average of 9.6 shoots $SE \pm 0.95$ (4 weeks). *C. hyalinolepis* was negatively affected when burned twice, resulting in significantly fewer shoots than the control at the end of the experiment ($p = 0.006$). Burning twice resulted in an average of 3.07 $SE \pm 0.84$shoots while the control at six weeks had an average of 10.47 $SE \pm 1.00$shoots (Fig. 8). Burning plants once resulted in no mortality to *C. hyalinolepis*, but burning twice induced mortality in 26.7% of the samples.

Burn and Flood Treatment

Burning and flooding for one month produced plants that had an average of 2.67 $SE \pm 0.80$ shoots one-month post treatment and burning and flooding for two months resulted in an average of 0.6 $SE \pm 0.25$ shoots per pot compared to the control at 12 weeks, which had an average of 13.8 $SE \pm 1.36$ shoots (Fig. 9). Burning and flooding *C. hyalinolepis* for two months reduced the number of shoots produced, resulting in significantly fewer shoots produced than the control plants one month after the treatment ended (12 weeks) ($p < 0.0001$). Burning and flooding one month induced mortality in 33.3% of samples post treatment and burning and flooding for two months killed *C. hyalinolepis* plants in 66.7% of samples.

Flood Treatment

Flooding plants did not negatively impact shoot production at any treatment level. Flooded plants had more shoots than the control after being flooded for two, four, and six months, respectively. It had the opposite impact of inducing mortality. The number of shoots produced after six months of flooding was significantly, and positively, different than the control plants ($p = 0.04$). Flooding resulted in the following average shoot numbers post treatment, with the average shoot number of control plants in parentheses for comparison: 1 month flood = 16.53 (11.5), 2 months flood = 15.13 (13.8), 4 months flood = 20.27 (16.93), and 6 months flood = 25.87 (20.47) (Figs. 10-13).

Greenhouse Experiment 2

An interaction between treatments and weeks occurred at the beginning of the initial analyses ($F_{24,347} = 23.79$, $p < 0.0001$), therefore we compared treatments within weeks. For clarity, the results are discussed by treatment type and are represented in figure 13. At the beginning of the experiment there were no differences in average pixel number between treatment groups and the control and within treatments ($F_{4,1} = 1.86$ $p = 0.50$). However, differences were present among treatments for the other weeks ($F > 2.48$, $p < 0.05$).

Clipping Treatment

Plants that were clipped to the soil level had significantly different aboveground biomass (i.e., the number of pixels of *C. hyalinolepis*) than the control at week 4 ($p < 0.0001$), week 8 ($p = 0.02$) and week 12 ($p = 0.02$). Clipped sedges had less aboveground biomass than the control group and flooded plants, but more than the burn and flood treatment from weeks four through twelve. Clipped sedges had less aboveground biomass than both control and flooded plants at week 20 ($p = 0.03$; $p < 0.0001$). However, by the end of the experiment at 24 weeks, clipped plants had similar aboveground biomass to the control group.

Burn Treatment

Plants subjected to the burn treatment maintained significantly less aboveground biomass than flooded sedges from week 4 through the end of the experiment (weeks 4-24: $p < 0.0001$). Burned plants also maintained a lower aboveground biomass from the control group between weeks 4 and 20 (p values ranged from $p < 0.001$ to $p = 0.01$), but by the end of the experiment the biomass was similar to that of the control ($p = 0.09$).

Burn and Flood Treatment

Burned and flooded plants had less aboveground biomass as compared to the control group from weeks 4 through 24 (*p* values ranged from $p < 0.0001$ to $p = 0.004$). While burning and flooding reduced aboveground biomass in *C. hyalinolepis* plants, it did not cause mortality.

Flood Treatment

The production of above ground biomass by *C. hyalinolepis* was not negatively affected by flooding. Flooding produced a higher average biomass throughout the experiment than all other treatments. It produced significantly more aboveground biomass than the control at weeks 4 through 24 (*p* values ranged from $p = 0.0001$ to $p = 0.04$). The flood treatment was only similar to the control plants aboveground biomass at week 8.

DISCUSSION

Wetlands provide essential ecosystem services. However, the natural disturbance regime of wetlands is critical for maintaining these services. In WREP wetlands in Oklahoma, more intense and/or more frequent disturbance may be needed to break the feedback loop keeping the wetlands dominated by a monoculture forming sedge. *C. hyalinolepis* is a native sedge with invasive characteristics and is well adapted to flooding and fire. The most effective method to reduce the negative impacts of *C. hyalinolepis* is to use a combination of disturbances. Restoring the natural disturbance regime of fire in connection with the natural cycle of riverine flood pulses may be more effective. Results from the field study indicate that burning alone had no impact on reducing *C. hyalinolepis* presence in the wetlands, but results from the greenhouse study suggest that combining disturbance mechanisms was effective in reducing aboveground biomass and effectively inducing mortality of *C. hyalinolepis* shoots. Combining different disturbance mechanisms or altering the intensity and timing of disturbances has the potential to be effective in reducing the cover of some invasive plant species (Zhang and Shea 2012, Firn et al. 2010, Schooler et al. 2010, Davies et al. 2009, Turner 1988).

Field Study

Prescribed fire increased the cover of *C. hyalinolepis*, particularly after the winter burn. Two factors influencing this effect of fire could be the season of burn and the intensity of the fires. Spring burns in March or April may be more effective at reducing *C. hyalinolepis* while plant nutrients and energy are invested in producing seeds, when flowering occurs, and when new tillers are being produced. The prescribed fires were likely not intense enough. The winter burn would have been more intense than the summer burn because of the buildup of dead vegetative material and low moisture content. The summer burn had low intensity fires due to the high moisture content of the vegetation. Several of the summer burned areas had to be reclassified as unburned because the fire consumed less than ten percent of the vegetation. Sedges are dissimilar to many grass species by having their apical meristem below ground, protecting the plant from fire (Strong et al. 2013). Many also reproduce vegetatively and can respond quickly to fire through rapid tiller production (Kimura and Tsuyuzaki 2011, Auclair et al. 1976). More intense fires may have had more of an effect if they had been able to burn hot enough to kill rhizomes below the soil surface. Burns in sedge-dominated systems have had little, if any effect on reducing sedges or increasing species richness, or results were very short term (Kost and De Steven 2000, McWilliams et al. 2007).

One of the objectives of this study was to determine if prescribed fire could be used to improve species richness on restored WREP wetlands in central Oklahoma. We did not observe any changes in species richness or forb cover post-fire or in conjunction with the decrease in litter and increase in the amount of bare ground. Prescribed burns implemented once did not increase species richness or reduce aboveground biomass of *C. hyalinolepis*. In fact, *C. hyalinolepis* has been documented to respond positively to fire in Florida (Cypert 1973). Burned wetlands often experience increases in graminoid cover with a concomitant decrease in forb cover after fire (Flores et al. 2011, Warners 1997, Bowles 1996, Johnson and Knapp 1995), or experience results that were short term with prior conditions returning within less than a year (Schmalzer et al. 1991). Results from the prescribed burns in *C. hyalinolepis*-dominated wetlands were similar in that any changes we observed were temporally short. In contrast to our study of fire in sedge dominated wetlands, Middleton (2002) did observe an increase in species richness post fire in sedge meadows. However, many other examples of burning in sedge-dominated wetlands indicated little to no long-term changes in reducing sedge cover or improving species richness (McWilliams et al. 2007, Kost and De Steven 2000), which is similar to our results. McWilliams et al. (2007) suggest that the effects of burning are more evident when looking at individual species and that the moisture gradient of the wetland likely has more of an impact on plant species community than prescribed fire (Norton and De Lange 2003, Ford and Grace 1998, Taylor et al. 1994). The rapid growth rate and high biomass produced by sedges could provide one explanation of why species richness does not increase as rapid growth and litter buildup allow sedges to outcompete other species.

Several species besides *C. hyalinolepis* were routinely encountered in the WRP wetlands and exhibit invasive characteristics. *Cardiospermum halicacabum* became very dominant during 2017 and has the potential to become a problematic invasive species on the

WREPs due to its quick growth, ability to spread, ample seed production, and seeds that can survive flooded environments before germinating (Noble and Murphy 1975). Mature plants also have the ability to adapt and survive under a range of conditions, including the ability to survive floods (Dempsey 2011). This species is also considered invasive in other regions (Gildenhuys et al. 2013, Hui et al 2011). Although native, *Ambrosia trifida*, similar to *C. hyalinolepis*, has characteristics that make it a potential invasive species that can limit overall diversity in the ecosystems it overruns. *A. trifida* seeds germinate earlier than other species and the seeds have the ability to germinate in a wide range of conditions, giving it a competitive advantage (Abul-Fatih and Bazzaz 1979). Also, *A. trifida* produces large leaves and is taller than many species, resulting in more biomass, which allows the plants to outcompete understory species and reduce overall diversity (Abramova 2012, Bassett and Crompton 1982). This indicates that the WREP wetlands may be prone to other invasive species even if the cover of *C. hyalinolepis* is reduced.

Greenhouse Study

The interaction of disturbance mechanisms that involve fire, mowing, or grazing have been successfully used to reduce invasive plants when followed by flooding to a depth and duration that prevents any remaining leaves, shoots, and roots from obtaining oxygen (Herndon et al. 1991, Ball 1990, Smith and Kadlec 1985b). Combining disturbance methods (i.e., burning and flooding in combination) was more effective at reducing overall sedge biomass than any one management option alone. The most effective method is to burn and immediately flood the wetland to a level above all living sedge material for a minimum of two months. Once the sedge is burned and aboveground biomass is removed, inundating the plant will deprive the roots of oxygen, and if no method such as rapid underwater shoot extension can be produced by the plant to get oxygen, (Cronk and Fennessy 2001), anoxic conditions will eventually kill the plant or damage plant tissues depending on the species' tolerance to flooded conditions (Crawford 2003). Many WREPs have water control structures that allow them to flood and hold water, making this a viable management option for many landowners. Another benefit of using prescribed fire to remove litter followed by flooding will create more bare ground, which will allow other species to germinate when the water is eventually drawn down (Kimura and Tsuyuzaki 2011, de Szalay and Resh 1997, Kirkman 1995, van der Valk 1986, Smith and Kadlec 1985b). Flooding alone at the depth and duration in our study had no negative impacts and instead increased the number of shoots and aboveground biomass of *C. hyalinolepis*.

Many native sedge species become dominant monocultures in wetlands (Bernard et al. 1988, Gorham 1979) and are resilient to disturbances such as grazing (McGranahan et al. 2014) and fire. Sedges (*Carex* spp.) have been shown to have high growth rates in shallow water (Hultgren 1988) and respond positively to grazing due to efficient nutrient uptake from the rhizomes. Grazing causes shoots to remain in a physiologically younger state with high nutrient concentrations (Bernard 1988). Our greenhouse experiment indicates that clipping sedges to mimic grazing resulted in no discernable difference in above ground biomass before and after clipping, which has been demonstrated by other studies (Boyd and Svejcar 2012, Dovel 1996). Grazing could be effective only if the grazer repeatedly eats shoots down to the soil, as clipping sedges to the soil level twice in our study reduced the number of shoots produced. Burning *C. hyalinolepis* plants twice did reduce the number of shoots produced and killed several plants. Burning consecutively in quick succession is an unlikely option as landowners would be required to provide a litter substitute to carry the fire because not enough dry litter would be produced naturally in such a short amount of time following the initial burn. However, this could provide additional insight on using different burn approaches such altering the season of burn or considering fire intensity and taking into account fire dynamics such as fuel moisture, fuel load, and weather conditions (Twidwell et al. 2013). Future research could focus on the short-term and long-term effects of different burning techniques and combinations of disturbance on *C. hyalinolepis.*

Conclusions and Implications

Although *C. hyalinolepis* plants may not be completely eradicated from the wetland, reduction of *C. hyalinolepis* cover presents an opportunity to create more patchiness on the landscape. Maintaining a complex of various habitat patches in wetlands has proven successful for bird conservation (Gabrey et al. 1999). Patches of *C. hyalinolepis* can be managed for the benefit of species such as king rail (*Rallus elegans*) and American bittern (*Botaurus lentiginosus*) for nesting habitat (Dechant et al. 1999, Meanley 1953). Both *Thripsaphis ballii* (Jessie et al. 2018) and although not yet documented in Oklahoma, Duke's skippers (*Euphyes dukesi*) use *C. hyalinolepis* as host plants (Calhoun 1995).

Many invasive grass species are adapted to fire and are more difficult to remove (Grace et al. 2001). Invasive graminoids in wetland ecosystems may have developed similar adaptations. For example, invasive *Phalaris arundinacea* (reed canarygrass) shows high genetic diversity and phenotypic plasticity, making it a successful invasive species in wetlands (Lavergne and Molofsky 2007), but fire has been shown to reduce dominance of this species when the timing of burns is taken into consideration (Lavergne and Molofsky 2006). Therefore, future research could investigate the impacts of the timing of fires on *C. hyalinolepis*. We observed a short term reduction in sedge cover due to the rooting behavior of feral hogs. This suggests that *C. hyalinolepis* could potentially be controlled through disking of the rhizomes and tillers as it has had some success with other invasive species (Bryson and Carter 2012, Annen 2010). Additionally, some *Carex* species have been killed with extensive flooding (Harris and Marshall 1963), suggesting the need for additional study on the impacts of the length and depth of flooding on *C. hyalinolepis*. Future research could also examine the interactions of disturbances such as fire and grazing. While use of herbicides could be studied in combination with disturbance methods, the impacts on wetland wildlife could be detrimental. Reduction in the invasive common reed (*Phragmites australis*) has been successful when glyphosate, a common herbicide ingredient, was applied (Knezevic et al. 2013). The potential problem with its use in wetlands is that some surfactants mixed with glyphosate can create a toxic environment (Moore et al. 2012).

Wetland restorations are critically important for restoring ecosystem services on the landscape and reversing the loss of wetlands throughout the U.S. However, provisioning of many of the important ecosystem services may not be completely achievable in these engineered ecosystems if the natural disturbance regime is not restored and maintained. The need to improve restored wetland function and diversity will continue to present challenges to scientists and managers, but the benefit outweighs the effort because it is necessary to protect wetlands for all the ecosystem benefits they provide (Ghermandi et al. 2010, Jenkins et al 2010). Overall, the control and reduction of *C. hyalinolepis* in restored wetlands will likely require a multi-pronged approach and continuous management to control the spread of this native invasive. The interacting elements of disturbances will produce differing results depending on the disturbance mechanisms and management goals. Simply returning one historic disturbance regime may not be enough to reduce impacts of invasive species and increase wetland plant diversity, but the combination of disturbance mechanisms offers the potential to reduce the impacts of a monoculture forming native species on wetland diversity.

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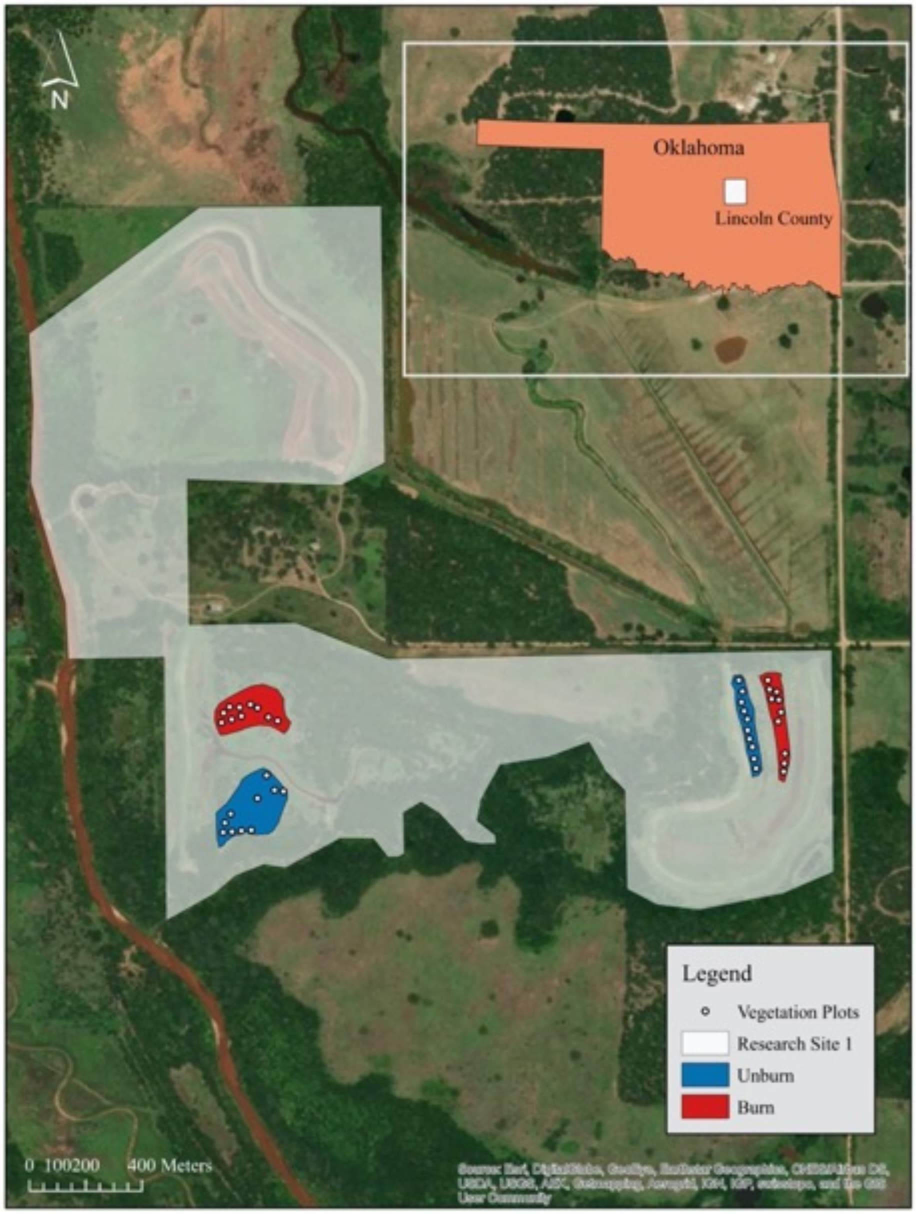
TABLES AND FIGURES

Figure 1. Research site 1 located on a Wetland Reserve Enhancement Program (WREP) property in Lincoln County, Oklahoma, USA. Research site 1 had two burn and two unburned units located in or near *C. hyalinolepis* stands on the property. Each burn and unburned unit had ten vegetation plots used as either a control or to monitor vegetation changes post fire.



Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community

Figure 2. Research sites 2, 3, and 4 on Wetland Reserve Enhancement Program properties in Lincoln County, Oklahoma, USA. Research sites 2 and 3 had two burn and two unburned units located in or near *C. hyalinolepis* stands on the property. One burn unit on both sites two and three were re-classified as unburned units and used as control units. Site 4 had one burn and one unburned unit located on the property. Each burn and unburned unit had ten vegetation plots used as either a control or to monitor vegetation changes post fire.

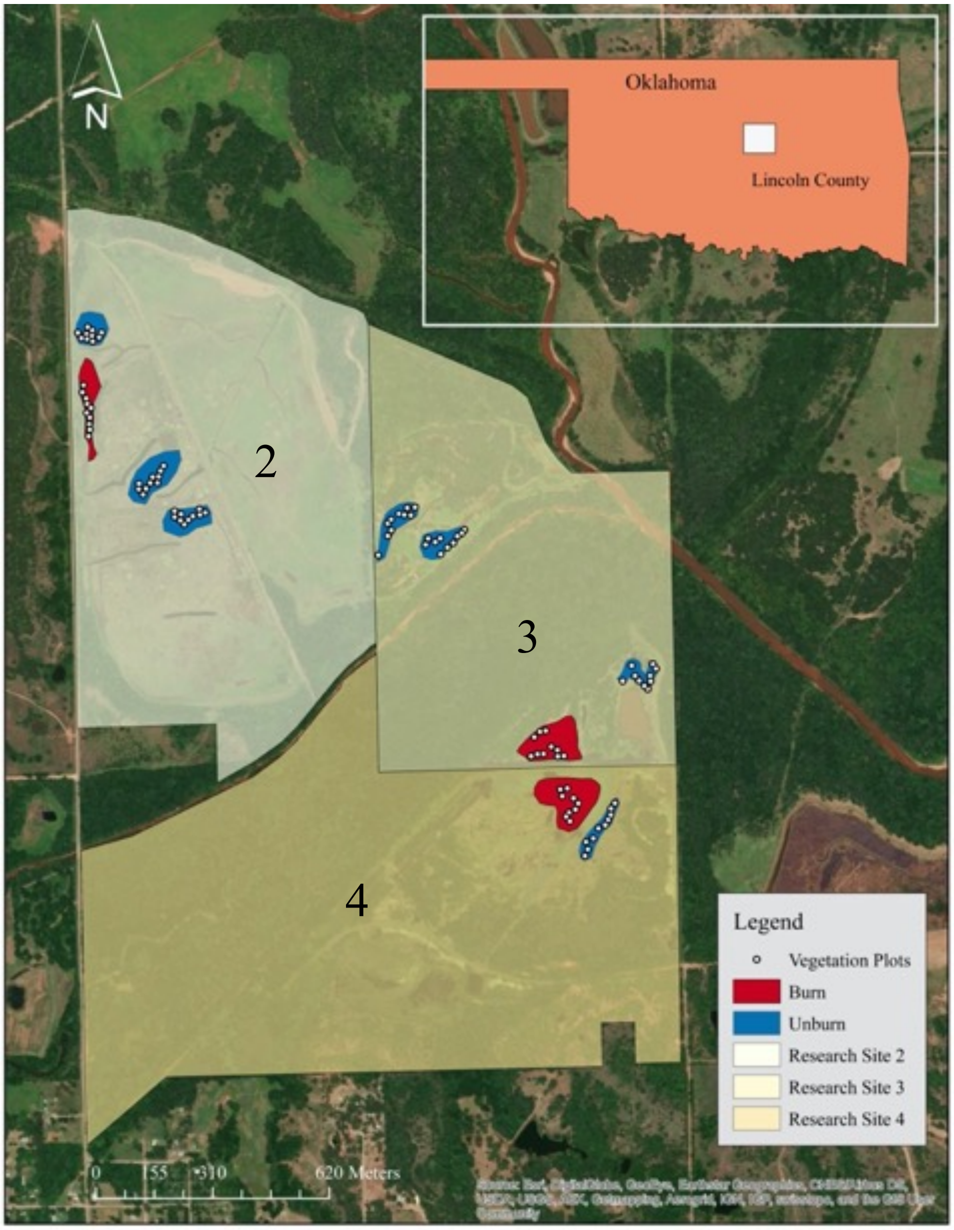


Figure 3. Diagram of vegetation plots used for data collection. Each 5 m x 2 m plot contained 10, 1 m x 1 m nested plots. Each 1 m x 1 m nested plot was used to monitor percent cover of functional groups and plant species composition. Ten 5 m x 2 m vegetation plots were placed on each burn and unburned unit.

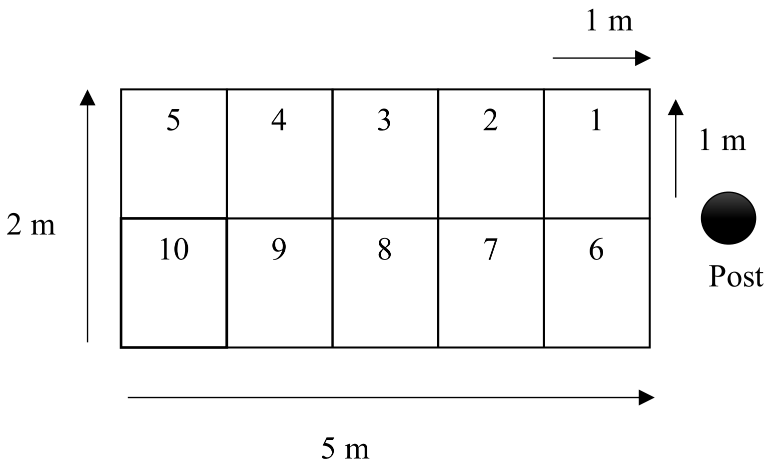


Figure 4. Example of the method used to determine aboveground biomass for individual *C. hyalinolepis* plants. A. Original image of an individual plant taken every four weeks. B. Background noise was removed from the photo to isolate *C. hyalinolepis* leaves. C. Images were converted to monochromatic black and white images and black pixels were used to determine aboveground biomass.

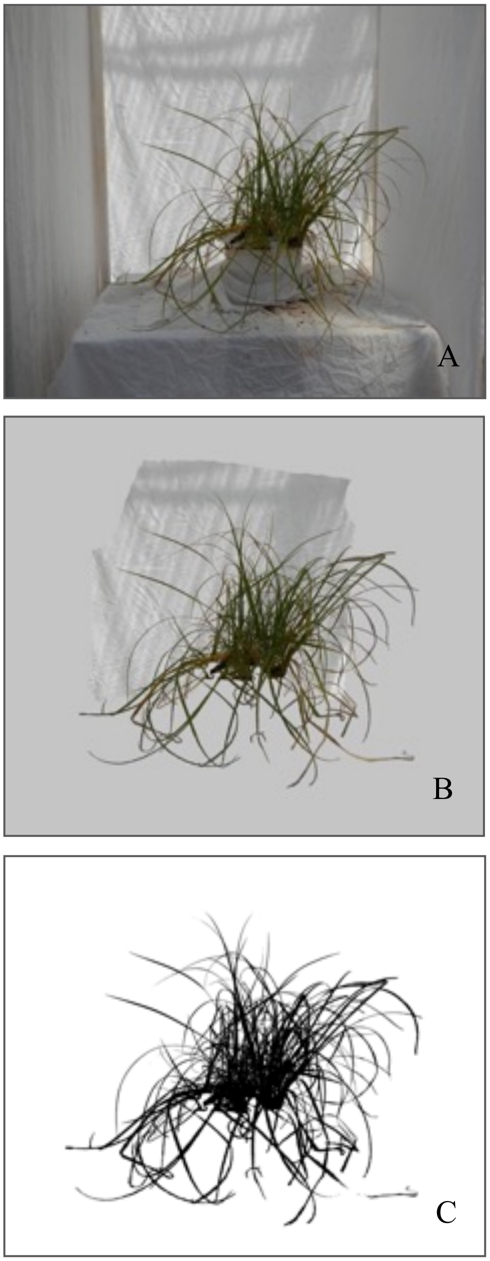


Figure 5. Linear model with a fitted regression line representing aboveground biomass of individual *C. hyalinolepis* plants dried to the gram compared to the total number of pixels making up the leaves in an image of the corresponding live individual *C. hyalinolepis* plants.

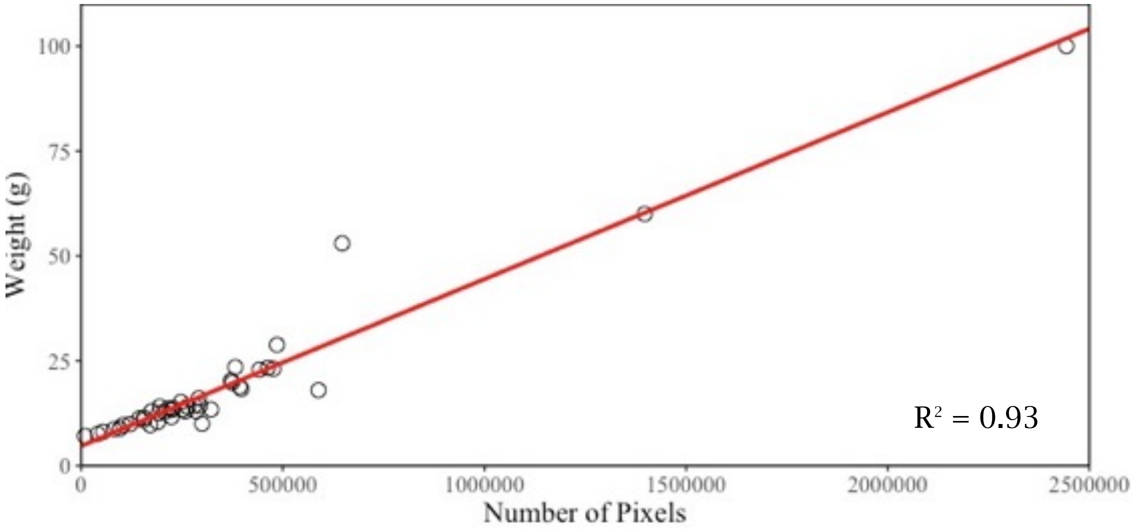


Figure 6. Change in mean number of shoots (±SE) produced per pot after *C. hyalinolepis* plants were clipped at the start of the experiment and again at week 2. Plants were clipped to the soil level (0 cm) and 1 cm, 5 cm, or 10 cm above soil level. *C. hyalinolepis* plants were monitored for one month post treatment.

Figure 7. Change in mean number of *C. hyalinolepis* shoots (±SE) produced per pot after plants were burned once or burned twice two weeks apart. *C. hyalinolepis* plants were monitored for one month post treatment.

Figure 8. Change in mean number of shoots (±SE) produced per pot after *C. hyalinolepis* plants were burned and flooded for one month or burned and flooded for two months. *C. hyalinolepis* plants were monitored for one month post treatment.

Figure 9. Change in mean shoot number (±SE) for *C. hyalinolepis* plants flooded for one month. *C. hyalinolepis* plants were monitored for one month post treatment.

Figure 10. Change in mean shoot number (±SE) for *C. hyalinolepis* plants flooded for two months. *C. hyalinolepis* plants were monitored for one month post treatment.

Figure 11. Change in mean shoot number (±SE) for *C. hyalinolepis* plants flooded for four months. *C. hyalinolepis* plants were monitored for one month post treatment.

Figure 12. Change in mean shoot number (±SE) for *C. hyalinolepis* plants flooded for six months. *C. hyalinolepis* plants were monitored for one month post treatment.

Figure 13. Change in mean aboveground biomass (±SE) over six months for *C. hyalinolepis* plants under various treatments and represented by pixel number. *C. hyalinolepis* plants were burned once burned and flooded for one month, flooded for one month, and clipped to the soil level. Plants were monitored for six months.

Figure 6

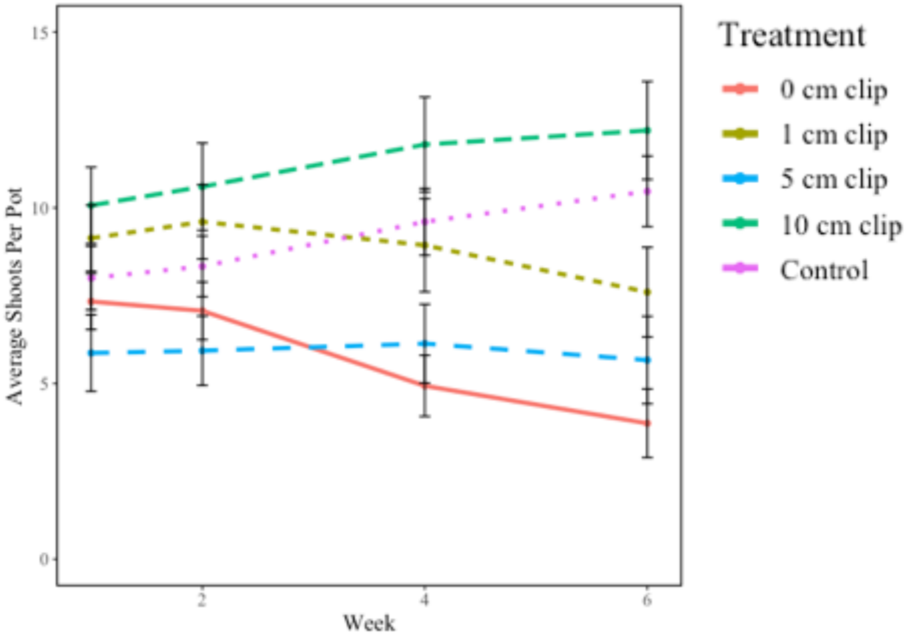


Figure 7

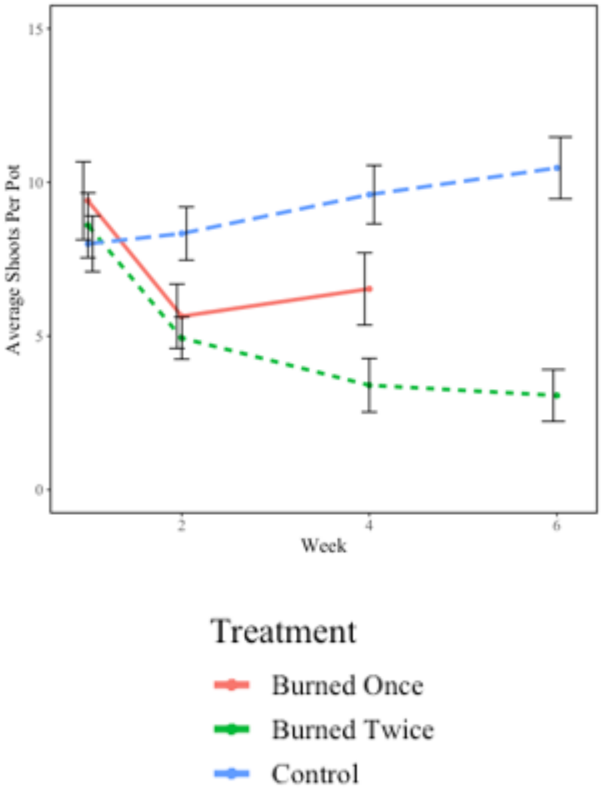


Figure 8

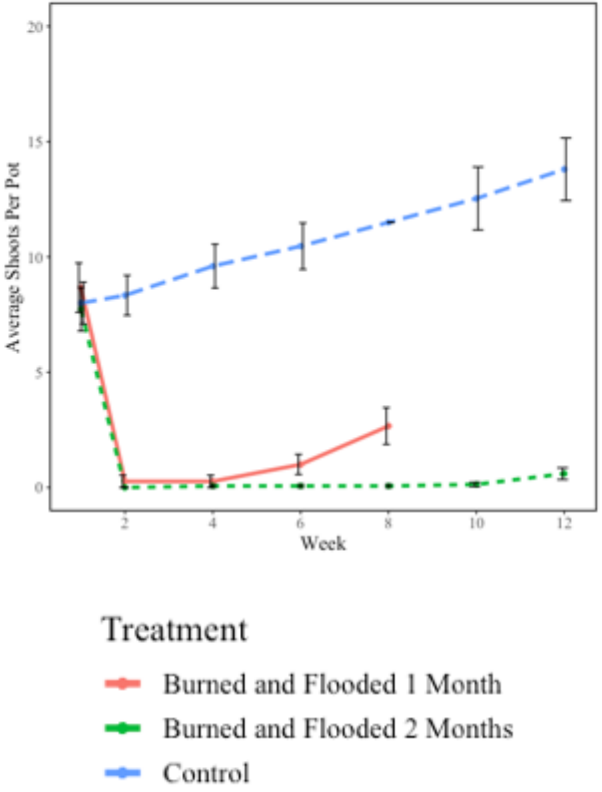


Figure 9

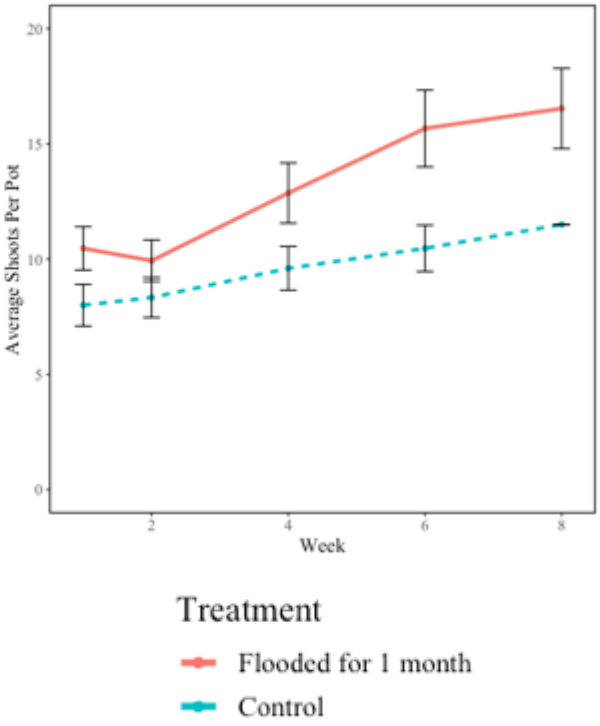


Figure 10

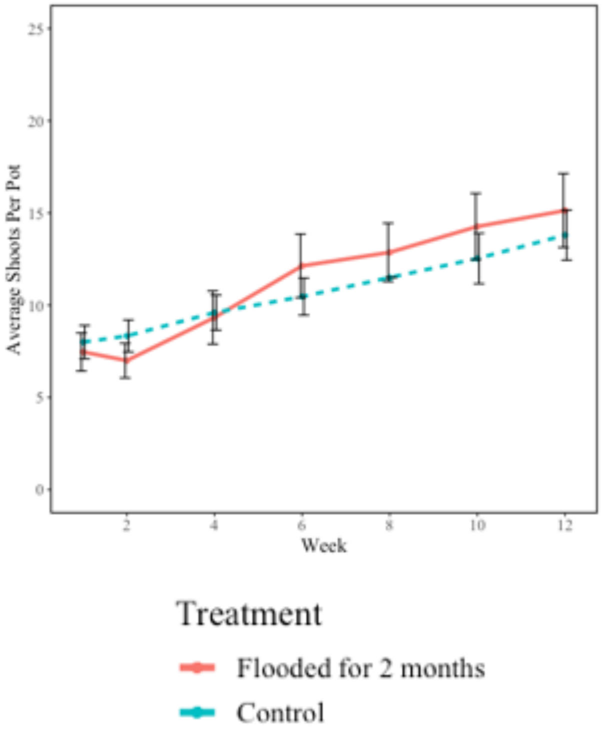


Figure 11

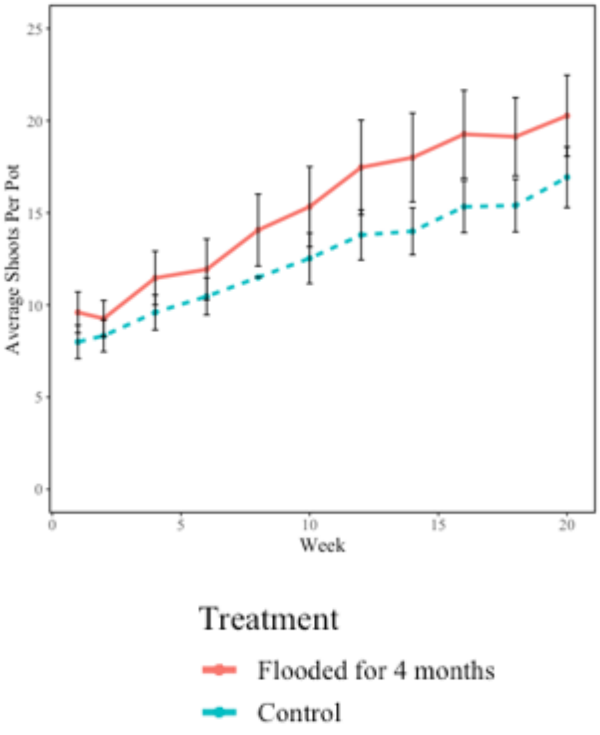


Figure 12

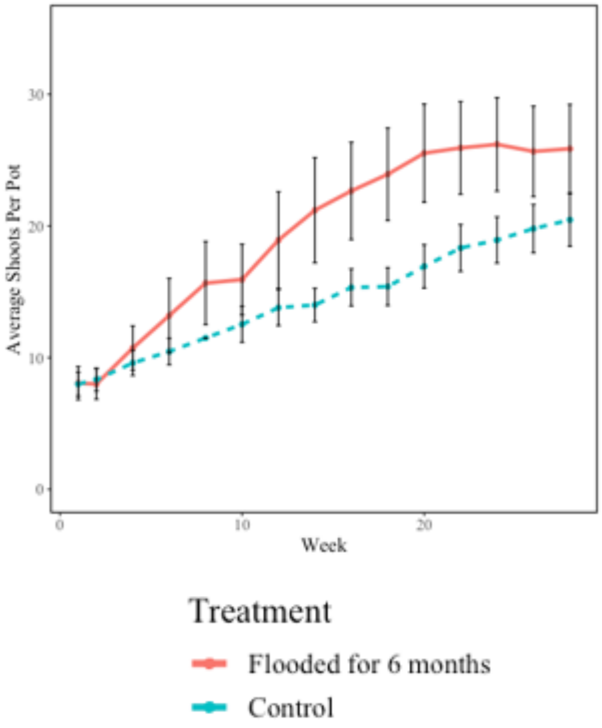
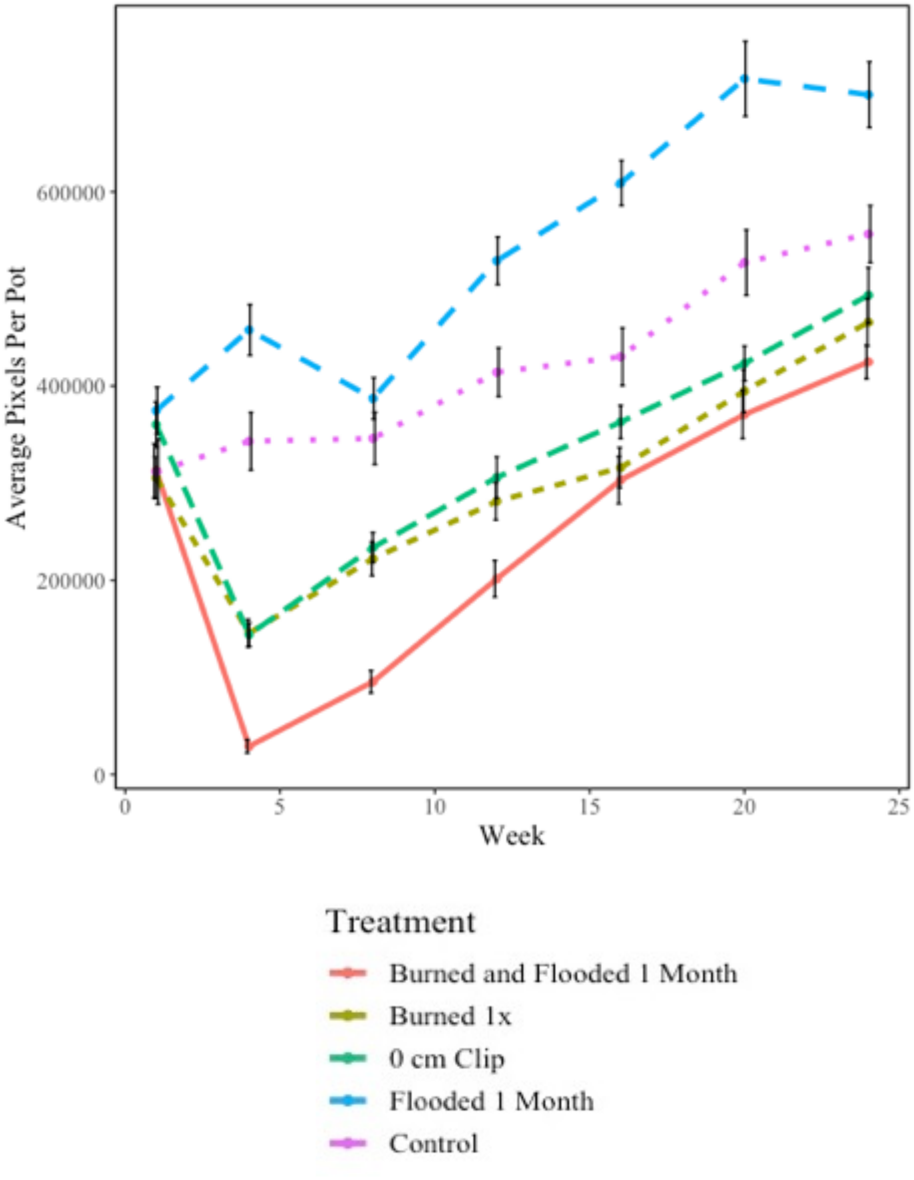


Figure 13



Average Pixels Per Pot
0
200000
400000
600000
0
5
10
15
20
25
Week
Treatment
Burned and Flooded 1 Month
Burned 1x
0 cm Clip
Flooded 1 Month
Control

CHAPTER II

PRESCRIBED FIRE ON NATIONAL WILDLIFE REFUGES: A CASE STUDY ON ARANSAS NATIONAL WILDLIFE REFUGE

ABSTRACT

Southeastern coastal Texas is comprised of fire dependent ecosystems. Fires were common during pre-European times and were historically anthropogenic in origin. In general, the estimated mean fire return interval ranged from an average of 1-15 years, although the accuracy of historical accounts is more qualitative in nature and some local areas may have had substantially longer fire frequencies. We suggest that the accuracy and historical accounts of both wildfire and prescribed fire on the Texas southeastern coast may provide perspective and context on the role fire plays in these ecosystems. Using the normalized burn ratio and differenced normalized burn ratio methods, we mapped prescribed fires on the Aransas National Wildlife Refuge between 1985 and 2013. Results indicate that the refuge is maintaining fire dependent ecosystems with an extensive burn program that includes a fire return interval between 2 and 10 years on a majority of the refuge with some locations having much longer intervals. The northwestern and southern portions of Aransas and most of Matagorda Island burned frequently, while the middle of the refuge burns less frequently. Keeping running live oak (*Quercus virginiana*) from spreading and preserving the openness of the herbaceous wetlands and grasslands is essential for the continued survival of overwintering Whooping Cranes (*Grus americana*) and other wildlife. Quantifying the current fire regime will be useful for future management efforts on Aransas National Wildlife Refuge.

KEY WORDS: Burn regimes, fire dependent ecosystem, mean fire return interval, normalized burn ratio, LANDFIRE, Aransas National Wildlife Refuge

INTRODUCTION

As a historic disturbance mechanism, fire resets succession, can promote plant germination, create a mosaic of habitat types, and provide habitat for fire-adapted species (Stambaugh et al. 2014a, Turner et al. 1997). Fire is important to ecosystem development and maintenance, particularly in sustaining grasslands and prairies (Nelson et al. 2006, Keeley and Rundel 2005, Morgan and Lunt 1999). While fire plays an important role in natural ecosystems, fire suppression has become a dominant paradigm of land management over the past century (Bowman et al. 2011, Dombeck et al. 2004, Keane et al. 2002). In fact, throughout most of the 20th-century, fire suppression resulted in altered fire regimes and excessive accumulation of fuel leading to more extreme and severe wildfires (Ryan et al. 2013, Stambaugh et al. 2014a). Following World War II, the explicit focus on fire suppression was enhanced as an increase in manpower and surplus military equipment allowed additional resources to be used to fight wildfires, and since then, federal agencies have continued to be more reactive rather than proactive when addressing fire (Dale 2006, Dombeck et al. 2004).

The United States Fish and Wildlife Service (USFWS) has been successful in maintaining prescribed fire programs on national wildlife refuges and provides a good case study to examine the impacts of prescribed fires on the landscape. For example, the USFWS has incorporated policies that focus on using prescribed burns to improve wildlife habitat, reduce fuel loads, and maintain fire dependent ecosystems on lands the agency manages. The USFWS burns approximately 121,000 ha annually and more recently, has burned 206,248 ha between 2016 and 2017 (USFWS 2017, 2016). The agency was given clear instruction for developing and implementing management objectives under the National Wildlife Refuge System Improvement Act of 1997 (Dolin 2003, Federal Register 1997). Each refuge is encouraged to manage for historical conditions, including the use of fire when applicable (Meretsky et al. 2006, Schroeder et al. 2004). However, the USFWS faces a challenging conundrum when re-introducing fire to the landscape. For example, does the agency attempt to return the landscape to historic conditions or do they manage the landscape for current objectives that result in vegetation types that are different from what was historically present? Moreover, if restoring the landscape to historical conditions is an objective, how are historic conditions determined, and what date from the past is the target? Understanding the effects of fire on vegetation community development and change is important for wildlife refuges because answering these questions can help guide future management efforts. This is particularly relevant for wildlife refuges that contain critical habitat for endangered species. The need to protect endangered species often guides and prioritizes management activities on refuges. The objective of this case study is to discuss the current fire regime in the context of historical regimes by mapping the current fire regime and comparing it to literature describing fire regimes of the past.

Aransas National Wildlife Refuge (ANWR) provides critical habitat for endangered species, and prescribed burns are an important part of management for wildlife on the refuge. It benefits the refuge to understand the spatial patterns of fires related to vegetation in order to maintain critical habitat. We performed a descriptive analysis of the current fire regime of ANWR using remote sensing and GIS techniques. We also examined accounts of historic burn regimes (LANDFIRE, Stambaugh et al. 2014b, Guyette et al. 2012, Frost 1998) in comparison to the current prescribed fire program at ANWR to foster a better understanding of the ecological relationship of prescribed fires to vegetation types on the refuge. Further, we will discuss the implications and challenges of implementing a prescribed fire program at ANWR relative to current and future management objectives of the refuge.

ARANSAS NATIONAL WILDLIFE REFUGE

ANWR is a remnant of a coastal prairie fire-dependent ecosystem and is located along the southeastern coast of Texas, USA, at the junction of Aransas, Calhoun, and Refugio counties (Fig. 1). The refuge is composed of 5 administrative units: Aransas, encompassing 19,126 ha; Tatton, a 3,063 ha unit that serves to protect a remnant of low upland dark soil coastal prairie; Matagorda Island, a barrier island protecting the Texas coast, with the refuge portion of the island comprising 22,939 ha; Myrtle Foester-Whitmire, which encompasses 1,392 ha and provides nesting and overwintering waterfowl habitat; and Lamar, which comprises 396 ha of salt marsh habitat and native coastal woodlands. The refuge is buffered by 5,234 ha of bay area waters (for a more detailed description, see Aransas National Wildlife Refuge Comprehensive Conservation Plan and Environmental Assessment (USFWS 2010)). San Antonio Bay borders the refuge to the northeast and Saint Charles Bay borders the western/northwestern side. Carlos Bay and Mesquite Bay are part of the Gulf Coast Intracoastal Waterway between Aransas and Matagorda Island. The eastern portion of Matagorda Island abuts Espiritu Santo Bay to the north, while the Gulf of Mexico is located to the south of the island. Aransas is comprised of Galveston-Mustang-Dianola soil associations, while Matagorda Island is primarily comprised of sand soils of the Galveston-Adamsville soil types. The Myrtle Foester-Whitmire Unit is comprised of the Livia-Francitas soil association (USFWS 2010, Guckian 1988).

The refuge is comprised of fire-adapted vegetation communities. On the main portion of the refuge (Aransas Unit), the most prevalent vegetation type is the Texas coastal bend live oak-redbay forest that includes Texas live oak (*Quercus fusiformis* Small), live oak (*Quercus virginiana* Mill), redbay (*Persea borbonia* (L.) Spreng.), Darlington's oak (*Quercus hemisphaerica* Bartram ex Willd.), yaupon (*Ilex vomitoria* Aiton), and American beautyberry (*Callicarpa americana* L.) (USFWS 2010). The live-oak redbay forest is also associated with grassland patches that tend to include seaside bluestem (*Schizachyrium scoparium var. littorale* (Nash) Gould), brownseed paspalum (*Paspalum plicatulum* Michx.), big bluestem (*Andropogon gerardii* Vitman), Indiangrass (*Sorghastrum nutans* (L.) Nash), whip nutrush (*Scleria triglomerata* Michx.), and gulfdune paspalum (*Paspalum monostachyum* Vasey). The second predominant vegetation type is the Texas coastal bend interdune swale grassland, which is adapted to fire and periodic flooding. Plant species affiliated with this community consist of saltmeadow cordgrass (*Spartina patens* (Aiton) Muhl.), switchgrass (*Panicum virgatum* L.), Carolina fimbry (*Fimbristylis caroliniana* (Lam.) Fernald), saltmarsh fimbristylis

(*Fimbristylis spadicea* (L.) Vahl), and largeleaf pennywort (*Hydrocotyle bonariensis* Comm. Ex Lam.). Coastal prairie and salt/brackish tidal marsh compose portions near the Intracoastal Waterway.

HISTORIC FIRE REGIME

LANDFIRE

We used the models developed by LANDFIRE to examine the historic burn regime on ANWR. LANDFIRE, the Landscape Fire and Resource Management Tools Project, was developed by the US Forest Service and the Department of the Interior to manage wildland fire-related fuels and vegetation with a reference era based on pre-European conditions (Reeves et al. 2009). Thirteen wildland fuel layers designed to be nationally consistent across the conterminous United States were developed based on GIS layers, satellite imagery, and biophysical inventories from georeferenced field data (Ryan and Opperman 2013, Rollins 2009). LANDFIRE uses a 30 meter-grid spatial resolution, which matches the resolution of the Landsat imagery used in the mapping the fires on ANWR. We accessed the Mean Fire Return Interval (MFRI) and Fire Regime Group (FRG) GIS layers to compare historical burns on the ANWR to the current fire regime. The MFRI represents the time between fires based on the assumed historic fire regime and the data are assigned to 1 of 22 classes ranging from 0-5 (continuous to frequently burned) years to >1,000 years (USGS 2013). The FRG represents the historic fire regime for a given area and represents 5 classes based on fire return interval and burn severity: Group I is a ≤ 35-year return interval with low and mixed severity fire, Group II is a ≤ 35-year return interval with replacement severity level fire, Group III is a 35-200 year fire return interval with low and mixed severity fire, Group IV is a 35-200 year fire return interval with replacement severity, and Group 5 is a 200+ year fire return interval at any level of severity.

A majority (59.9%) of the refuge is considered to be in Fire Regime Group III according to LANDFIRE (Appendix D, Figures 1 and 2). Group II comprises 36.9 % of the refuge, meaning a 35 year or less fire return interval with replacement severity fire. Group I is found primarily on the edges of the refuge and Matagorda Island, otherwise it appears patchy throughout the landscape. This indicates that the model estimates the refuge historically burned between 35 and 200 years with low and mixed severity fire. The most common mean fire return interval based on LANDFIRE is 81-90 years (59.84% of the refuge), followed by 26-30 years (23.65% of the refuge), with the third most common fire return interval of 0-5 years comprising 10.07% of the refuge. (Appendix D, Figures 3 and 4).

In addition to LANDFIRE, several other studies have attempted to provide an estimate of the historical fire regime of the United States and more specifically, Texas (Table 3). Frost (1998) created an estimate of pre-settlement fire regimes of the United States based on elevation, vegetation, fire scars, and historical accounts. Frost (1998) believed that landscapes with little elevation change, continuous fuel, and little to no natural fire breaks indicated a short fire return interval. Any ignition would be able to burn larger portions of the landscape and the larger the landscape available to burn, the more ignitions were possible. Guyette et al. (2012) took a different approach to analyze the historic fire regime of the United States based on climatic factors. The authors concluded that the shortest fire return intervals were found in places with warm temperatures, and in some cases higher levels of precipitation, where rapid vegetation growth provided continuous fuel.

Research by Stambaugh et al. (2014b) highlights that little quantitative information exists on the fire regime for this region, but qualitative historical accounts indicate that burns occurred quite frequently. Historical accounts suggest that burning was a common practice in Gulf coastal communities and throughout Texas (Sparks et al. 2012, Nyman and Chabrek 1995, Box et al. 1967, Lynch 1941), particularly the Southern Coastal Plain Region (Christensen 2000, Hanselka 1980) and low intensity fires helped shape the presence of open grasslands and savannah like ecosystems in the southeastern United States (Fowler and Konopik 2007, Boyd 1999). The landscape structure also facilitates the spread of fire as there is little change in elevation and large areas where natural fire breaks do not exist, allowing fires to spread over larger areas, thus increasing the likelihood that an ignition in one area has a greater potential to cause more fire as it can spread farther (Frost 1998).

CURRENT FIRE REGIME (1985-2013)

Mapping Prescribed Fires on the Refuge

We used a database of prescribed burns that occurred from 1985 thru 2013 containing the ignition dates and estimated acreage of burns on the refuge (Aransas, Matagorda Island, Tatton and the Myrtle Foester-Whitmire Unit; we include the Tatton Unit as part of the Aransas Unit from this point forward) provided by the USFWS. Using the information from the database, we mapped the boundaries of each burn using remote sensing techniques to create a spatial database containing locations and sizes of fires.

Cloud-free pre- and post-burn Landsat images from the US Geological Survey Earth Resources Observation and Science Center (http://glovis.usgs.gov) were acquired to compute the Normalized Burn Ratio (NBR; discussed below). The images were radiometrically corrected after download and calibrated from raw brightness values to top of atmosphere (TOA) reflectance using the dark object subtraction method, which standardizes bands and corrects for sunlight and topographic illumination errors (Chavez 1988).

Due to the requirement of cloud-free images within a short timeframe before and after the fire for the 28-year time span of this study, we used imagery from both the Landsat Thematic Mapper (TM) and Landsat Enhanced Thematic Mapper Plus (ETM+). Both the TM and ETM+ sensors were designed specifically to capture wavelengths that can aid in differentiating plant and soil moisture (Jensen 2016), which is useful in capturing changes pre and post fire. We used NBR for its ability to discern pre- and post-fire vegetation changes. NBR was developed to detect burn severity based on reflectance values from Landsat Bands 4 (near-infrared) and 7 (short-wave near-infrared). These bands are useful for discerning vegetation and plant moisture conditions (Key and Benson 2006, Brewer et al. 2005, Cocke et al. 2005). Specifically, the decrease in moisture content that occurs in vegetation post-fire due to plant cell death and the cessation of chlorophyll production causes a reduction in absorption of short-wave infrared wavelengths (Rogan and Yool 2001). The Normalized Burn Ratio is defined as:

$$\mathrm{NBR} = \frac{\mathrm{Band\ 4} - \mathrm{Band\ 7}}{\mathrm{Band\ 4} + \mathrm{Band\ 7}}$$

By differencing the NBR values from pre- and post-fire, it is possible to identify areas that have been burned during a specific time period. The differenced normalized burn ratio (dNBR) is defined as:

$$\mathrm{dNBR} = \mathrm{NBR}_{\mathrm{pre-fire}} - \mathrm{NBR}_{\mathrm{post-fire}}$$

The resulting dNBR values range from -2 and 2, with positive values indicating burned areas. We isolated and selected the burned areas, using values >0.15 to classify burned areas as this is value indicates a change in vegetation due to the burns (Key and Benson 2006). We then created maps based on the work of Robertson et al. (2007) describing the spatio-temporal movement of fires that occurred on the refuge over time. To model the spatial and temporal change in fire patterns over 5-year increments, we used the Program R StampR package (Long and Robertson 2018, Robertson et al 2007). Burns were categorized to show areas that were burned during the first time increment (i.e., burns conducted from 1985-1989), burns that were conducted during the second increment but not the first (i.e., 1990-1994), and areas that were burned during both (i.e., 1985-1994) time frames, which highlights areas the refuge burns continuously.

Vegetation Classification

We used the Terrestrial Ecological Systems classification system created by NatureServe and the Missouri Resource Assessment Partnership (MoRAP) to map the vegetation communities present on the refuge (Ludeke et al. 2010a, Ludeke et al. 2010b, NatureServe 2009). National Agriculture Imagery Program data from 2004 and 2005 were used to develop the Terrestrial Ecological Systems classification, but because the images were taken between 2004 and 2005, we could only relate burn data with vegetation data from 2004-2013. Detailed vegetation classes were merged into groups based on dominant plant height and category (i.e., forest vs. shrubland).

A total of 375 burns were mapped on the refuge from 1985-2013. We were unable to map 106 burns due to the inability to obtain cloud free digital images for analysis. The quick vegetation regrowth post-fire in the sub-tropical climate of south Texas means accessing data more than three months post-fire would likely not indicate a burn had occurred (Lonard et al. 2004). Wildfire may have occurred on the refuge complex, but only the date of one wildfire was known and was not included in the results. On average, the refuge conducts 13 prescribed burns per year with an average burn size of 183 ha. The most common burn sizes were those that ranged between 81-202 ha (37.1% of all burns), followed by burns between 203-405 ha (26.1%) (Table 1).

According to the ANWR Comprehensive Conservation Plan, most burns should occur during the winter (December-February) and summer months (June-August). Between 1985 and 2013, 45.6% (n=171) of the burns were conducted during the winter months and 20.0% (n=75) occurred during the summer months. The remainder of burns occurred during the fall (26.13%, n = 98; September-November) and spring (8.27%, n=31; March–May).

Large portions of the refuge were burned more than once between 1985 and 2013 (Figures 2 and 3), resulting in high mean fire return intervals. Aransas has developed administrative burn units, which we used in conjunction with the mapped burns, to develop average fire return intervals for each unit. We were able to account for 58 of the missing burns by using old notes and descriptions from refuge staff to make an estimate about the location or the administrative unit where the burn occurred on the refuge.

However, our estimate of the fire return interval is still slightly conservative. All of the administrative burn units designated by the refuge on Matagorda Island are burned at an average rate of 4.8 years between burns, with a range of 1.8–14 years. The mean Aransas fire return interval of units that are burned is 7.5 years, with a range of 1.3-28 years (Figures 4 and 5). There are sections of the Aransas Unit that were not burned during the prescribed fire program and likely have a much longer fire return interval (greater than 28 years). Most of the Myrtle Foester-Whitmire Unit has been burned in recent years. The refuge did not acquire the land designated as the Myrtle Foester-Whitmire Unit until 1993 and did not start burning on the unit until 2004.

Fire rotation is an important component of understanding a fire regime and can help determine available biomass and species composition of an area. The fire rotation for the refuge is calculated as:

$$\text{Fire rotation} = \frac{\text{numbers of years}}{(\text{area burned/total area})}$$

We included the full time span of the prescribed burn plan on the refuge using all units, even though some were not burned until well into the program. The fire rotation indicates that the entire land area of the refuge burns every 13.9 years. We also mapped burns in five year increments to show the locations where the refuge burns repeatedly. Figures 6 A-E show where the refuge burned during the previous five years (e.g., burns from 1985-1989 that did not overlap areas from 1990-1994), areas during the second five years that were burned (e.g., burns from 1990-1994 that did not overlap areas from 1985-1989) and the overlap between the two time frames. The maps show that the refuge consistently burns the same locations. The burns largely occurred on the Tatton Unit, the southern coast of Aransas and the majority of Matagorda Island, which is consistent with the refuge's management goals of reducing woody plant stature and dominance in the coastal prairies. The vegetation types that were burned repeatedly were the grasslands and shrublands. The vegetation types that were burned the most included coastal and sandsheet deep-sand grassland swale marsh, coastal sea ox-eye daisy flats, and coastal salt and brackish low tidal marsh, (Table 2). Between 2004 and 2013, the vegetation types that were burned the least or were not burned at all included coastal and sandsheet deep sand live oak shrubland, coastal and sandsheet deep sand live oak forest and woodland, and coastal and sandsheet deep sand grassland.

The mean fire return interval of Guyette et al. (2012), Stambaugh et al. (2014b), and Frost (1998) all suggest that the average time between burns on the Texas gulf coast varied between one and twelve years, which is more often than the average suggested by LANDFIRE, indicating that burns historically occurred at a much higher frequency. ANWR therefore likely has a historically higher fire return interval than what LANDFIRE indicates. While the exact historical fire frequency estimates vary between these sources, the overall trend is that the Gulf Coast of Texas was a fire-dependent ecosystem. Variation in vegetation will inherently exist on smaller scales across the landscape and will affect fire frequency. Additional factors support the more frequent fire return interval. For example, the warm climate of the Texas Gulf Coast contributes to rapid vegetation regrowth and fuel replenishment that is conducive to the increased presence of fire on the landscape (Guyette et al. 2012).

There is considerable vegetative variability on ANWR, but what makes the refuge unique is the landscape structure and climate which support the ability to have a frequent fire return interval with a prescribed burn program. Aransas relies almost entirely on human ignition to maintain fire on the landscape. ANWR continues to implement a prescribed fire program that mimics the frequency of burns in some historic accounts but may be higher than others. However, returning the landscape to prior conditions based on historic fire regimes may not be in line with management needs on the refuge.

OBJECTIVE-BASED MANAGEMENT FOR ENDANGERED SPECIES

A critical objective of management on ANWR is to maintain the appropriate extent of oak communities, open grassland, and wetlands for federally-listed endangered species such as the whooping crane (*Grus americana*) and Aplomado falcon (*Falco femoralis*). Whooping cranes and Aplomado falcons have specific habitat requirements that fire helps maintain. Cranes tend to avoid the areas of dense live oak woodland and shrubland. Continuous prescribed burns help set vegetation back to early successional stages and lower overall plant height as cranes prefer visual acuity and field of view below 1.4 meters to allow for predator detection (Armbruster 1990). The whooping crane prefers unobstructed visual areas to roost, feed, and nest (Timoney 1999) as do Aplomado falcons for hunting (Macías-Duarte et al. 2004, Perez et al. 1996).

Winter burns are conducted at ANWR to create additional food resources for the whooping cranes because burned locations provide areas for opportunistic feeding due to removal of vegetation that make prey easier to spot (Venne and Frederick 2013, Chavez-Ramirez 1996). Blue crabs (*Callinectes sapidus*) and clams are high in crude protein and constitute a large proportion of whooping crane diets by volume, but when population levels of these species are low, there is an increased risk of mortality for the cranes (Pugesek et al. 2013, Nelson et al. 1996, Hunt and Slack 1989). The upland winter burns provide access to small reptiles, amphibians, and acorns, which may be particularly important during times of low crab availability (Chavez-Ramirez et al. 1996, Hunt and Slack 1989). Specific objectives can be met with prescribed fire such as maintaining the grasslands and prairie habitat as well as the coastal savannah habitat that benefits not only whooping cranes and Aplomado falcons, but also National Wildlife Refuge migratory bird program focal species such loggerhead shrikes (*Lanius ludovicianus*), seaside sparrows (*Ammodramus maritimus*) and painted buntings (*Passerina ciris*), which inhabit and rely on habitat similar to whooping cranes (USFWS 2010).

Certain ecosystems may be more susceptible to woody plant encroachment due to climatic conditions that promote invasion. ANWR may be more susceptible to woody invasion due to rainfall and overall productivity levels because it is located in a subtropical climate (Archer et al. 2017, Ratajczak et al. 2012) and therefore, the refuge must consider encroaching woody species as a potential problem that prescribed fire can help ameliorate. Early descriptions of the vegetation on the refuge indicated that 75% of the refuge was covered in live oak and the remainder was either grassland or salt marsh near the shore (Hanselka 1980, Lehman 1965). The encroachment of woody species can increase or decrease the existing fuel load and create a feedback loop that results in a changed fire regime that can alter the ecosystem permanently (Zouhar et al. 2008, Brooks et al. 2004).

Shortly after the establishment of the refuge in the late 1930s, cattlemen noted that burns were needed to reduce and prevent brush buildup on the refuge (Halloran 1943). Summer burns are conducted to help combat woody plant encroachment from live oak and other species with invasive characteristics such as the camphor tree (*Cinnamomum camphora* (L.) J. Presl), Chinaberry (*Melia azedarach* L.), Chinese tallow

(*Triadica sebifera* (L.) Small), saltcedar (*Tamarix gallica* L.), and mesquite (*Prosopis glandulosa* Torr.) on marsh habitat (USFWS 2010). A primary habitat management goal on the refuge is to limit live oak, which has been described as an aggressive native species that has invasive characteristics that can alter the vegetation composition of the refuge (USFWS 2010). Studies indicated that burning can top kill live oak, but stem densities did not change or increase in response to cool season burns and repeated summer burns show a decrease in stem densities and openness of the thickets (Hays 1999, Kelley 1980). Woody plant encroachment on coastal grasslands is a global problem and a potential problem for the conservation of whooping crane and Aplomado falcon habitat. There is a need to understand the role of fire in the ecosystem to guide future maintenance and restoration efforts to the landscape.

Many wildlife refuges must decide between managing habitat with the goal of restoring the landscape to previous historic conditions or managing for specific wildlife species. Mimicking a historic fire regime may be an irrelevant concept when more than one management goal needs to be met. Other wildlife refuges likely face the same competing objectives of maintaining the historic fire regime while meeting refuge specific management goals or actively managing for endangered species dependent on fire prone habitats. Examples include the key deer (*Odocoileus virginianus clavium*) (Carlson et al. 1993) and the red-cockaded woodpecker (*Picoides borealis*) (Wilson et al. 1995). One example of choosing modern management actions over historic use includes using refuge lands for corn production to provide adequate food reserves for the overwintering Rocky Mountain population of greater sandhill (*Grus canadensis tabida*) cranes rather than restoring the land to pre-European conditions (Schroeder et al. 2004).

The risk of following the historic burn regime according to LANDFIRE would likely create a scenario of increased woody plant encroachment, including from oak species along the northern part of ANWR and mangroves along the coast. The loss of open estuarine marsh habitat would be detrimental to the cranes. The need to manage habitat for critically endangered and threatened species should take precedence over returning the landscape to historic conditions. The loss of biodiversity is a larger ecological concern if species were to go extinct, therefore maintenance of biodiversity should be a more important primary objective than historic restoration, particularly on refuges with critical habitat designations.

The decision to recreate the historic burn regime is confounded when different sources provide different frequencies. The burn frequency on Aransas is much shorter than LANDFIRE indicates. The historic description of fire in coastal Texas is better represented by Guyette et al. (2012) and Stambaugh et al. (2014b). LANDFIRE is a tool designed by and for federal agencies, which the USFWS may not wish to rely on in the case of the endangered whooping cranes. Because Aransas is required to conserve habitat that the whooping crane population depends on, they burn a large portion of the refuge at a much shorter frequency of 1-4 years than the historic fire return intervals of 80–90 years suggested by LANDFIRE. The cranes have acclimated to the shorter burn frequency and it may be more detrimental to return to the natural historical burn regime. The historic frequency suggested by LANDFIRE could lead to woody plant encroachment which is habitat the cranes avoid (Faanes 1992).

Overall, the burn regime on ANWR is different from what is indicated by LANDFIRE’s historical view of natural fire since it does not take into account anthropogenically induced prescribed burns that have been occurring for many years on the land where the refuge was established. While it is important to take the historical natural fire regime into consideration (i.e., fires caused by lightning), focusing on the management needs of each individual refuge is more important in the long run. The short fire return interval benefits endangered species that use habitat on ANWR and therefore should be continued to ensure habitat is available for vulnerable wildlife species in the future.

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TABLES AND FIGURES

Table 1. Record of fires on Aransas National Wildlife Refuge 1985-2013.

Years	No. of fires	Sum of actual burned area (ha)	Avg. size of fires (ha)	Min. mapped fire size (ha)	Max. mapped fire size (ha)	Sum of burned area according to USFWS estimate (ha)	Avg. size of fires according to USFWS estimate (ha)	No. of fires not mapped	Sum of area burned by USFWS estimate (not mapped)	Average size of fires by USFWS estimate (not mapped)
1985-1989	24	8,994.95	374.79	9.34	1,264.70	8,380.76	349.20	9	1,296.61	144.07
1990-1994	50	7,816.44	156.33	8.15	863.22	8,441.01	168.82	19	2,273.08	119.64
1995-1999	98	16,344.04	166.78	2.05	1,024.79	21,488.44	219.27	49	10,196.01	214.03
2000-2004	60	9,347.95	157.30	5.52	398.36	12,827.81	213.80	11	2,783.83	253.07
2005-2009	93	15,161.95	163.03	7.74	669.72	21,733.64	233.69	15	3,026.24	201.75
2010-2013	50	10,907.57	218.15	5.08	1,356.17	16,433.07	328.66	3	1,156.19	385.38
1985-2013	375	68,662.95	183.10	2.05	1,356.17	89,308.73	238.15	106	21,023.34	198.33

Table 2. The vegetation types present on Aransas National Wildlife Refuge and the percentage of hectares of each type burned by prescribed fires between 2004 and 2013.

Group	Vegetation Communities	Total Hectares Present on ANWR	Total Hectares Burned Between 2004-2013
Grassland	Coastal sandsheet: deep sand grassland, Coastal bend: riparian grassland, Gulfcoast: coastal prairie, Gulfcoast: coastal prairie pondshore, Gulfcoast: salty prairie	13,037.43	9,090.22
Forest/ Woodland	Coastal and sandsheet: deep sand live oak mesquite woodland, Coastal and sandsheet: deep sand live oak forest and woodland, Coastal bend: floodplain hardwood forest, Coastal bend: floodplain live oak forest, Coastal bend: floodplain live oak hardwood forest, Native invasive: deciduous woodland, Non-native invasive: Chinese tallow forest, woodland, or shrubland	3,514.45	680.692
Wetland/ Marsh	Coastal and sandsheet: deep sand grassland swale marsh, Coastal bend: floodplain herbaceous wetland, , Coastal: salt and brackish high tidal marsh, Coastal: salt and brackish low tidal marsh, Coastal: sea ox-eye daisy flats, Coastal: tidal flats, Native invasive: common reed, South Texas: algal flats	5,856.35	1,624.58
Shrubland	Coastal and sandsheet: deep sand live oak shrubland, Coastal and sandsheet: deep sand live oak swale marsh, Coastal and sandsheet: deep sand shrubland, Coastal bend: riparian evergreen shrubland, Coastal: salt and brackish high tidal shrub wetland, Coastal: mangrove shrubland, Gulf coast: salty prairie shrubland, Native invasive: baccharis shrubland, Native invasive: mesquite shrubland, Non-native invasive: salt cedar shrubland, South Texas: clayey mesquite mixed shrubland	10,065.61	4,628.39

Table 3. Comparison of fire regime studies of the Texas Gulf Coast Region.

Study	Fire Return Interval (years)	Scale
This Study	1.3-9.3	Fine (30 m^2)
Frost (1998)	1-3	Broad (U.S.)
Stambaugh et al. (2014b)	1-12	Broad (Texas)
Guyette eta l. (2012)	2.01-4	Broad (1.2 km^2)
LANDFIRE	35-200	Fine (30m^2)

Figure 1. Location of Aransas National Wildlife Refuge and its five administrative units (Aransas, Tatton, Lamar, Matagorda Island, and Myrtle Foester-Whitmire) along the Texas Gulf Coast.

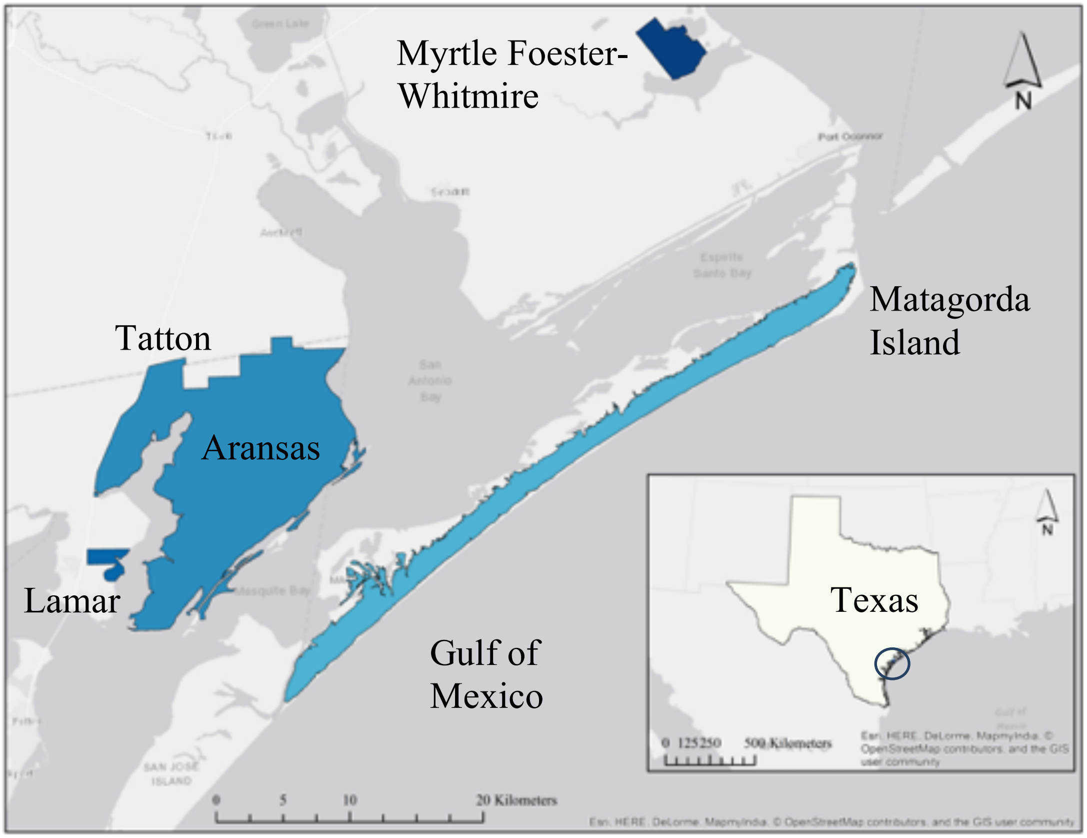


Figure 2. Repeated prescribed fires represented by the number of times an area was burned on Aransas and Lamar Units between 1985 and 2013.

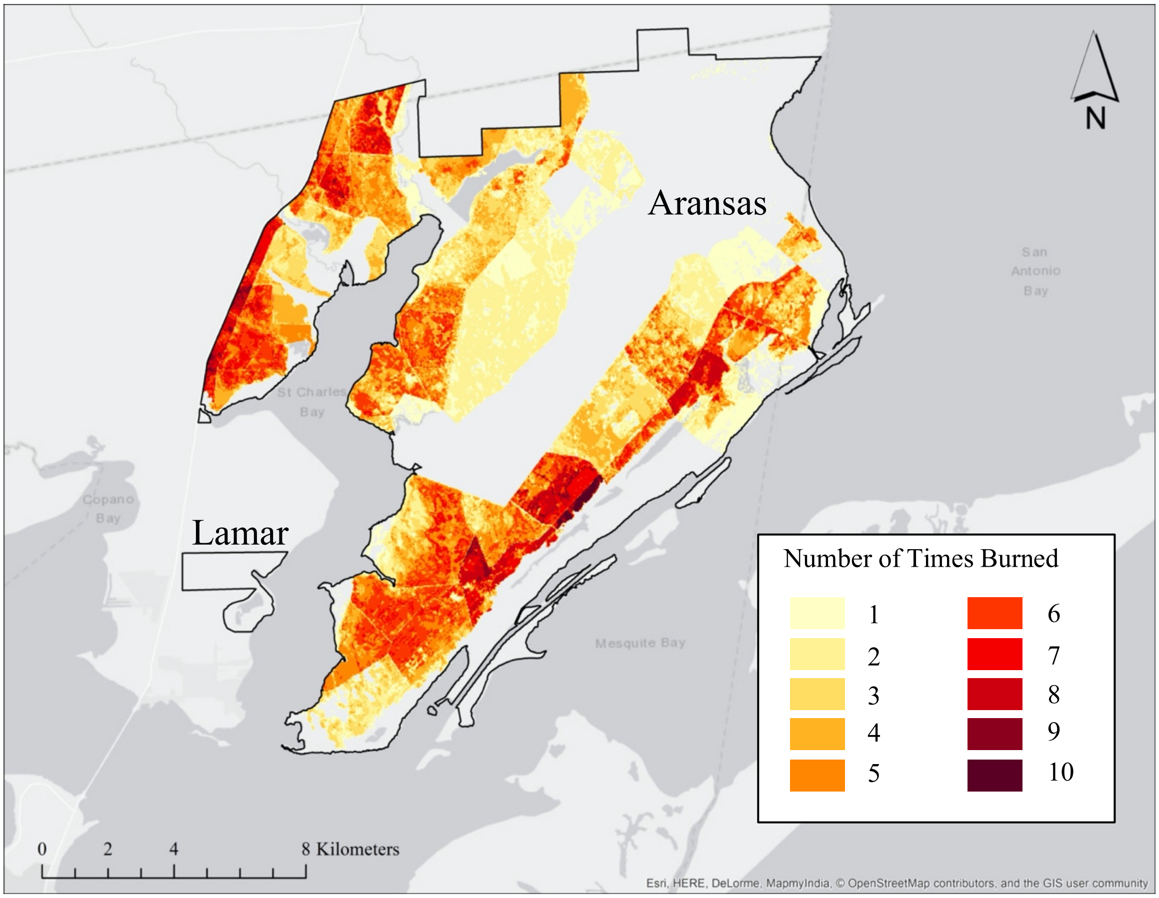
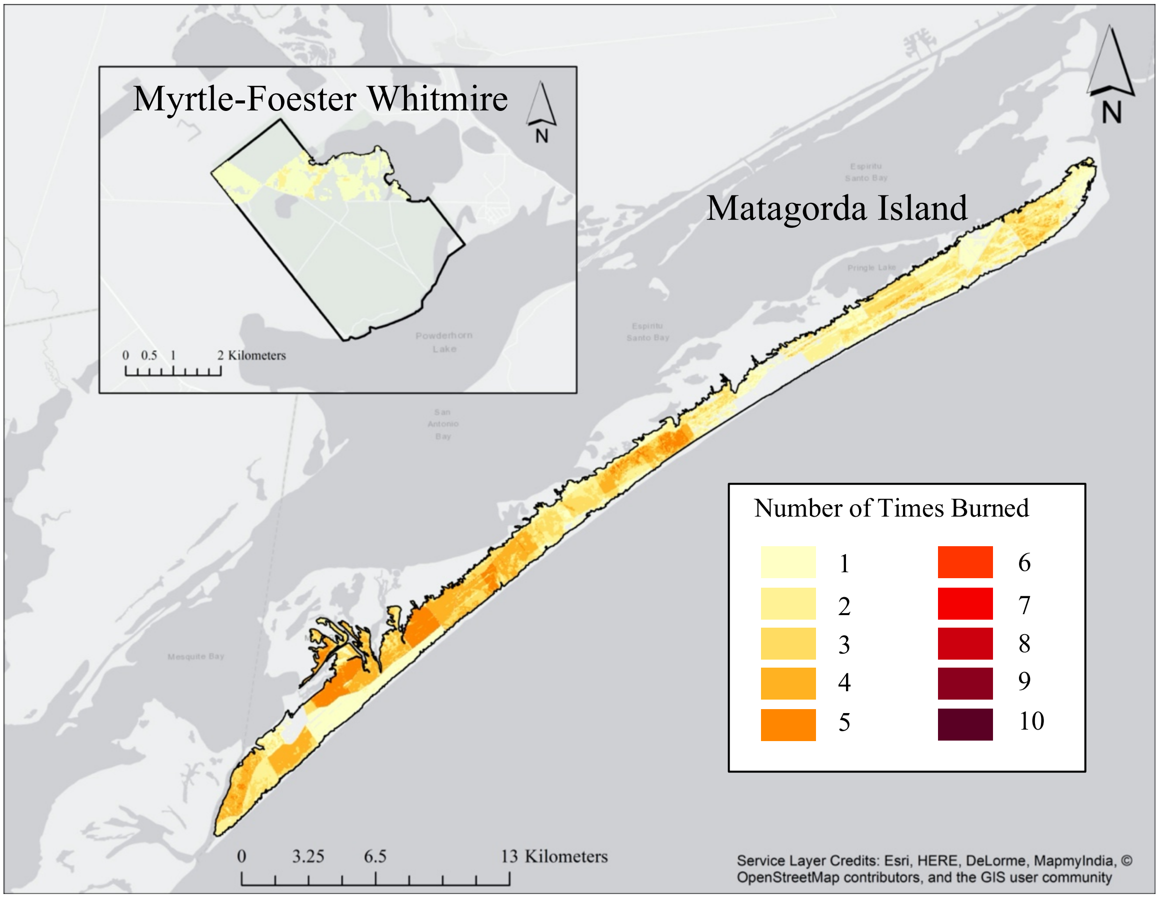


Figure 3. Repeated prescribed fires represented by the number of times an area was burned on Matagorda Island and the Myrtle Foester-Whitmire Unit (inset) between 1985 and 2013.



Service Layer Credits: Esri, HERE, DeLorme, MapmyIndia, © OpenStreetMap contributors, and the GIS user community

Figure 4. Average fire return interval (28 years/number of times burned) per administrative prescribed burn unit on Aransas between 1985-2013. Blue areas with 28.1+ years fire return interval indicate areas where there were no burns during the 1985-2013 time frame.

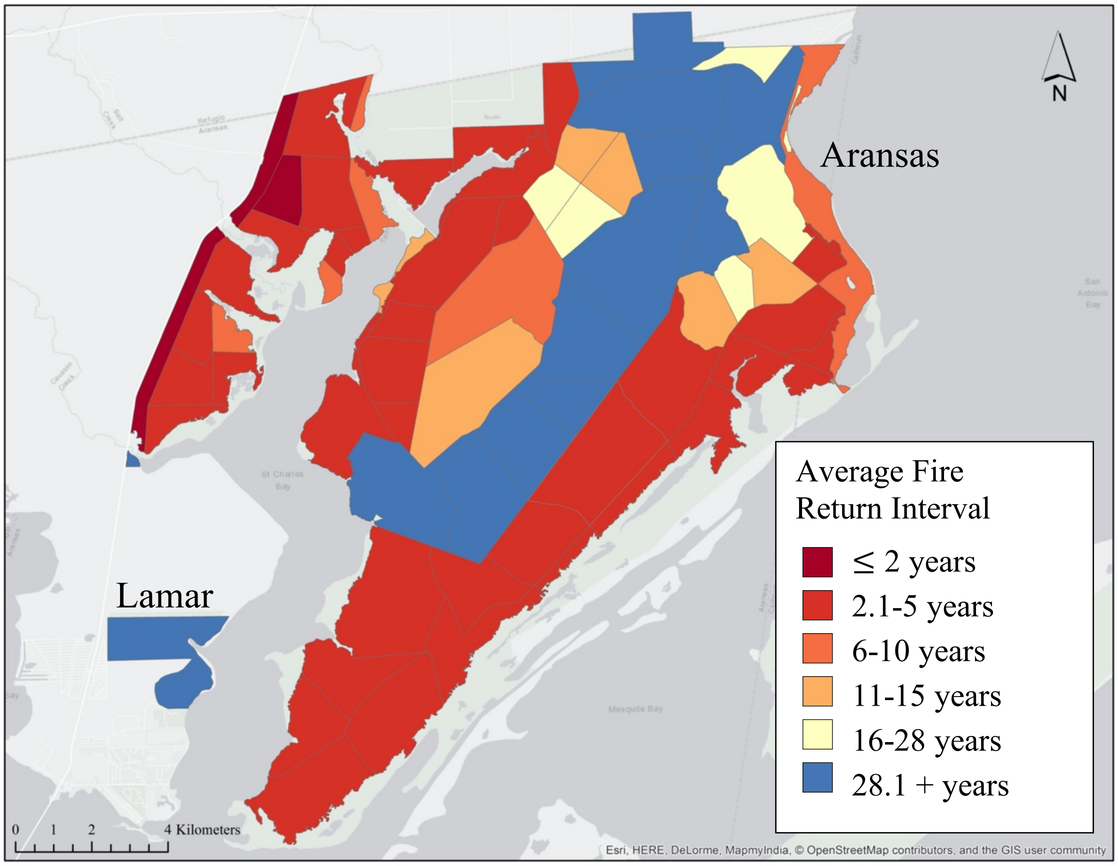


Figure 5. Average fire return interval per administrative prescribed burn unit on Matagorda Island between 1985-2013 (28 years/number of times burned) and the Myrtle-Foester Whitmire Unit between 2004-2013 (9 years/number of times burned), including unmapped burns. Blue areas with 28.1+ years fire return interval indicate areas where there were no burns during the 1985-2013 time frame.

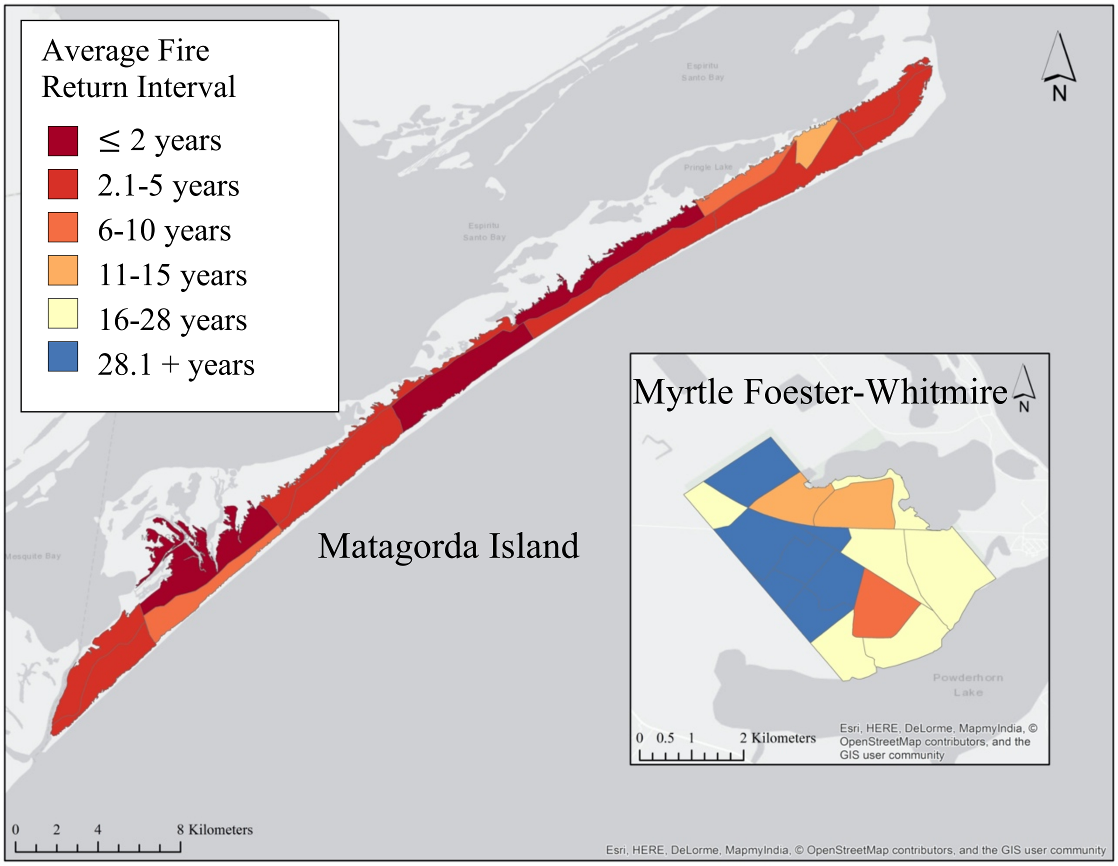
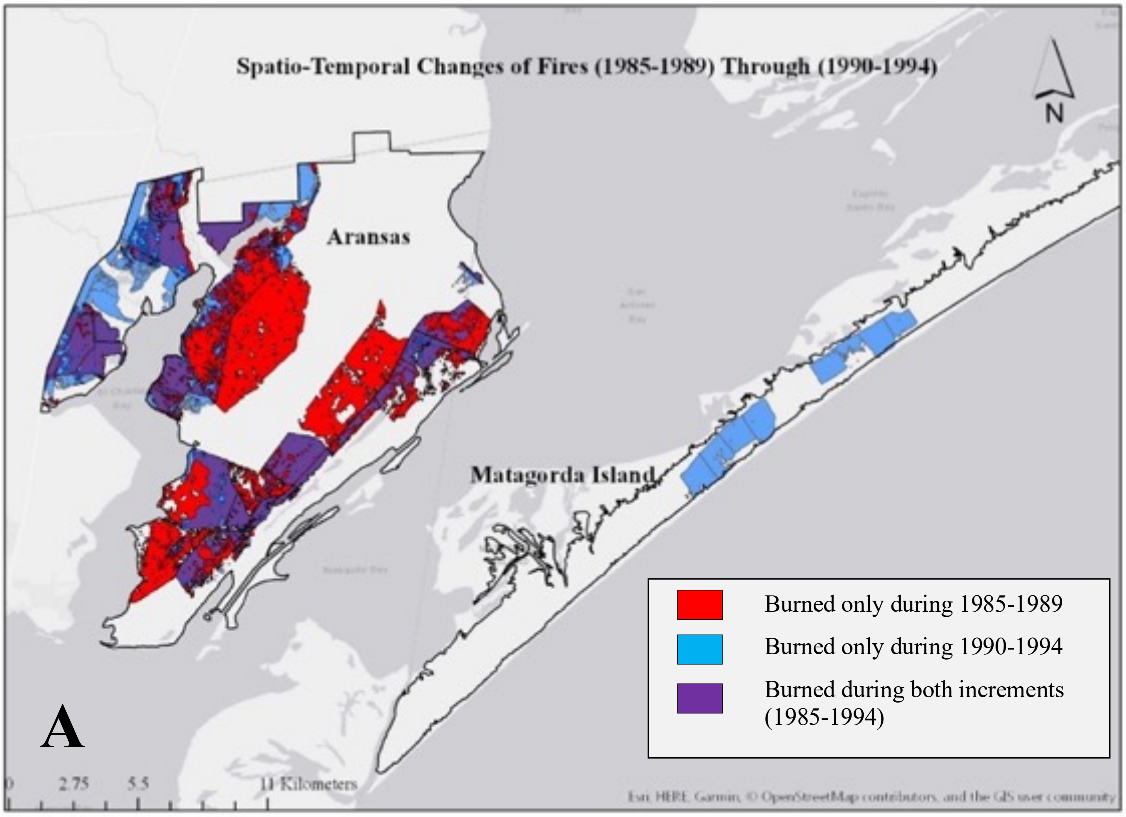
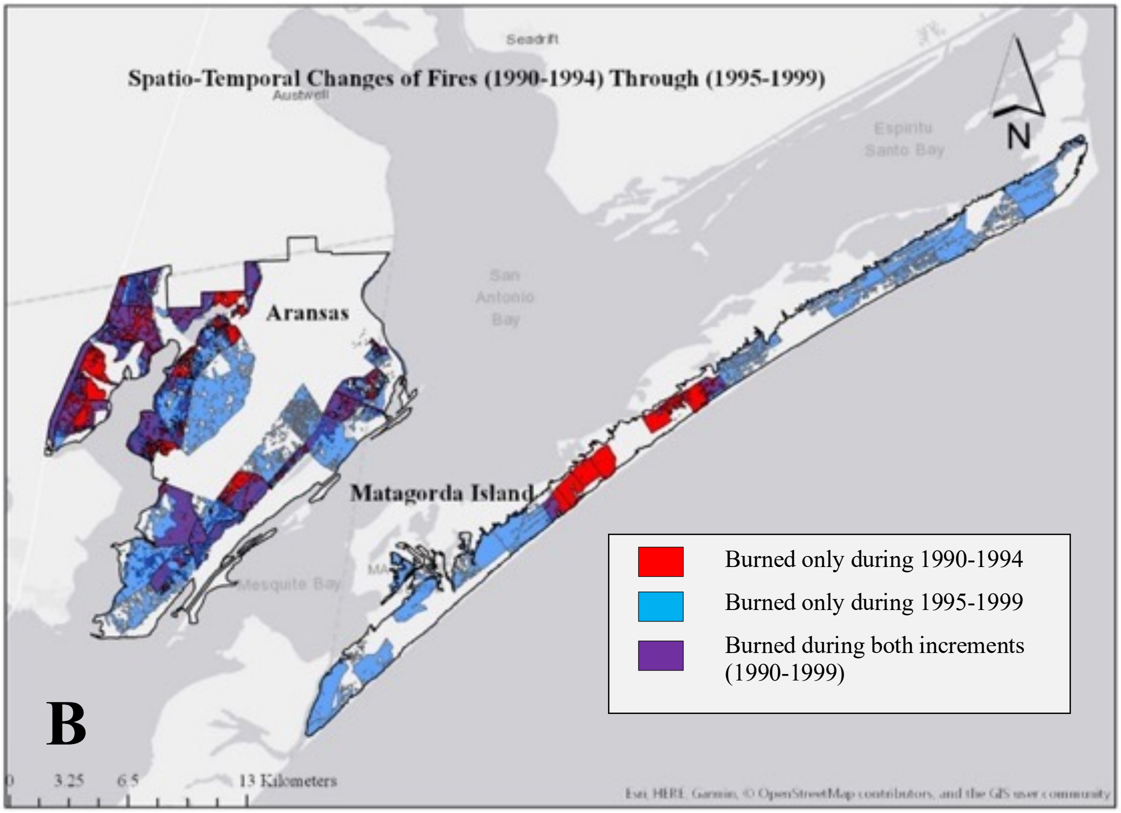


Figure 6. Areas burned and categorized in five year increments which show areas that are burned during the first set of five years, areas that are burned in the second set of five years, and areas that are burned during both time frames. **A**. Spatio-temporal changes of fires (1985-1989) through (1990-1994). **B**. Spatio-temporal changes of fires (1990-1994) through (1995-1999). **C**. Spatio-temporal changes of fires (1995-1999) through (2000-2004). **D**. Spatio-temporal changes of fires (2000-2004) through (2005-2009). **E**. Spatio-temporal changes of fires (2005-2009) through (2010-2013).




Spatio-Temporal Changes of Fires (1990-1994) Through (1995-1999)
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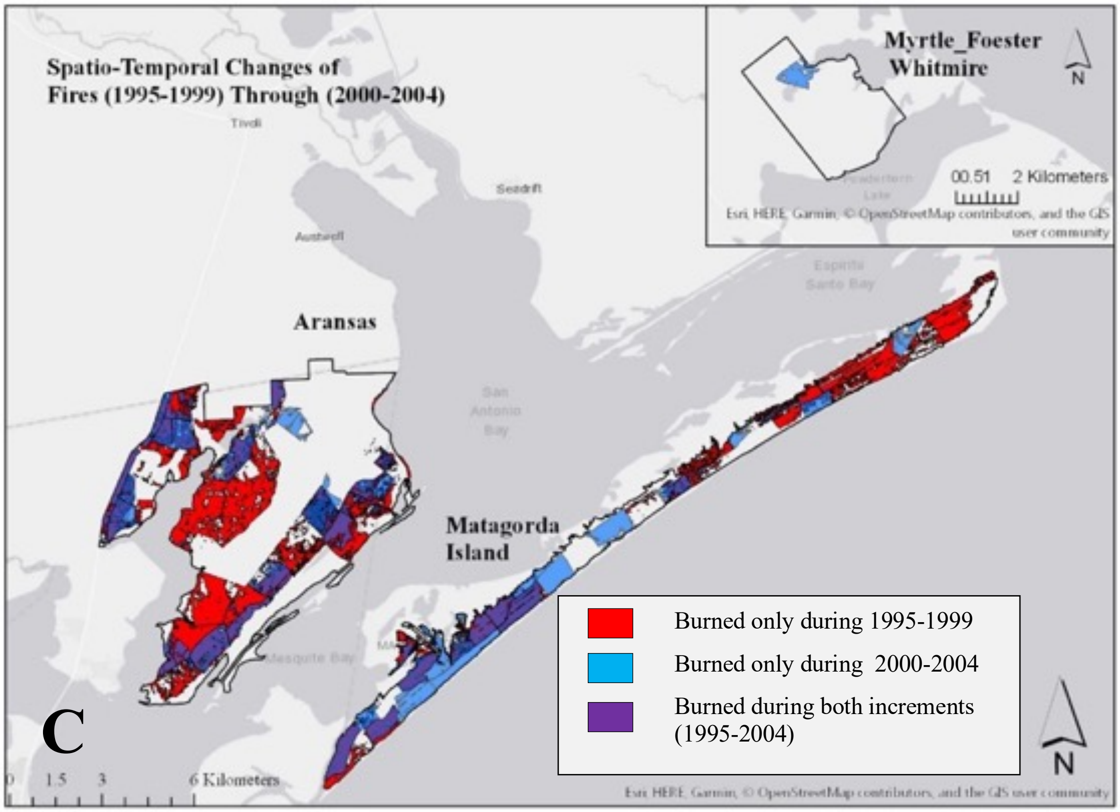
Spatio-Temporal Changes of Fires (1995-1999) Through (2000-2004)
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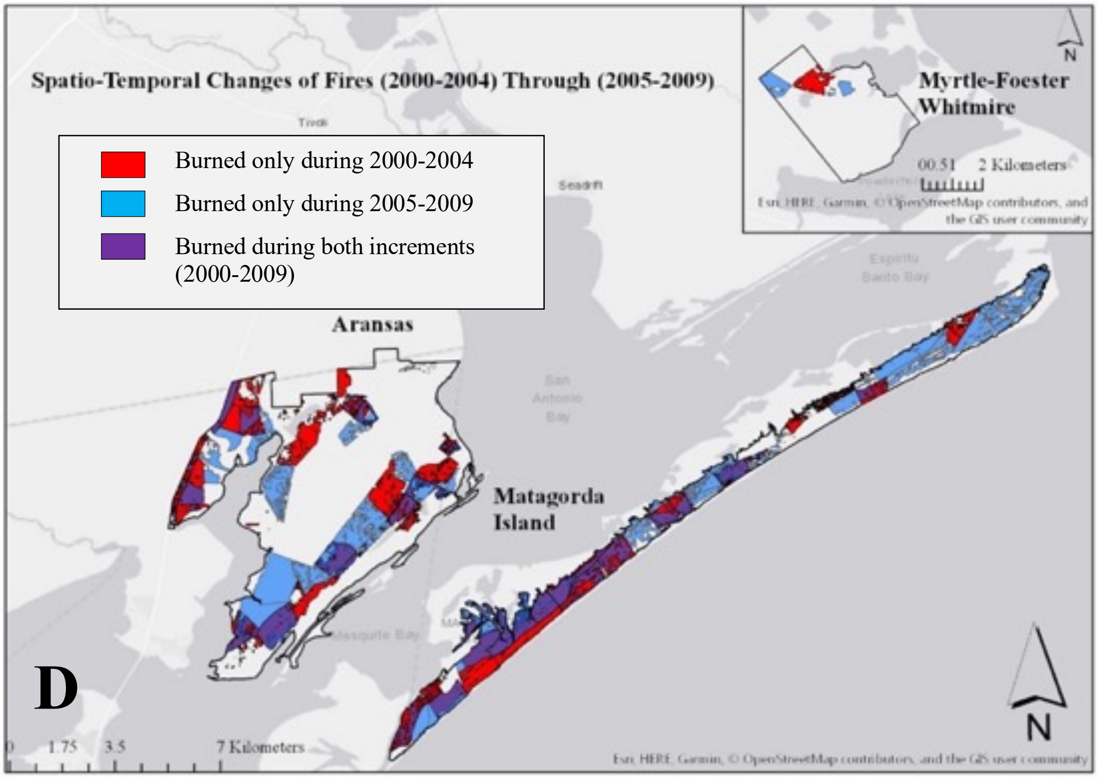
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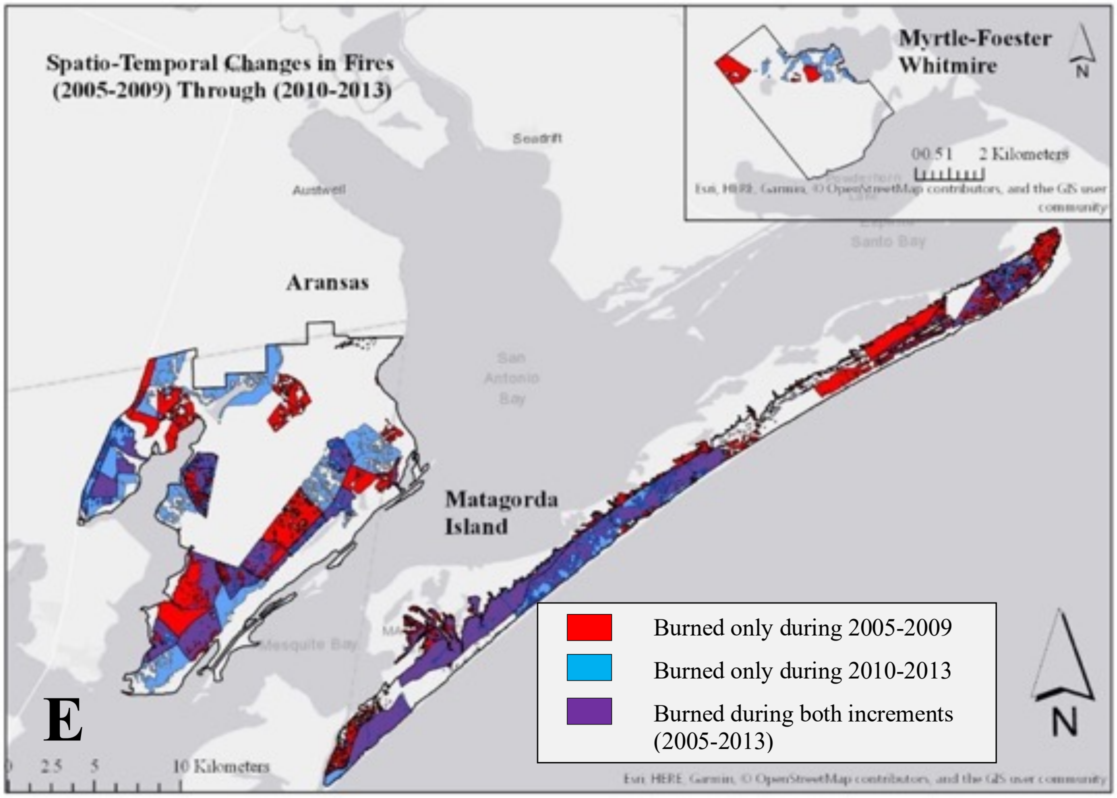
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CHAPTER III

WHOOPING CRANE (*Grus americana*) OVER-WINTERING HABITAT USE AT ARANSAS NATIONAL WILDLIFE REFUGE AND IMPLICATIONS FOR FUTURE CONSERVATION

ABSTRACT

The endangered Aransas-Wood Buffalo whooping crane (*Grus americana*) population overwinters on a limited area along the Gulf Coast of Texas, USA, on the Aransas National Wildlife Refuge. The population is growing and will eventually need additional suitable habitat outside of the current refuge boundaries to achieve the goal of down listing the species from endangered to threatened. The objectives were to determine spatial and temporal patterns in whooping crane occurrences during the winter and to determine the spatial relationship between environmental variables and whooping crane occurrence on Aransas National Wildlife Refuge. This study used sixty years of crane location data coupled with spatial analyses to develop predictive models that forecast crane habitat use based on landcover and refuge management activities as well as determining spatial patterns of the population that could impact future management. The results indicated that the amount of wetland and tidal flat habitat as well as the distance required for cranes to reach water and wetlands influence habitat selection. Crane presence increased in areas that were closer to herbaceous wetlands with more wetland habitat available nearby. Cranes were significantly clustered along the southern coast of Aransas, on the Lamar Unit and adjacent land, as well as on Matagorda and Sand Jose Islands. Cranes were also highly clustered to the east of Aransas, across San Antonio Bay. The areas of high clustering correspond to crane habitat preference. Protecting additional preferred habitat can help the crane population continue to grow and establish territories outside of refuge boundaries.

KEYWORDS: Aransas National Wildlife Refuge, endangered species, prescribed fire, spatial aggregation

INTRODUCTION

The endangered whooping crane (*Grus americana*) population in North America once contained approximately 1,300 birds during the mid-1800s (Allen 1952). However, by the early 1900's, overhunting and habitat loss caused a rapid decline in the population, reducing it to approximately 15 birds, bringing whooping cranes to the brink of extinction. Through intensive conservation efforts, the only remaining wild population of whooping cranes, known as the Aransas-Wood Buffalo population (AWBP), has significantly increased over the last 75 years. While protected areas have been established, the goals of down-listing the whooping crane from endangered to threatened and eventually removing it from the endangered species list are dependent on understanding crane habitat selection in conjunction with population dynamics and crane behavior, particularly on their known wintering grounds. The AWBP are philopatric and return to the same locations each year to breed in Wood Buffalo National Park, Alberta and Northwest Territories, Canada (Johns et al. 2005, Timoney 1999, Lewis 1995) and to overwinter on Aransas National Wildlife Refuge (ANWR) along the Texas gulf coast, USA (Stehn and Prieto 2010, Lewis 1995).

According to the international recovery plan for the whooping crane (CWS and USFWS 2007, Endangered Species Act 1973), three stable populations of reproductive birds capable of surviving environmental disturbances are required and each must be self-maintained for ten years before the species can be down-listed from “endangered” to “threatened”. However, as the AWBP is the only wild viable population in the world, a higher minimum of 1000 individuals and 250 breeding pairs is required if two other self-sustaining populations cannot be maintained. It is likely that this target population will allow the species to withstand stochastic environmental events such as hurricanes and severe drought, as well as remain genetically sound (CWS and USFWS 2007). One-hundred and eighty-three pairs of breeding whooping cranes were counted and the population was comprised of approximately 505 birds in 2018 (Harrell and Bidwell 2019). More than 67 pairs with average territory sizes of 172 ha will likely exceed the protected boundaries of the refuge (Gil-Weir et al. 2012, Miller et al. 1974). The question remains, will there be enough habitat with essential resources on Aransas and surrounding lands to support the needed 1,000 birds to delist the whooping crane? The exponential population growth of the AWBP over the last 70 years has created a need to evaluate habitat selection to protect additional critical habitat, particularly as the population continues to expand their overwintering territory (Gil de Weir 2006).

Research regarding whooping crane behavior and habitat use has led to several conclusions about habitat selection during the breeding season and migration, but less is known about habitat selection on overwintering sites. Breeding sites in boreal Canada are dominated by graminoids, open water, and a mosaic of ponds and marshes with calcium carbonate and lime-rich bottoms (Timoney et al. 1997, Lewis 1995). Generally, cranes select habitat that is visually open and away from forested wetlands; typically more than 90 meters from the nearest visual obstruction, and even further from human developments (Pearse 2017, Austin and Reichert 2005, Baker et al. 1995). Cranes also prefer shallow water for foraging and roosting with depths no deeper than one meter (Austin and Richert 2005). Migration is risky for whooping cranes as they experience increased mortality rates due to collisions with powerlines, unavailable habitat, and higher likelihood of human disturbance (Stehn and Wassenich 2008, Johns et al. 1997, Faanes 1992, Lewis et al. 1992). Whooping cranes may be more opportunistic when selecting resources during migration. However, studies show that most roost sites are located less than one kilometer from wetlands (Belaire et al. 2014, Austin and Richert 2005).

Generalizations about habitat use during winter include preference for visual openness (Hunt 1987) and shallow water (Pickens et al 2017, Wright et al. 2014). Cranes are widely known to use salt marsh habitat in known territories (Chavez-Ramirez and Slack 1999, Chavez-Ramirez 1996) and prefer upland habitat closer to salt marshes on ANWR (Hunt 1987). Anecdotal evidence has suggested cranes use recently burned upland habitat on ANWR (Chavez-Ramirez 1996, Hunt 1987), which is important as managers at ANWR routinely conduct summer and winter prescribed burns to maintain crane habitat by reducing woody plant encroachment. However, beyond some of these generalizations, most winter studies of whooping cranes have focused on resource availability and crane behavior (Pickens et al. 2017, Tiegs 2017, LaFever 2006, Chavez-

Ramirez and Slack 1999, Chavez-Ramirez 1996, Hunt 1987) rather than on the attributes of essential habitat.

Current threats to endangered species include habitat degradation, habitat loss, and invasive species, among a growing list of other threats (Venter et al. 2006). Declining species have a higher risk of extinction if they exhibit traits such as low population density, low reproductive rates, or have a small geographic range (Purvis et al. 2000). As threats to these species become compounded, preventing the extinction of threatened and endangered species has become a priority. Setting aside protected areas has been a preferred method of ensuring suitable habitat remains for species of conservation concern. For many endangered species, protecting habitat is essential for their survival. For example, species listed under the Endangered Species Act with critical habitat designations were more likely to see increasing or stable populations than those without protected critical habitat (Taylor et al 2005). It is important to understand the characteristics of the landscape that drive species use of habitats to protect or restore habitat that is more valuable to a particular species. While critical habitat has been designated for the whooping crane on ANWR, the refuge and the cranes still face threats. The intracoastal waterway is causing erosion and degrading available wetland habitat (Evans and Waring 1993). Habitat is being lost to woody plant encroachment from both oaks on the upper portions of the refuge and black mangroves (*Avicennia germinans*) along the coast. Finally, habitat is threatened by rising sea levels and climate change as well as weather disasters such as hurricanes. If habitat is degraded or lost, we need to be able to replicate the habitat and landscape characteristics preferred by whooping cranes. Additionally, as the population is growing and expanding use of habitat beyond the borders of the protected area, investing in protecting additional habitat should be guided by crane habitat preference.

While resource availability affects crane habitat preference, research is still needed on winter habitat selection on ANWR and what landscape level factors may be driving selection to help focus future conservation efforts by recommending habitat types and locations for the growing whooping crane population. Our objectives were to 1. determine spatial and temporal patterns in whooping crane occurrences on ANWR during the winter 2. Evaluate the spatial relationship between environmental variables and whooping crane occurrence on ANWR 3. Provide recommendations for future conservation efforts for wintering whooping cranes.

METHODS

Site Description

ANWR was created to protect the habitat of the whooping crane in 1937 (Evans and Waring 1993, Stevenson and Griffith 1946) and was eventually declared critical habitat for the whooping crane in 1978 (USFWS 1978). ANWR is located along the southeastern coast of Texas, USA, north of Corpus Christi and south of Austwell (Figure 1). There are several administrative units that comprise the refuge complex. The main unit of the refuge is Aransas (also referred to as Blackjack Peninsula), which is comprised of 19,126 hectares and is bordered by Aransas Bay and San Antonio Bay. The second largest section of the refuge, Matagorda Island, is 22,939 hectares and is bordered by Mesquite Bay to the southwest and Espiritu Santo Bay to the northwest and by the Gulf of Mexico on the south. Carlos Bay and Mesquite Bay are part of the Intracoastal Waterway between Aransas and Matagorda Island. The remaining units of the refuge are known as the Tatton Unit (3,063 hectares), which is to the northwest of the Aransas unit, the Lamar Unit (396 hectares), which is across Saint Charles Bay to the west of Aransas, and the Myrtle Foester-Whitmire Unit (1,392 hectares), which is located approximately 37 kilometers northeast of Aransas. The vegetation on the northern half of Aransas consists of live oak (*Quercus virginiana*) dominated woodlands. Along the southern half of the Aransas unit and Matagorda Island, grasslands, salt marsh, and tidal flats are prevalent, with saltmeadow cordgrass (*Spartina patens* (Aiton) Muhl.) and sea-oxeye daisy (*Bocconia frutescens* L.) common species. The climate is humid and subtropical, with an average rainfall of 883 mm and a yearly average temperature range between 19.5 and 25.5°C. Elevation ranges from sea level to 28 meters throughout the refuge. For the remainder of the document, ANWR will refer to the Aransas unit, the Tatton unit, the Lamar unit, and Matagorda Island unless otherwise specified.

The refuge managers have implemented a variety of management activities over the years. Prior to being designated as a national wildlife refuge, winter prescribed burns were conducted between 1919 and 1935 when under private ownership (Halloran 1943). Management on the refuge between 1945 and 1981 included brush reduction via roller chopping, disking, and planting when needed. Cattle were grazed on the refuge through the middle of the twentieth century (Allen 1952). However, the current management program focuses on the use of prescribed burns, which the refuge began implementing regularly in 1985. ANWR, similar to much of the Texas Gulf Coast region, has experienced substantial landcover change in the form of woody plant encroachment (Saintilan and Rogers 2015, Brown and Archer 1999). Summer burns are conducted to reduce the spread of oaks, while winter prescribed burns are used to create opportunistic foraging opportunities for whooping cranes (USFWS 2010).

Whooping Crane Data

Since 1950, the United States Fish and Wildlife Service (USFWS) has conducted regular surveys of the AWPB to monitor the size of the population and document crane locations during the winter months (October through May). Prior to 2011, aerial surveys were conducted on a bi-weekly basis with one pilot and one observer in a Cessna single engine aircraft. Flights were conducted at a height of 61 meters above sea level at an average speed of 167 km/h and targeted known crane territories (Stehn and Taylor 2008). During surveys, observers recorded each crane location and the number of cranes at each location on paper maps. Of note, the historic survey method described above has received some scrutiny for its unsystematic approach and inability to detect all individuals within the population as the population has continued to grow. A new survey methodology was introduced in 2012 to address this issue (Strobel and Butler 2014). Strobel and Butler (2014) concluded that the previous method of aerial surveys resulted in a probability of less than one that all whooping cranes were surveyed and a new survey methodology was implemented in 2011 (Butler et al. 2014). However, because the population was so small, it is likely that surveyors knew the landscape and where cranes were likely to be located and therefore could account for almost all of the population. The ability to have spatial data over decades before the availability of GPS provides useful insights on long term patterns and trends. We are focused on long-term patterns in whooping crane occurrences and habitat use and we relied on the historic data from 1950 to 2010 for our analyses. The entire dataset contained 37,373 observed crane locations (Table 1).

Ancillary Data Acquisition

Landcover

We acquired landcover data from the Ecological Systems Classification project, which was a collaboration of the Texas Parks and Wildlife Department, the Texas Natural Resources Information Service, and the Missouri Resource Assessment Partnership (MoRAP) at the University of Missouri. The collaboration created a high-resolution vegetation (10 meter resolution) landcover database for Texas (Ludeke et al 2010a, Ludeke et al. 2010b, NatureServe 2009). Using the geospatial vegetation data and descriptions of the vegetation, we condensed and reclassified the vegetation layer into nine landcover types: open water, bare ground, grassland, herbaceous/grassland wetland, row crops, tidal flats, urban, woody shrubland, and woody-shrub wetland (Figure 2). (See Appendix E, Table 1 for a list of combined vegetation associations).

Fire Data

ANWR implemented a prescribed burn program on the refuge in 1985 to create habitat and forage opportunities for the whooping cranes. A list of prescribed burns conducted on ANWR between 1985-2010 was provided by the USFWS. We mapped the extent and location of fires on ANWR using the normalized burn ratio and differenced normalized burn ratio methodology (Key and Benson 2005) based on burn date and Landsat satellite imagery (see Appendix F for detailed explanations of these indices). Because it has been suggested that whooping cranes use recently burned habitat (Chavez-Ramirez et al. 1996), we created a fire-related variable for the predictive statistical models based on the mapped burns.

Analyses Overview

To address our first objective of determining the spatial and temporal patterns in whooping crane occurrences on ANWR during the winter, we used exploratory spatial data analysis, including spatial statistics, and GIS analyses, including the kernel density and mean center methods. To address the second objective of evaluating the spatial relationship between environmental variables and whooping crane occurrence on ANWR, we developed two global generalized linear mixed models (GLMM). The first, a Poisson GLMM, was used to determine if the distance to landcover types influenced the number of cranes present. The second, a binomial GLMM, was used to determine if the amount and type of landcover available affected crane presence/absence. The landcover predictor variables in both GLMM analyses were bare ground, grassland, herbaceous-grassland wetland, tidal flat, urban, woody shrub wetland, woody shrubland, row crops, and open water. The landcover variables were chosen to reflect known whooping crane preferences. For example, distance to urban areas could indicate avoidance of human disturbances. For the first model, the distance to nearest burned habitat was included, while for the second model, the number of times an area was burned was included as the fire-related predictor variable. We chose not to include variables related to climate and weather as they were too spatially broad to influence the analyses at a fine scale.

Density and Aggregation Spatial Statistics

To track the movement of the crane population over the course of the study, we used the mean center calculation to determine the geographic center of the documented crane locations, which shows the central tendency of movement of the crane population over time.

Kernel density estimation provides a visual representation of the density of spatial features, in this case, mapped crane point locations. It is a non-parametric approach to estimating the probability density function for a variable of interest, in this case, the cranes. A kernel is constructed around each point with a smoothing parameter (bandwidth) guiding the extent of the kernel, and the overlapping kernels are summed to create a grid of density values. Values can be weighted by the number of cranes present at each observation point location, which is key in this instance since the observation record often includes multiple cranes at a single point. Kernel density estimation is useful for indicating where point locations are most densely clustered, but the statistic cannot determine whether or not the observed patterns are random or are being driven by some underlying ecological processes (Kalinic and Krisp 2018). Therefore, it is primarily used in an exploratory manner to identify 'hotspots' of occurrences that can then be used to investigate the underlying processes. Kernel density was calculated using crane locations by decade from 1950-2010 to show the progression of areas of highest crane density and how they change over time (Worton 1989). Kernel density is calculated using the kernel function:

$$\hat{f}_h\,(x) \;=\; \frac{1}{nh^2}\sum_{i=1}^{n} K\left[\frac{x - X_1}{h}\right]$$

Where, K = probability of occurrence, X_1 = independent sample from an unknown density (crane point locations), n = random sample of independent points, and h = bandwidth. The default bandwidth calculated in ArcMap is based on the mean center of the point data. We calculated KDE using the default bandwidth for each year between 1950 and 2010. We chose the median value and ran kernel density estimations based on a bandwidth of 1857 meters. The bandwidth is calculated as:

$$Search\ Radius\ =\ 0.9\ \times\ min\left(SD,\sqrt{\frac{1}{ln(2)}}\times D_m\right)\times n^{-0.2}$$

Where, D_m = the weighted median distance from the mean center, n = the sum of the population field values (number of cranes per location), and SD = standard distance.

We then used the Jaccard Index to determine if the areas of high kernel density were similar from year to year or whether the 'hotspots' of crane occurrences were changing over time, as this would indicate expanded resource use. The Jaccard index was calculated for every five years from 1950-2010 and each segment of five years was compared to the 1950-1954 area of high density. The Jaccard index calculates a range from zero to one, the closer to one, the more overlap between areas of kernel density are present (Rice and Belland 1982). The Jaccard index is calculated as:

$$J_{(I,j)} = a/(a+b+c)$$

Where, i,j = area of kernel density in five year divisions, a = area (hectares) of overlap between kernel densities, using the bandwidth calculated above, for i and j, b = area of kernel density for time period, and c = area of kernel density for time period j.

Generalized Linear Mixed Models

We developed two global models to assess landcover associations with crane locations on ANWR based on two different response variables (number of cranes and crane presence/absence). We used a subset of 24,038 crane spatial point locations collected over 25 years between 1985-2010, which corresponds closely to when the vegetation data were created and to the prescribed burn program on ANWR. This allowed us to include either the amount of burned area or the number of times an area was burned as a predictor variable.

The first global model, a Poisson GLMM was used to determine whether distance to landcover type increased the number of cranes present. Predictor variables used in the Poisson GLMM included distance to landcover type (measured in meters) for the following: bare ground, grassland, herbaceous grassland wetlands, tidal flat, urban, woody shrub wetland, woody shrubland, burned area, row crop, and open water. Planar distance in meters from each crane spatial point location to the closest polygon of landcover type was used to calculate distances.

The second global model, a binomial GLMM was used to determine whether crane presence or absence was influenced by the amount and type of landcover available. Model variables included: number of fires conducted in an area, bare ground, urban, tidal flat, open water (we did not distinguish between fresh, brackish, or saltwater), herbaceous grassland wetland, grassland, woody shrubland, woody-shrub wetland, and row crop. The study area was determined by drawing a bounding polygon around all crane locations from the full dataset of mapped crane locations (1950-2010). Next, a digital hexagon grid was overlain on the study area. Each side of individual hexagons was 750 meters, resulting in the area of each hexagon being slightly smaller than the average crane territory (Stehn and Prieto 2010). Landcover types (square meters) were summed for each hexagon across the study area. For the binomial model, the number of cranes was converted to a presence/absence variable. If crane points were located in a landcover type (e.g., tidal flat) they were re-coded as present and given a value of one for that landcover within each hexagon, and if no cranes were located within a landcover class, they were re-coded as absent and given a value of zero. The variable, “number of burns conducted” was created by using all mapped burn polygons (created using NBR and dNBR, see

Appendix F). A new polygon layer was created using the count overlapping polygons tool in ArcGIS. The burns were then summed per hexagon to create the predictor variable. Generalized linear mixed models were run using Program R (2014). Spatial statistics were conducted using ArcGIS Desktop Version10.4.1 (ESRI 2016).

We performed a backward stepwise model selection using AIC for both global models to select the best model from the global model for each response variable and the set of alternative models. Backward stepwise selection is a common method used to determine the most parsimonious model. Models were considered competitive if they were within two units of the model with the lowest AIC value.

RESULTS

Population Distribution Change

The mean center of the crane population moved 1,000 to 2,600 meters between 1950-2010 (Figure 4). The mean center exhibited a trend of moving towards Matagorda Island from the coast of ANWR. Between 1950 and 1960 the population mean center moved ESE by 1,231 meters. The mean center of the population then shifted southwest by 2,537 meters between 1960 and 1970. The population mean center then shifted ENE by 2,967 meters between 1970 and 1980. Looking at the decade 1980-1990, the mean center moved northeast by 2,542 meters. It eventually shifted southeast between the 1990s and 2000s by 1,130 meters.

The whooping crane locations showed a pattern of spatial aggregation (Figure 3). The areas of high density were similar during the late 1950s and early 1960s when compared to the kernel density hot spots of the early 1950s (Jaccard index = 0.708 (1955-1959) and 0.622 (1960-1964)), but continued to be more dissimilar by the 2000s when compared to the early hotspot (Jaccard index comparing 1950-1954 to 2005-2010 = 0.195) (Table 6). Areas of high density began to expand towards Matagorda Island during the 1970s until areas of high crane density were located on both shorelines of ANWR and Matagorda Island. Areas of high density were associated with the Lamar unit and areas adjacent to the north and east, Matagorda and San Jose Islands, the southern edge of ANWR, and directly to the east of ANWR and approximately 7.5 -13 kilometers south of the town of Sea Drift.

Using ArcMap, we clipped the MoRAP landcover layer to the areas of highest density and were able to get a more detailed description of habitat that the cranes use. This layer was then used to indicate where potential areas are located near the refuge with similar habitat. Within the areas where kernel density was highest, the habitat types that comprised most of the area (90.30%) included coastal: tidal flat (2.60%), Gulf Coast: salty prairie (4.92%), South Texas: algal flats (5.51%), Coastal and Sandsheet: deep sand grassland (10.02%), Coastal and sandsheet: salt and brackish high tidal marsh (10.59%), coastal and sandsheet: salt and brackish low tidal marsh (18.12%), and open water (38.55%). These habitats have low vegetation heights and are usually dominated by graminoids, bluestem spp., *Paspalum* spp., *Physalis* spp., croton spp., *Spartina* spp, *Distichlis* spp, and blue-green algae.

Generalized Linear Mixed Models

Detection of cranes was predicted by distance to several landcover types. The closer to water and herbaceous-grassland wetlands, the more likely more cranes would be detected. The likelihood of detecting cranes increased as distance to herbaceous grassland-wetlands decreased ($p = 0.01$). Also, as the distance to open water decreased, the likelihood of crane detection increased (*p* = 0.000). Furthermore, the closer to woody shrubland habitats, the likelihood of detecting cranes decreased (*p* = 0.01). Between 1985 and 2010, crane locations were an average of 35 (SE ± 0.54) meters from herbaceous-grassland wetlands and an average of 137 (SE ± 1.57) meters from water (Table 2), with yearly averages ranging from 13-81 meters and 61-247 meters. Crane locations tended to be further away from areas of bare ground and row crops at average distances of 9,239 (SE ± 28.76) and 16,484 (SE ± 31.00) meters, with yearly averages ranging from 8,073-10,531 meters and 15,228-17914 meters. The stepwise model selection with AIC produced three comparable models for the distance to habitat (Table 4). Model selection suggests that three models performed well (ΔAICc <2), but the best model was model 3, which included variables of distance to: herbaceous-grassland wetlands, water, row crops, urban habitat, woody shrubland, and woody-shrub wetland, and burned area. The global model performed poorly compared to these models (ΔAICc ≥ 3.65).

Crane presence/absence was predicted by the type of landcover available. In summary, the more herbaceous-grassland wetland landcover, the more burned habitat, and the more tidal habitats were available, the likelihood of crane presence increased. The likelihood of crane presence increased in locations that contained more herbaceous-grassland wetlands (*p* = 0.000), more tidal flats (*p* = 0.000), and a larger amount of burned area (*p* = 0.000). As habitat coverage of water (*p* = 0.001) and grassland also increased (*p* = 0.001), so did the likelihood of crane presence. As the area of woody-shrub wetland increased, so did the likelihood of crane presence (*p* = 0.01). This may be tied to the fact that there are small patches of woody wetlands within the herbaceous-grassland wetland habitat that the cranes prefer. More than 75% of cranes locations were located in either water or herbaceous-grassland wetlands (Table 3). Based on habitat available, cranes are using habitat that is available in the smallest amounts. Within the study area, water comprised the largest landcover type at 111,928 hectares, followed by herbaceous-grassland wetlands at 53,828 hectares. However, tidal flats covered only 3,112 hectares, indicating that this landcover provides important habitat because they are using it a higher proportion when there is much less available. The stepwise model selection with AIC produced two comparable models for amount of habitat available (Table 5). Model selection results for presence/absence based on landcover type indicate models two and three performed well (ΔAICc <2), but model three performed best. The variables included in this model were herbaceous-grassland wetland, tidal flats, water, grassland, bare ground, burned area, and woody-shrub wetland. The global model and model 1 performed poorly compared to models two and three (ΔAICc ≥ 3.24).

DISCUSSION

Protection under the Endangered Species Act has allowed the AWBP to begin recovering from a population low of 15 birds, but the requirements to down-list the whooping crane have not been met. Limited habitat serves as a potential limiting growth factor of the population (Lewis 1995). It can affect overwintering survival, competition for needed resources, and can potentially affect success during the breeding season (Norris et al. 2003). While habitat is not necessarily a restricting variable yet, it could become a problem in the near future as the AWBP continues to grow exponentially. While the number of individuals in the AWBP is increasing, our results indicate that the core area used by the population has also expanded. The results indicate that the mean center of the population has moved towards Matagorda Island. The results of the kernel density analysis indicate that although one area of high density remains on Aransas, the rest have spread towards Matagorda Island and towards San Jose Island, the Lamar Unit of ANWR, and areas to the east of San Antonio Bay. Areas of high density and crane location indicate that cranes may prefer protected areas, such as the spread of high density areas to the Lamar Unit, all of Matagorda Island, and the area in and surrounding Welder Wildlife Flats, which is located on the eastern side of San Antonio Bay, further from Aransas. The AWBP has also continued to stay clustered on refuge habitat and near protected areas in the vicinity of the refuge, with only a one percent chance that the clustering was random in nature. Areas of high crane density have expanded, indicating the importance of these area for overwintering habitat.

Territoriality could be a partial explanation for the clustering. The territory size of overwintering cranes has been changing as the population increases (Stehn and Prieto 2010). The average size of territories on ANWR has decreased as the population has grown (Harrell and Bidwell 2013, LaFever 2006). The current average territory size at 172 hectares, coupled with over 183 breeding pairs, will likely exceed the boundaries of the refuge (Harrell and Bidwell 2019, Gil-Weir et al. 2012, Miller et al. 1974). Territoriality is a common behavior in cranes (Alonso and Alonso 1999) and juvenile cranes and young pairs may be simply establishing new territories next to previously established territories and not seeking potential sites farther from the rest of the overwintering population, as suggested by Stehn and Prieto (2010). Essentially, as new pairs are formed, the juveniles establish territory as close as they can to their parents, thus they are forced to move further away from the core center of the territories yet remain close to other pairs on the refuge, indicating that the suitable habitat immediately surrounding the refuge is important for whooping crane winter survival.

Whooping crane food preferences influence the selection of foraging habitat. Cranes spend a majority of their time foraging during winter (LaFever 2006). The quality of resources such as water salinity and depth affect food availability, abundance, and nutritive quality (Wozniak et al. 2012, Darnell and Smith 2004). Cranes require a high abundance of blue crabs in order to survive the winter (Pugesek et al. 2013) and the availability of crabs is directly related to the amount of wetland habitat. This further supports our results that being closer to wetland habitat and open water in addition to having more wetland habitat available increases the likelihood of crane presence. The size of blue crabs was found to be lower in inner marsh areas rather in the connected ponds of the outer marsh area on ANWR (Hoeinghaus and Davis 2007). Creating connectivity between these habitats (i.e., wetlands and tidal flats) would be an asset because cranes can benefit from larger crabs as they provide a primary source of crude protein in their diet (Nelson et al. 1996, Hunt and Slack 1989).

Crane behavior and habitat preferences influence aspects of habitat use. Cranes prefer unobstructed views up to one kilometer (Faanes 1992) and shallow water for roosting during migration (Armbruster 1990), making the logical assumption that similar requirements may be preferred for overwintering roosting locations. Cranes spend more time foraging and resting in open water and marshes, whereas more alert behavior has been demonstrated in the uplands on ANWR (Bishop et al. 1987). Variation in seasonal water levels impact crane habitat use because prey densities change, which leads to more movement to other habitat types on the refuge when food becomes scare (Kang and King

2014, Chavez-Ramirez and Slack 1999). The availability of additional food sources in uplands when blue crab abundance is low due to high salinity levels or drought may draw the cranes to the upland woody-shrub habitat on the refuge. Cranes may also be drawn to the woody-shrub uplands to access sources of fresh water. Whooping cranes will drink at inland ponds when brackish water near the coast becomes too saline (Lewis 1995).

The results of the binomial model showed cranes were likely to be present on woody-shrub wetland habitat. This could be related to the refuge's fire management. ANWR routinely conduct prescribed burns to create suitable habitat for cranes. Winter burns are conducted to provide feeding opportunities such as access to small reptiles and amphibians as well as exposed acorns (Venne and Frederick 2013, Chavez-Ramirez et al. 1996, Hunt and Slack 1989). Summer burns are conducted to maintain the openness of the coastal prairie grasslands and wetlands and to help prevent further encroachment of woody species. Prescribed fires can reduce woody plant cover that can in turn reduce visual obstruction of potential predators to cranes (Chavez-Ramirez and Wehtje 2012). The data were too temporally broad to indicate that cranes used recently burned habitat, but it could indicate that burned habitat provides open areas and potential sources for opportunistic foraging. This could also be related to patches of woody-shrub wetland interspersed with habitat cranes prefer.

Future threats that face whooping cranes include shoreline erosion and major loss of tidal flat and wetland habitat due to the Gulf Intracoastal Waterway (GIWW) that exists between their two major habitat locations (ANWR and Matagorda Island) (Davis et al. 2009, Evans and Waring 1993). The GIWW development and maintenance resulted in an 11% loss of critical crane habitat from erosion, particularly affecting low marshes and tidal flats (Sherrod and Medina 1992, Evans and Waring 1993). Additionally, vessels traversing the GIWW impact the sedimentation and water flow levels of wetland habitat near the waterway (Davis et al. 2009), affecting habitat quality and food sources. Since the development of the GIWW, known crane territories have been altered, particularly on Sundown Bay near Mustang Lake, Dunham Bay, and South Matagorda Island (Labuda and Butts 1979).

The future of whooping crane habitat protection should focus on several factors. First, the prescribed burn program on ANWR should be evaluated, particularly in context with the threat of woody plant expansion into the areas where crane locations are densest. The burns help re-set succession and likely kill woody seedlings thereby limiting the expansion of woody plants. The burns also help maintain herbaceous-grassland wetlands that the cranes prefer. Second is to work on creating engineered wetlands with similar characteristics to the habitat found within the high density areas. This is particularly important with the threat of climate change and rising sea levels. Future research is needed to determine what is unique about the juxtaposition and interaction among the preferred wetland habitat types. Third, protecting areas where cranes are now clustering outside of the refuge should be a priority. Several of the areas where the cranes are already clustering or areas near the refuge that appear to provide the required habitat are partially protected in some cases. The cranes are clustering around Welder Flats Wildlife Management Area (WMA). Additional land surrounding this WMA could be set aside to create a larger protected area. Similarly, area surrounding Mustang State Park and area north of the Lamar Unit of ANWR would be prime areas to expand protection. Looking further from the refuge, areas around Long Lake, Mad Island Wildlife Management Area, Port Bay, and Swan Lake all demonstrate potential habitat within distance of the refuge that would benefit the cranes if protected. Fourth, reduce the impact of erosion from traffic on the gulf coast intracoastal waterway. The habitat on the coast of ANWR and Matagorda Island is important to the cranes, but this unfortunately buffers the intracoastal waterway. Work could be done to reduce the traffic using the intracoastal water way (e.g., limiting traffic to a few days a week).

Marshes and tidal flats, as indicated by our results, are two habitat types where the more acreage present, the more likely cranes are to use the habitat, highlighting the need to continue protecting these valuable habitat types for the endangered whooping cranes. Not one specific variable is more important than another in predicting crane habitat use or preference as they are just one of many factors. These additional factors could include how much boat traffic on the intracoastal water way influences habitat use, as well as salinity levels, abundance of crabs, and potentially other landscape level patterns and interactions among the landcover types. However, understanding the habitat preferences and where cranes cluster is important because sea level rise, habitat alteration, and unpredictable impacts from climate change are likely to affect the coastal overwintering habitat of whooping cranes (Chavez-Ramirez and Wehtje 2012).

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TABLES AND FIGURES

Table 1. Number of georeferenced crane locations used in spatial analyses by decade from 1950-2010.

Years	Number of georeferenced crane locations
1950-1959	1,408
1960-1969	2,878
1970-1979	5,893
1980-1989	5,971
1990-1999	9,260
2000-2010	11,963

Table 2. The average distance to each habitat type (meters) from crane locations on ANWR and Matagorda Island from 1985-2010 (+/- standard error).

Year	Bare Ground	Grassland	Herbaceous Grassland Wetland	Water	Row Crops	Tidal Flats	Urban	Woody-Shrub Wetland	Woody Shrubland	Burned Area
1985	9,996.6 (151.5)	678.0 (18.9)	34.6 (2.5)	78.9 (5.6)	16,177.5 (137.1)	251.4 (8.3)	4,180.8 (74.9)	1,553.7 (33.8)	890.2 (19.4)	1,817.6 (94.3)
1986	9,864.3 (177.5)	698.4 (23.5)	38.8 (3.6)	84.7 (9.3)	15,612.0 (165.2)	283.8 (12.1)	3,888.5 (88.5)	1,451.1 (37.7)	882.7 (26.1)	1,712.8 (111.3)
1987	10,499.6 (185.7)	628.6 (21.7)	28.2 (2.3)	60.8 (4.5)	15,252.8 (169.7)	233.9 (9.1)	3,684.5 (91.6)	1,384.2 (38.0)	855.1 (25.0)	1,596.6 (122.0)
1988	10,530.6 (134.7)	514.9 (15.2)	39.8 (3.2)	158.9 (9.0)	15,374.1 (119.7)	264.4 (8.7)	3,709.5 (67.8)	1,201.5 (26.2)	737.2 (18.7)	1,504.4 (85.6)
1989	9,945.0 (227.0)	541.3 (24.5)	49.5 (6.1)	174.8 (17.2)	15,936.7 (192.6)	309.6 (16.3)	3,754.7 (106.2)	1,416.2 (48.0)	715.7 (26.6)	1,864.8 (148.7)
1990	9,992.3 (155.5)	497.9 (16.4)	35.2 (3.4)	143.6 (9.4)	15,646.2 (149.1)	263.0 (9.8)	3,685.6 (77.7)	1,395.0 (34.5)	733.7 (19.0)	1,811.5 (103.3)
1991	9,771.1 (122.9)	538.3 (13.1)	25.8 (2.1)	148.2 (7.8)	15,562.4 (125.6)	248.6 (7.8)	3,624.5 (62.6)	1,390.6 (27.9)	766.1 (14.7)	1,837.6 (81.3)
1992	9,581.3 (145.2)	550.5 (14.8)	29.2 (2.5)	122.1 (7.1)	16,029.7 (138.3)	242.6 (8.8)	3,932.0 (74.7)	1,368.7 (31.7)	796.5 (16.8)	1,884.9 (93.6)
1993	9,403.3 (189.7)	538.2 (20.2)	28.9 (3.6)	152.1 (14.2)	15,228.1 (193.1)	274.0 (12.4)	3,504.7 (99.8)	1,362.1 (42.1)	780.7 (22.8)	1,985.5 (129.7)
1994	9,380.4 (143.3)	532.0 (15.2)	21.0 (1.6)	103.0 (4.1)	16,021.1 (157.6)	234.8 (7.5)	3,880.4 (74.0)	1,463.1 (29.6)	847.7 (17.4)	2,168.4 (97.1)
1995	9,519.7 (158.4)	520.9 (16.5)	35.7 (2.7)	135.7 (9.5)	16,203.8 (176.4)	267.8 (9.3)	3,950.9 (79.6)	1,505.9 (36.6)	811.8 (19.2)	2,360.2 (116.8)
1996	8,650.7 (127.5)	467.8 (12.9)	26.1 (2.0)	134.1 (7.1)	16,447.9 (148.2)	243.6 (7.4)	4,050.9 (64.7)	1,493.2 (28.8)	773.5 (14.2)	2,739.8 (100.0)
1997	8,844.7 (131.4)	555.2 (14.6)	24.9 (2.2)	126.2 (6.6)	16,172.3 (150.7)	248.2 (7.7)	3,993.5 (68.6)	1,474.2 (30.4)	819.1 (15.7)	2,536.7 (103.1)

Year	Bare Ground	Grassland	Herbaceous Grassland Wetland	Water	Row Crops	Tidal Flats	Urban	Woody-Shrub Wetland	Woody Shrubland	Burned Area
1998	8,652.1 (139.3)	633.5 (15.0)	13.0 (1.2)	102.3 (3.7)	16,883.3 (150.4)	232.2 (6.8)	4,267.3 (71.2)	1,560.0 (30.0)	905.9 (17.1)	2,661.2 (104.9)
1999	9,134.3 (143.9)	490.6 (14.5)	39.3 (3.1)	148.2 (10.3)	15,498.3 (152.8)	248.6 (9.5)	3,837.9 (73.7)	1,333.9 (29.6)	743.7 (17.3)	2,545.9 (113.5)
2000	8,954.9 (132.6)	494.1 (13.3)	38.6 (3.1)	188.0 (9.8)	16,086.1 (162.9)	272.8 (8.1)	3,765.1 (62.1)	1,370.1 (26.2)	806.9 (15.8)	3,055.8 (116.3)
2001	8,579.3 (118.4)	554.4 (12.2)	18.7 (1.2)	88.8 (3.2)	17,038.1 (128.5)	232.3 (5.6)	3,983.7 (59.7)	1,565.9 (26.7)	856.1 (13.6)	3,076.7 (97.6)
2002	8,628.8 (124.2)	513.6 (11.6)	19.1 (1.2)	98.1 (3.0)	17,298.5 (128.1)	222.9 (6.0)	4,124.8 (61.7)	1,587.0 (30.8)	847.3 (13.9)	2,849.2 (95.1)
2003	8,836.2 (126.9)	444.7 (11.3)	22.4 (1.5)	116.2 (4.1)	17,418.8 (132.3)	199.6 (5.5)	4,233.2 (62.0)	1,525.2 (28.0)	756.1 (13.7)	2,682.3 (90.2)
2004	8,977.3 (127.3)	489.9 (12.1)	26.2 (1.8)	114.8 (5.2)	17,379.9 (127.7)	209.6 (5.9)	4,281.2 (62.1)	1,639.7 (31.1)	803.7 (14.4)	2,691.6 (90.0)
2005	9,135.4 (118.0)	502.5 (12.6)	32.2 (2.0)	118.6 (4.8)	17,328.0 (128.4)	225.0 (5.9)	4,163.4 (59.0)	1,682.6 (32.8)	802.4 (14.5)	2,615.8 (90.3)
2006	9,087.9 (160.5)	431.1 (15.9)	30.1 (3.3)	147.1 (7.9)	17,456.5 (179.3)	212.1 (7.5)	3,990.9 (82.0)	1,651.9 (46.8)	761.5 (19.0)	2,608.3 (129.9)
2007	8,769.5 (157.9)	522.7 (15.9)	30.8 (2.7)	115.7 (6.9)	17,645.0 (164.0)	210.4 (7.5)	4,232.0 (78.8)	1,674.2 (39.0)	838.5 (19.1)	2,842.3 (116.9)
2008	9,051.7 (149.1)	423.5 (14.8)	65.0 (4.5)	236.0 (14.1)	17,497.9 (164.6)	287.8 (9.8)	3,809.4 (71.5)	1,593.4 (39.3)	691.0 (18.1)	2,493.5 (114.0)
2009	8,350.9 (167.8)	341.0 (14.9)	67.2 (5.5)	246.9 (13.7)	17,913.8 (206.)	267.1 (11.5)	3,831.7 (83.6)	1,515.0 (46.3)	649.0 (19.2)	2,892.8 (144.6)
2010	8,072.6 (207.4)	428.1 (20.1)	80.8 (8.6)	211.9 (18.8)	17,469.3 (258.9)	277.3 (15.7)	3,851.6 (103.6)	1,524.6 (50.1)	733.3 (26.9)	3,241.1 (184.8)

Table 3. The percent of cranes found in each landcover type by year from 1985 through 2010. Cranes were documented more than 75% of the time in water or herbaceous-grassland wetland landcover.

Water	Water	Herbaceous Grassland Wetland	Grassland	Tidal Flats	Woody Shrubland	Woody Shrub Wetland	Row Crops
1985	57.3	37.8	1.7	2.9	0.3	0	0
1986	58.1	38.1	1.2	1.5	1.1	0	0
1987	57	41.6	0.4	0.6	0.4	0	0
1988	40.5	45.7	4.6	1.8	5.2	2.2	0
1989	41.5	41.9	5.3	3.6	7.5	0	0.2
1990	38.4	49.1	5.4	2.8	4	0.3	0
1991	31.9	55.7	4.8	3.8	3.6	0.2	0
1992	36.9	52	3.3	4.3	3.4	0	0
1993	34.5	55.9	3.9	3.4	1.7	0.6	0
1994	38.8	53.7	2	4.9	0.5	0	0
1995	45	44.3	6.3	3.1	1.3	0	0
1996	34.4	56.6	3.4	5	0.7	0	0
1997	35	57.1	3.8	3.4	0.7	0	0
1998	30.8	64.2	1.2	3.4	0.3	0	0
1999	41.2	45.8	6.8	2.9	3.1	0.2	0
2000	30.9	56.4	7.4	2	2.1	1	0.2
2001	37.6	57	1	4	0.2	0.1	0
2002	34.9	59.3	0.7	4.8	0.2	0	0
2003	33.4	56.1	3.4	6.5	0.6	0	0
2004	35.7	55.8	2	5.2	1.1	0.2	0
2005	36.8	53.1	3.9	4.7	1.5	0	0
2006	30.1	58.4	5.7	4.5	1.2	0.1	0
2007	34.4	55	2.4	5.4	2.7	0	0
2008	28.4	49.4	11.8	2.9	7	0.5	0
2009	24.9	53.1	14.2	4.2	3.5	0.1	0
2010	46.6	37.8	7.2	3.4	4.7	0.3	0

Table 4. Model selection analysis based on a generalized linear model using backwards stepwise selection for the importance of distance to habitat type based on variables included in each model, the number of parameters, AIC value, and difference between the models.

Model	Variables (Distance to Each Habitat Type)	K	AICc	ΔAICc	ω_i
Global	Bare ground, grassland, herbaceous grassland wetland, water, row crops, tidal flats, urban area, woody shrub wetland, woody shrubland, and burned area	11	46791.13	3.66	0.07
Model 1	Grassland, herbaceous grassland wetland, water, row crops, tidal flats, urban area, woody shrub wetland, woody shrubland, and burned area	10	46789.30	1.83	0.18
Model 2	Grassland, herbaceous grassland wetland, water, row crops, urban area, woody shrub wetland, woody shrubland, and burned area	9	46788.12	0.65	0.32
Model 3	Herbaceous grassland wetland, water, row crops, urban area, woody shrub wetland, woody shrubland, and burned area	8	46787.48	0.00	0.44

Table 5. Model selection analysis based on a generalized linear model using backwards stepwise selection for the importance of amount of habitat type available affecting crane presence/absence based on variables included in each model, the number of parameters, AIC value, and difference between the models.

Model	Variables (Square Meters of Habitat Type Available)	K	AICc	ΔAICc	ω_i
Global	Bare ground, grassland, herbaceous grassland wetland, water, row crops, tidal flats, urban area, woody shrub wetland, woody shrubland, and burned area	11	860.52	5.25	0.04
Model 1	Bare ground, grassland, herbaceous grassland wetland, water, tidal flats, urban area, woody shrub wetland, woody shrubland, and burned area	10	858.58	3.31	0.11
Model 2	Bare ground, grassland, herbaceous grassland wetland, water, tidal flats, urban area, woody shrub wetland, and burned area	9	856.79	1.53	0.27
Model 3	Bare ground, grassland, herbaceous grassland wetland, water, tidal flats, woody shrub wetland, and burned area	8	855.27	0.00	0.58

Table 6. Jaccard similarity index results for area (hectares) of kernel densities based on crane locations by five year divisions from 1950-2010. The closer to one, the more overlap that exists between areas.

Years Compared	Jaccard Similarity Index
1950-1954, 1955-1959	0.708
1950-1954, 1960-1964	0.622
1950-1954, 1965-1969	0.456
1950-1954, 1970-1974	0.481
1950-1954, 1975-1979	0.421
1950-1954, 1980-1984	0.467
1950-1954, 1985-1989	0.347
1950-1954, 1990-1994	0.349
1950-1954, 1995-1999	0.293
1950-1954, 2000-2004	0.233
1950-1954, 2005-2010	0.195

Figure 1. Aransas National Wildlife Refuge is located on the Texas Gulf Coast just south of Austwell, Texas, USA. It is comprised of 5 administrative units that total 46,916 hectares: Aransas, Matagorda Island, Tatton, Lamar, and Myrtle Foester-Whitmire.

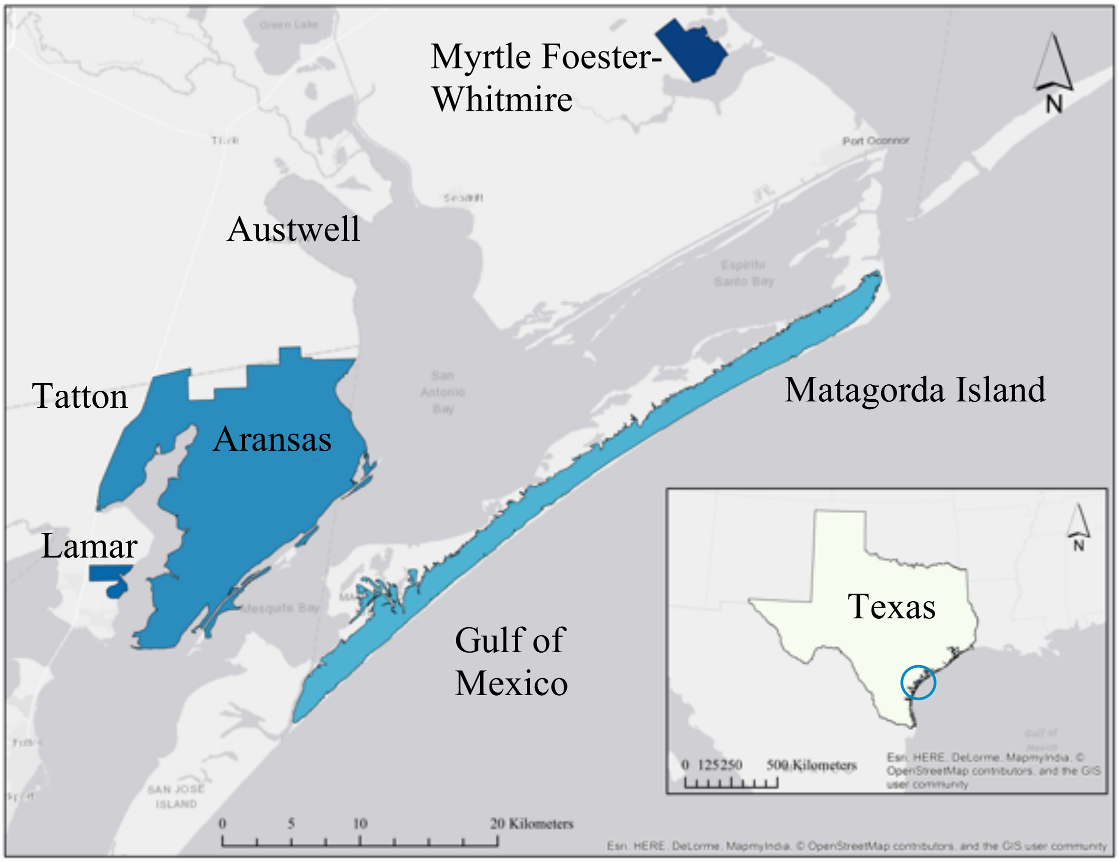


Figure 2. Landcover types on Aransas National Wildlife and the surrounding area.

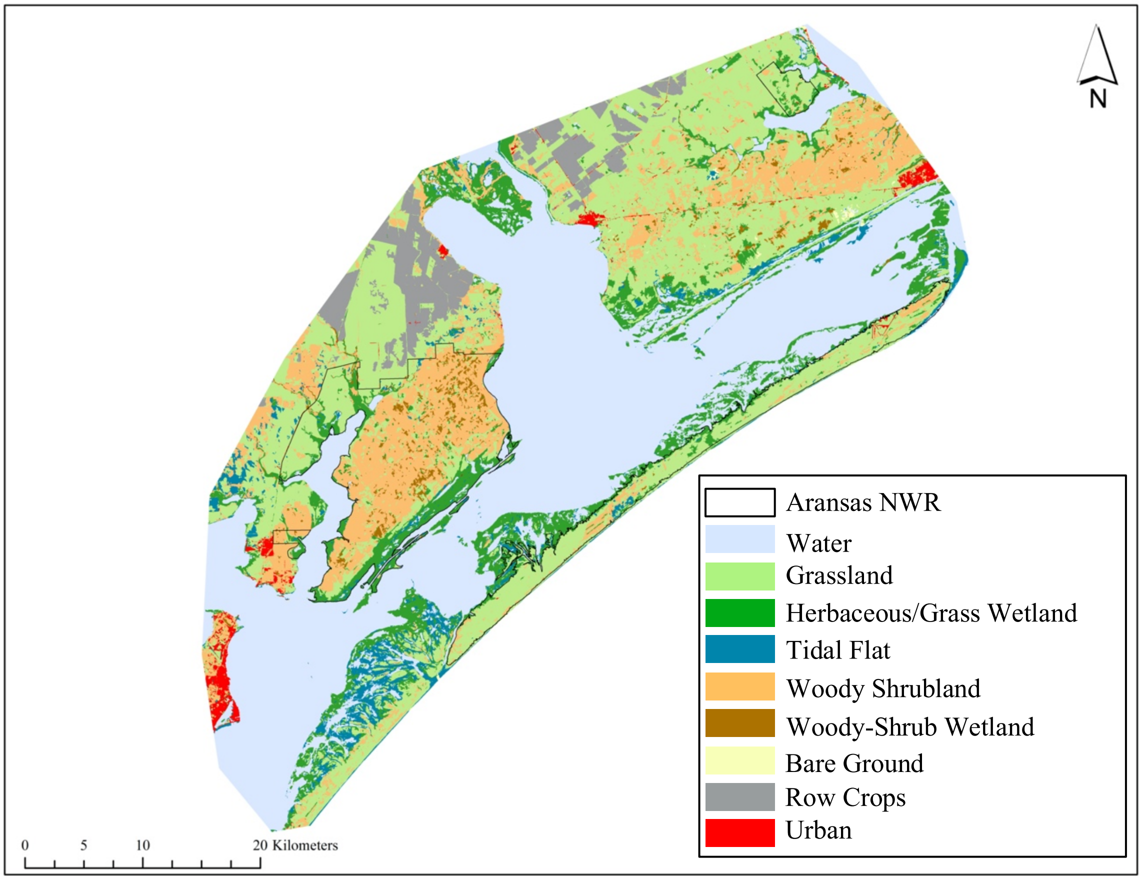


Figure 3. Kernel Density of whooping crane locations between 1950-1959 (A), 1960-1969 (B), 1970-1979 (C), 1980-1989 (D), 1990-1999 (E), and 2000-2010 (F).

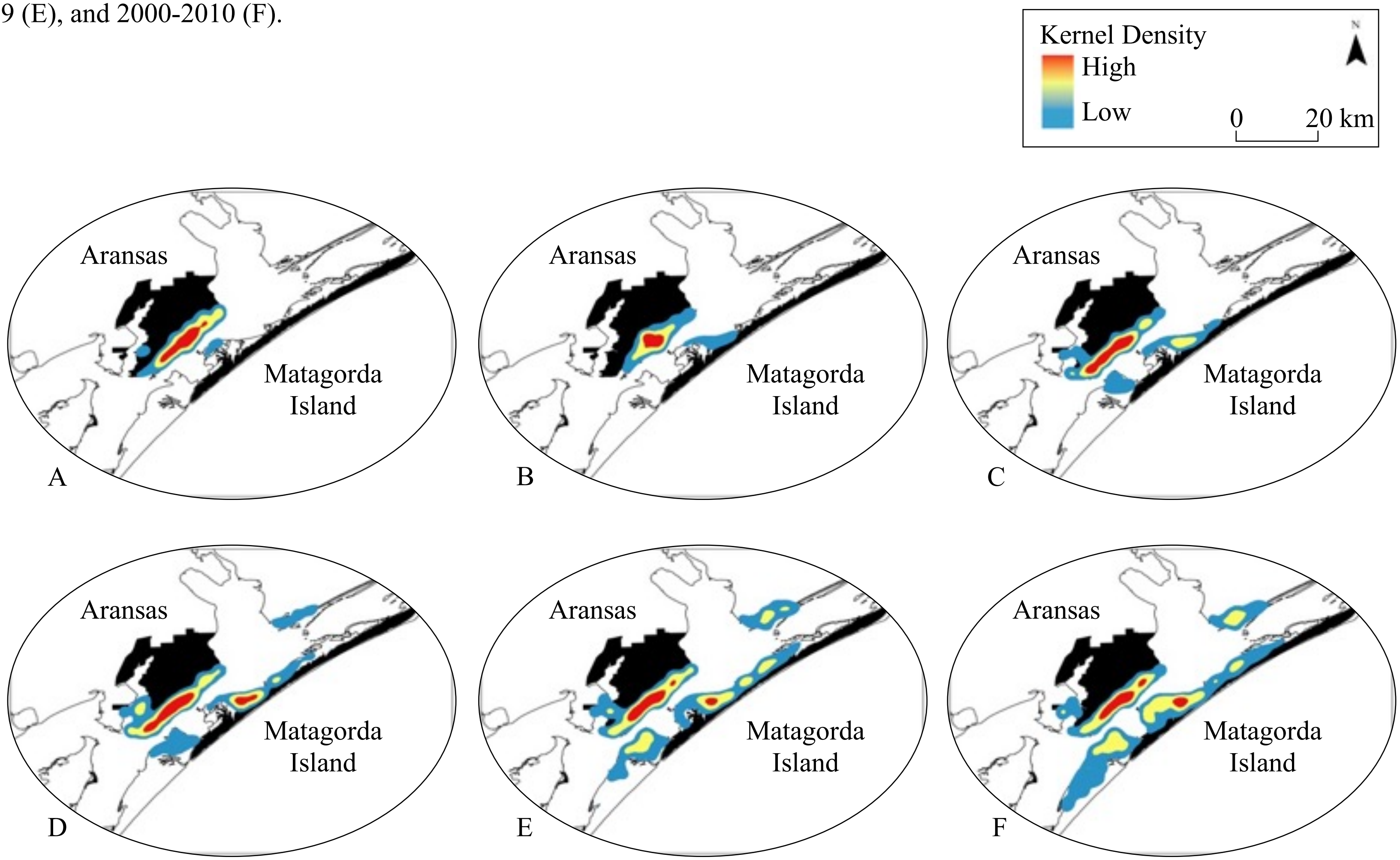
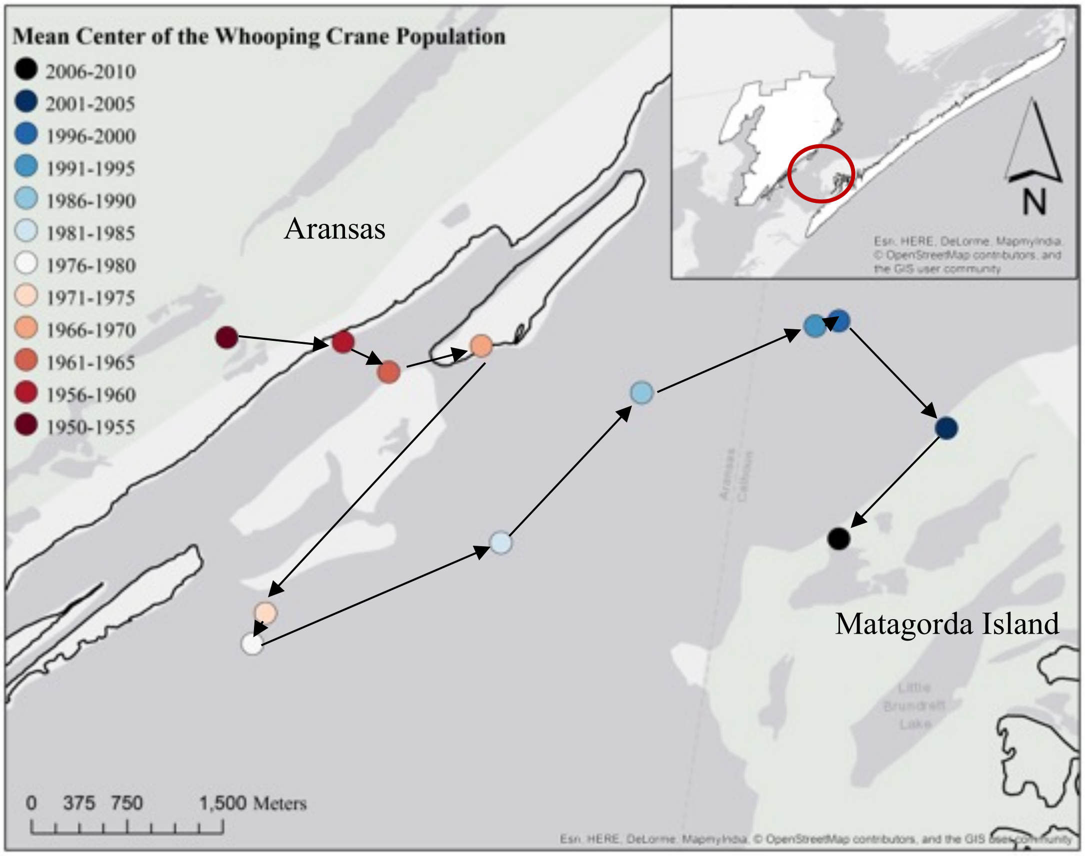


Figure 4. Mean center of the whooping crane population in five year increments between 1950-2010.



Appendix A

Table 1. Change in mean percent cover before and after summer and winter prescribed burns on control and treatment WREP research sites, Lincoln County, Oklahoma, USA $(\pm SE)$ $[\pm SD]$.

Species	Burn Season	Pre-burn 2014	Control 2014	Pre-burn 2015	Control 2015	Post-burn 2015	Post-burn 2016	Control 2016	Postburn 2017	Control 2017
Iva annua	Summer	5.028 (0.5364) [13.139]	7.569 (0.598) [18.900]	1.283 (0.245) [5.479]	2.643 (0.333) [8.811]	NA	9.779 (0.660) [16.155]	9.289 (0.696) [22.009]	0.000 (0.000) [0.000]	0.000 (0.000) [0.000]
Iva annua	Winter	0.273 (0.113) [2.263]	12.079 (0.812) [22.981]	NA	7.590 (0.667) [13.338]	0.820 (0.225) [3.187]	0.000 (0.000) [0.000]	5.890 (0.447) [10.954]	11.155 (0.816) [21.579]	5.591 (0.678) [13.559]
Persicaria Spp.	Summer	2.804 (0.418) [10.243]	17.766 (0.962) [30.430]	1.452 (0.289) [6.459]	1.697 (0.233) [6.161]	NA	6.562 (0.344) [9.153]	5.789 (0.344) [10.870]	0.614 (0.181) [2.678]	0.395 (0.161) [2.284]
Persicaria Spp.	Winter	1.314 (0.256) [5.117]	4.757 (0.388) [10.982]	NA	3.360 (0.393) [7.863]	5.965 (0.710) [10.034]	0.170 (0.087) [1.236]	0.608 (0.111) [2.713]	7.033 (0.564) [14.931]	2.838 (0.403) [8.052]
Eleocharis spp.	Summer	0.164 (0.0606) [1.483]	4.108 (0.410) [12.952]	0.162 (0.058) [1.308]	5.633 (0.617) [16.318]	NA	0.233 (0.076) [1.851]	5.021 (0.481) [15.206]	0.086 (0.074) [1.097]	0.535 (0.181) [2.560]
Eleocharis spp.	Winter	0.333 (0.131) [2.613]	11.620 (0.666) [18.831]	NA	22.573 (1.551) [31.013]	0.620 (0.259) [3.660]	0.000 (0.000) [0.000]	13.918 (0.900) [22.053]	5.159 (0.606) [16.034]	7.923 (0.861) [17.223]
Cardiospermum halicacabum	Summer	2.417 (0.290) [7.095]	1.400 (0.179) [5.671]	3.751 (0.357) [7.979]	2.535 (0.267) [7.076]	NA	3.482 (0.306) [7.506]	2.777 (0.238) [7.517]	24.450 (2.0226) [30.000]	20.560 (2.165) [30.613]

Species	Burn Season	Pre-burn 2014	Control 2014	Pre-burn 2015	Control 2015	Post-burn 2015	Post-burn 2016	Control 2016	Postburn 2017	Control 2017
Cardiospermum halicacabum	Winter	0.188 (0.0723) [1.447]	1.269 (0.1402) [3.967]	NA	0.865 (0.177) [3.533]	0.000 (0.000) [0.000]	0.015 (0.015) [0.212]	2.240 (0.213) [5.215]	0.261 (0.056) [1.484]	2.446 (0.401) [8.024]
Persicaria amphibia	Summer	9.004 (0.836) [20.468]	2.047 (0.368) [11.651]	9.118 (0.882) [19.733]	2.944 (0.498) [13.174]	NA	5.81 (0.510) [12.490]	1.689 (0.309) [9.725]	8.723 (1.251) [18.557]	1.030 (0.301) [4.258]
Persicaria amphibia	Winter	0.008 (0.008) [0.150]	0.756 (0.192) [5.419]	NA	1.305 (0.347) [6.939]	0.015 (0.015) [0.212]	0.390 (0.209) [2.958]	0.963 (0.228) [5.590]	0.181 (0.086) [2.283]	1.658 (0.375) [7.510]
Ambrosia trifida	Summer	5.583 (0.621) [15.218]	1.972 (0.232) [7.348]	0.006 (0.006) [0.134]	0.0256 (0.010) [0.277]	NA	3.023 (0.325) [7.965]	0.636 (0.101) [3.19]	0.000 (0.000) [0.000]	0.255 (10.139) [1.959]
Ambrosia trifida	Winter	0.000 (0.000) [0.000]	7.735 (0.624) [17.647]	NA	1.123 (0.405) [8.100]	0.000 (0.000) [0.000]	0.000 (0.000) [0.000]	2.468 (0.486) [11.915]	4.919 (0.470) [12.432]	0.943(0.191) [3.829]
Woody Species	Summer	0.005 (0.005) [0.122]	0.281 (0.080) [2.535]	0.024 (0.012) [0.268]	1.070 (0.229) [6.054]	NA	0.010 (0.007) [0.173]	0.822 (0.171) [5.400)	0.114 (0.076) [1.131]	0.270 (0.139) [1.969]
Woody Species	Winter	0.403 (0.174) [3.477]	0.0438 (0.029) [0.806]	NA	0.048 (0.041) [0.814]	0.0950 (0.081) [1.150]	0.160 (0.113) [1.596]	0.100 (0.047) [1.154]	0.010 (0.006) [0.165]	0.148 (0.061) [1.210]

Appendix B

Figure 1. Change in mean percent cover (±SE) of litter before and after a summer prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. A small decrease was observed one year post burn. Litter levels on post burn sites were comparable to control sites two years post-burn.

Figure 2. Change in mean percent cover (±SE) of litter before and after a winter prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. No decrease in litter was observed one year post-burn. Two years post-burn, litter levels were much higher than on control sites, and three years post-burn, litter levels were much lower than on control sites.

Figure 3. Change in mean percent of bare ground (±SE) present before and after a summer prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. The mean percent of bare ground present was much higher one year post-burn, but had returned to levels similar to control conditions two years post-burn.

Figure 4. Change in mean percent of bare ground present (±SE) before and after a winter prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. The mean percent of bare ground present was much higher than control sites for all three years following the burn.

Figure 5. Change in mean percent cover (±SE) of forbs before and after a summer prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. Mean percent forb cover was higher on post burn sites one and two years post-burn as compared to control sites.

Figure 6. Change in mean percent cover (±SE) of forbs before and after a winter prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. Forb cover increased one and three years post-burn and remained higher than mean forb cover on control sites.

Figure 7. Change in mean percent cover of *Carex* spp. (±SE) before and after a summer prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. Mean *Carex* cover was slightly higher one year post-burn compared to pre-burn levels and was similar to control sties one and two years post-burn.

Figure 8. Change in mean percent cover of *Carex* spp. (±SE) before and after a winter prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. Cover of *Carex* increased one year post-burn, but mean percent cover was lower than control sites three years post-burn.

Figure 9. Change in species richness (±SE) before and after a summer prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. Species richness was slightly higher one year post-burn, but was similar to control sites two years post-burn.

Figure 10. Change in species richness (±SE) before and after a winter prescribed burn on control and treatment WREP research sites, Lincoln County, Oklahoma, USA. There was very little difference between species richness pre-burn and one and three years post-burn.

Figure 1

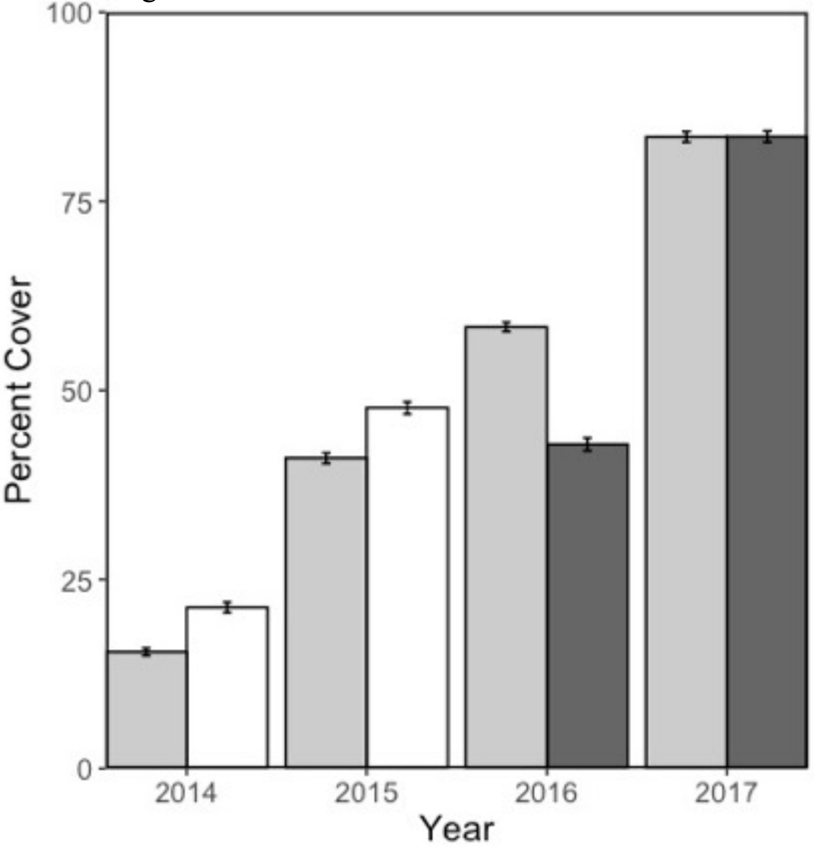


Figure 2

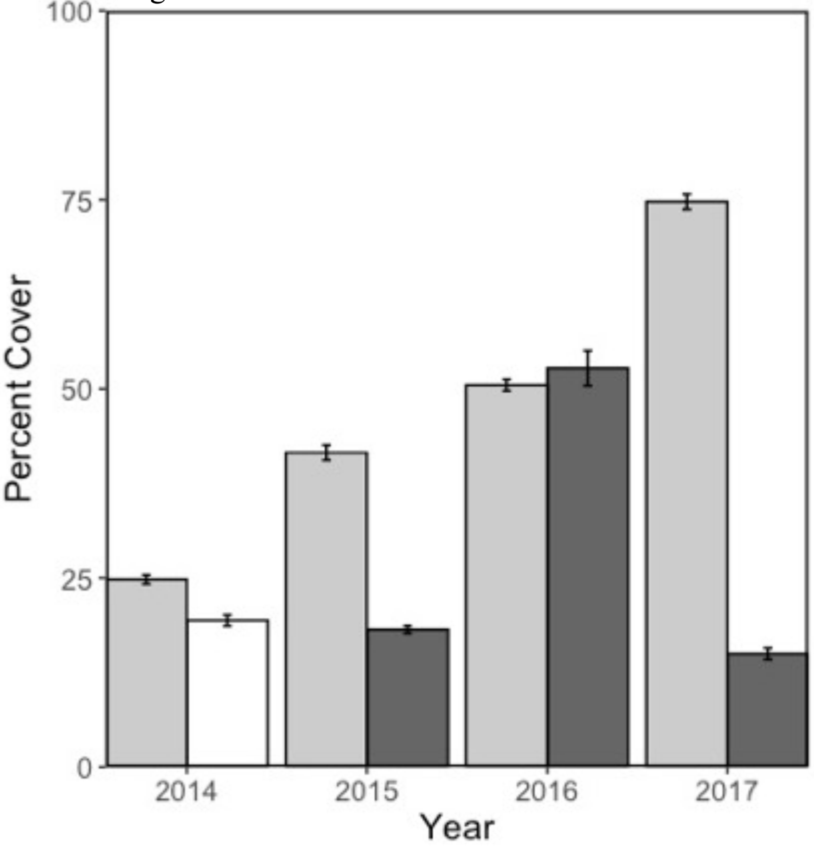
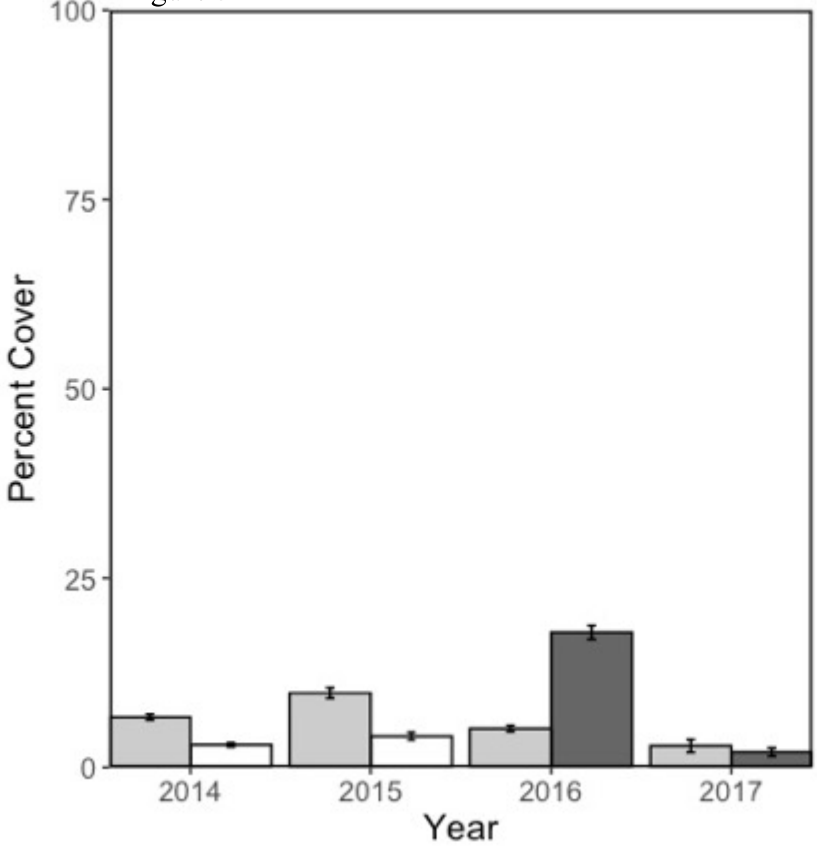


Figure 3



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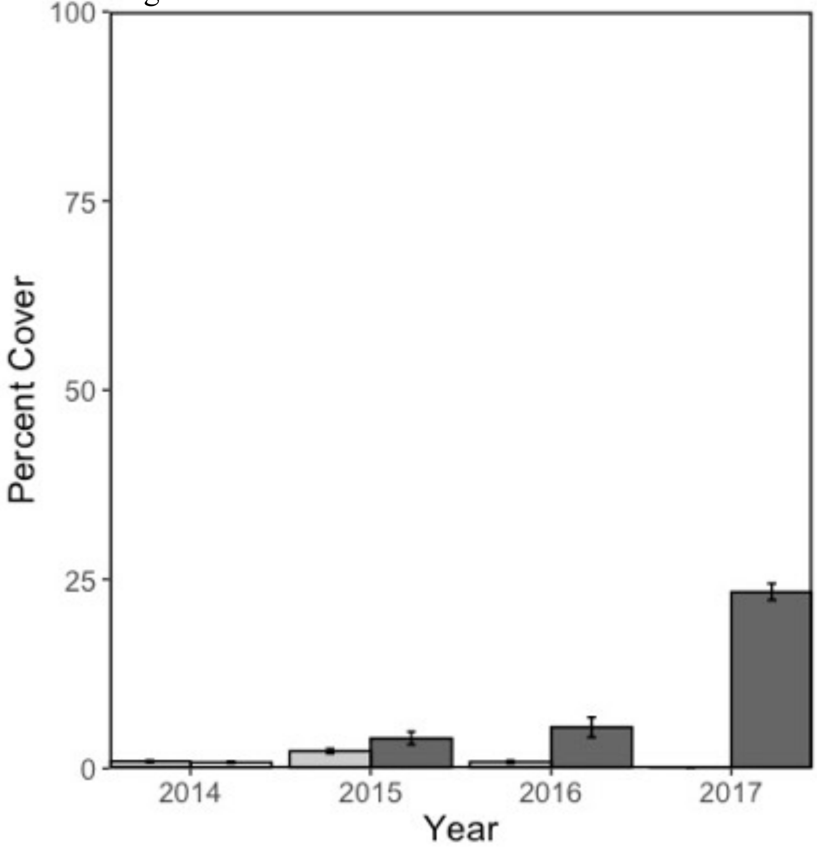

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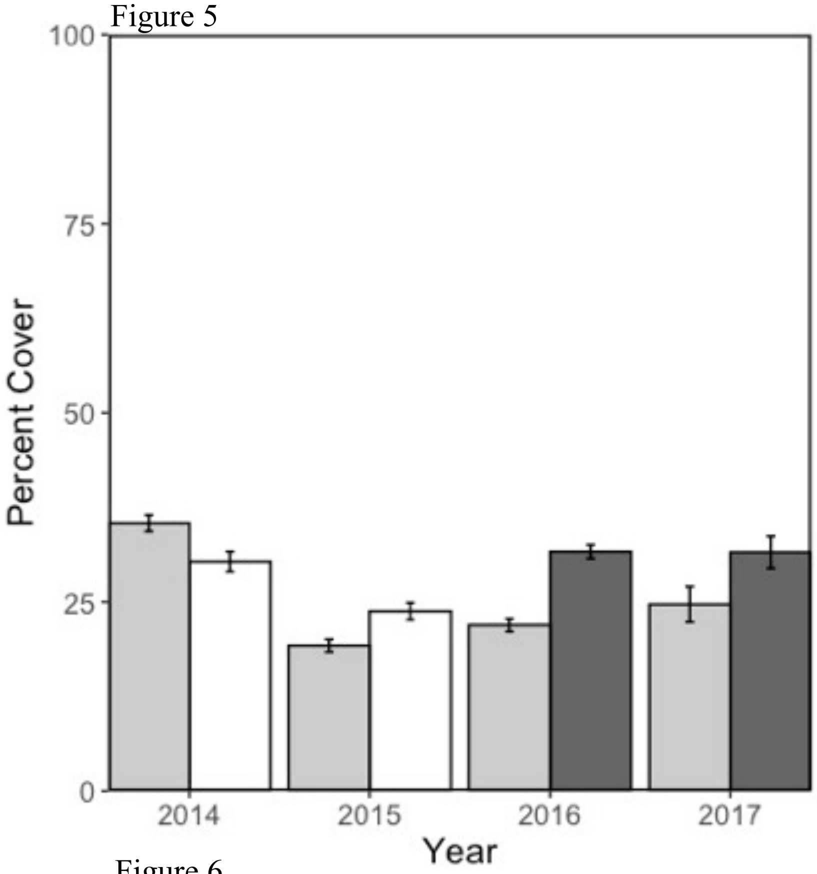

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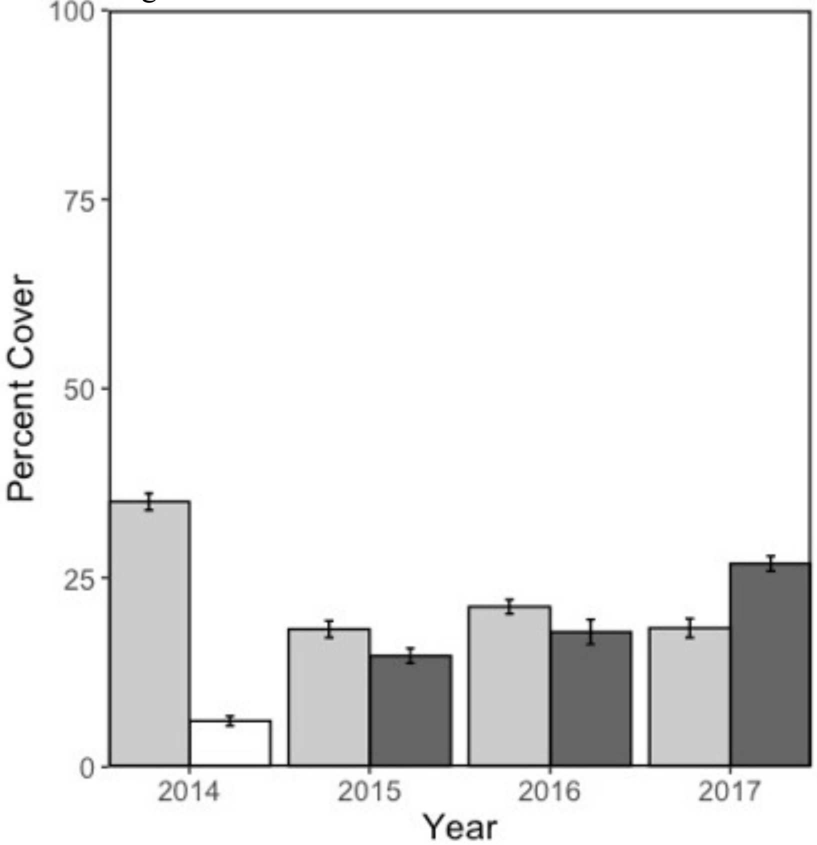

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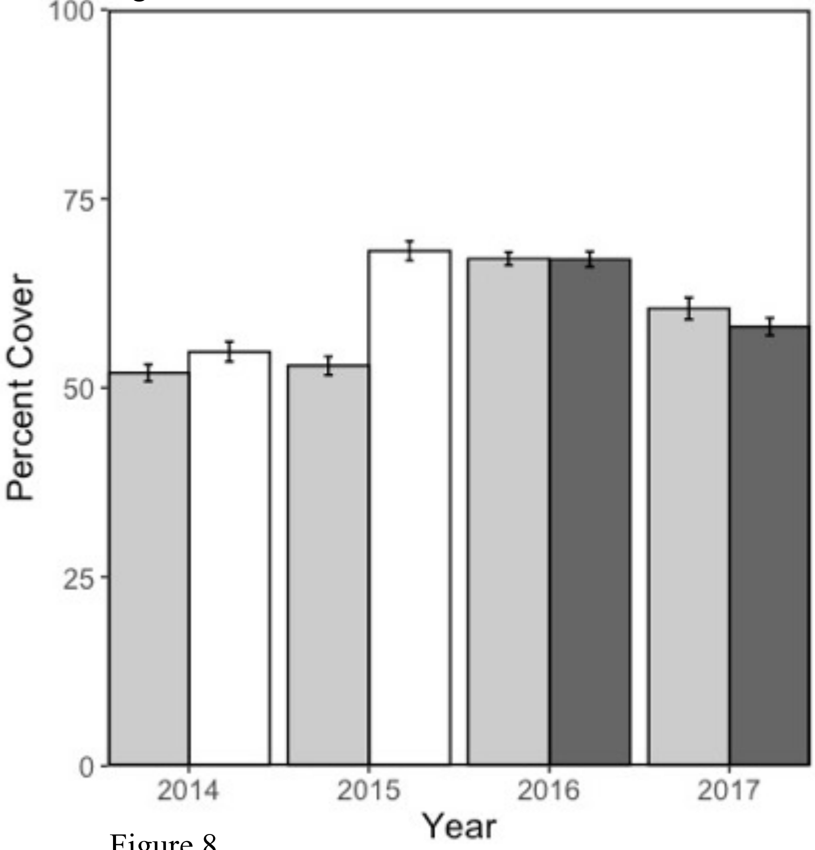


Figure 8

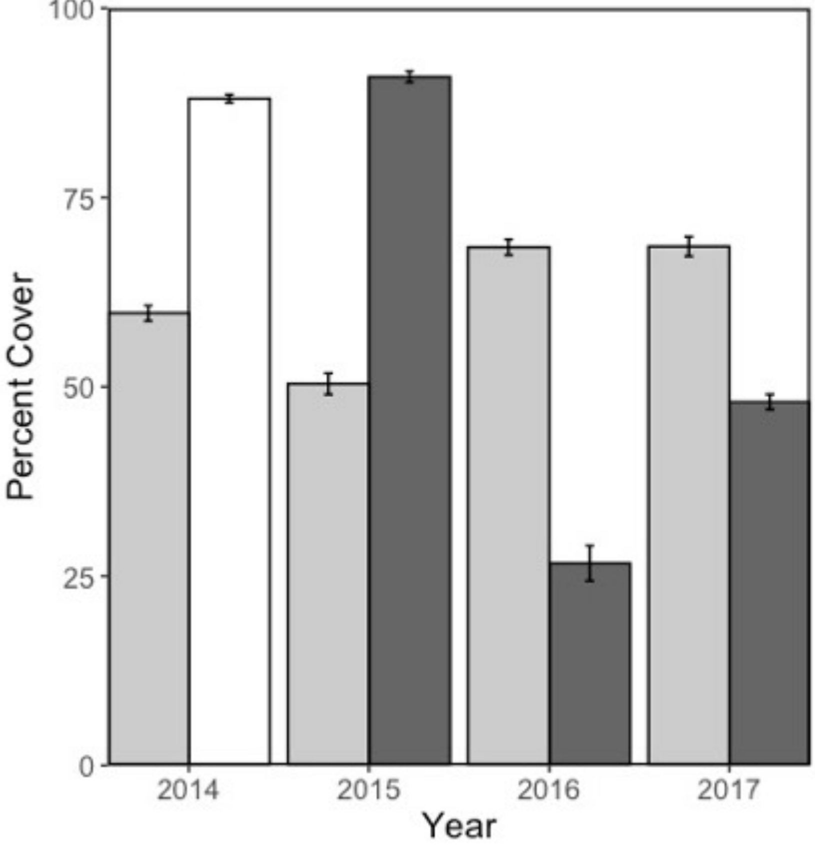


Figure 9

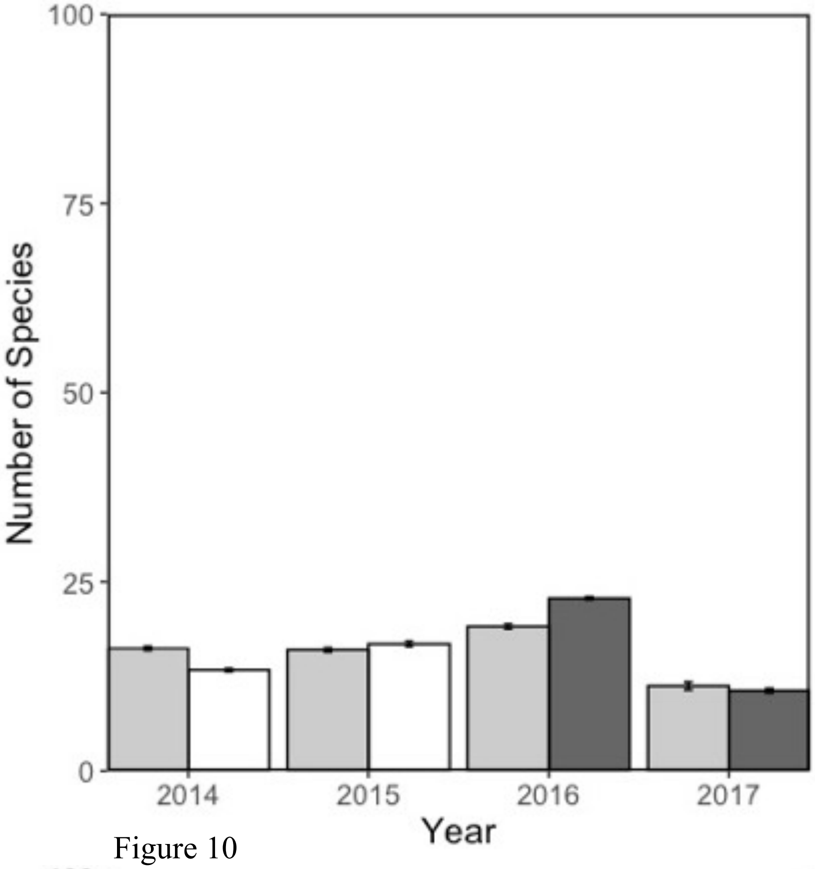
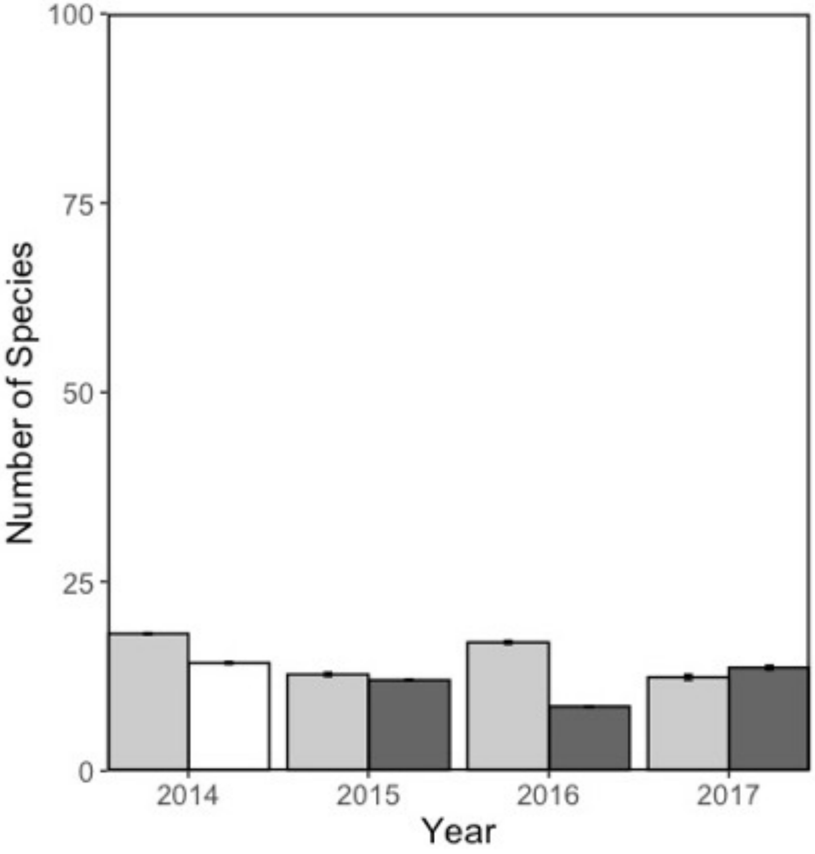


Figure 10



APPENDIX C

Table 1. Plant species documented at each of the research sites from spring 2014 through summer 2017.

Scientific Name	Common Name	Research Site 1	Research Site 2	Research Site 3	Research Site 4
Acer negundo	Box elder			x	
Alisma subcordatum	American water plantain			x	
Alopecurus carolinianus	Carolina foxtail	x	x	x	
Amaranthus tuberculatus	Roughfruit amaranth	x	x	x	
Ambrosia artemisiifolia	Common ragweed	x			
Ambrosia trifida	Great ragweed	x	x	x	x
Ammannia coccinea	Valley redstem	x	x	x	
Apocynum cannabinum	Hemp dogbane		x	x	
Asclepias syriaca	Common milkweed	x		x	
Azolla filiculoides	Pacific mosquitofern			x	x
Brassica juncea	Indian mustard		x		
Bromus inermis	Smooth brome			x	
Bromus racemosus	Bald brome	x	x	x	x
Bromus tectorum	Cheatgrass	x		x	
Cardiospermum halicacabum	Balloonvine	x	x	x	x
Carex spp.	Sedge spp.	x	x	x	x
Carex hyalinolepis	Shoreline sedge	x	x	x	x
Carya spp.	Hickory spp.	x		x	
Ceanothus cuneatus	Buckbrush	x			
Cephalanthus occidentalis	Common buttonbush	x	x	x	x
Chenopodium album	Common lambsquarters	x	x	x	x
Cichorium intybus	Chicory	x			
Conyza canadensis	Canada horseweed	x	x	x	x
Coreopsis tinctoria	Plains coreopsis		x	x	
Cuscuta spp.	Dodder spp.		x		x
Cynanchum laeve	Honeyvine milkweed	x	x		

Scientific Name	Common Name	Research Site 1	Research Site 2	Research Site 3	Research Site 4
Daucus carota	Wild carrot	x			x
Echinochloa colona	Jungle rice		x		
Echinochloa crus-galli	Barnyard grass	x			
Echinodorus berteroi	Upright burhead	x	x		
Eclipta prostrata	Eclipta	x	x	x	x
Eleocharis spp.	Spikerush spp.	x	x	x	x
Elymus virginicus	Virginia wild rye			x	
Erigeron strigosus	Daisy fleabane	x			
Eupatorium altissimum	Tall thoroughwort	x	x	x	
Eupatorium serotinum	Lateflowering thoroughwort	x	x	x	
Euphorbia marginata	Snow on the mountain	x	x		
Euphorbia prostrata	Prostrate sandmat	x	x		
Fraxinus spp.	Ash spp.			x	
Galium spp.	Bedstraw spp.	x	x	x	x
Geranium carolinianum	Carolina geranium	x	x		
Gleditsia triacanthos	Honey locust	x		x	
Helianthus annuus	Common sunflower	x	x	x	x
Hibiscus laevis	Halberdleaf rosemallow		x	x	x
Iva annua	Annual marsh elder	x	x	x	x
Juglans spp.	Walnut spp.	x			
Lactuca serriola	Prickly lettuce	x	x		
Lamium amplexicaule	Henbit	x	x		
Lathyrus hirsutus	Caley pea		x		x
Lemna spp.	Duckweed spp.	x	x	x	x
Lepidium spp.	Pepperweed spp.	x	x		
Lespedeza virginica	Slender lespedeza		x		
Ludwigia peploides	Floating primrose	x		x	x
Lythrum alatum	Winged lythrum	x	x	x	
Melilotus spp.	Sweet clover spp.			x	x
Melothria pendula	Guadeloupe cucumber	x		x	x
Myosurus minimus	Tiny mousetail	x			

Scientific Name	Common Name	Research Site 1	Research Site 2	Research Site 3	Research Site 4
Nelumbo lutea	American lotus				x
Packera plattensis	Prairie groundsel	x	x		
Parthenocissus quinquefolia	Virginia creeper	x		x	
Passiflora suberosa	Wild passionfruit	x			
Persicaria amphibia	Water knotweed	x		x	x
Persicaria hydropiperoides	Swamp smartweed	x	x		x
Persicaria lapathifolia	Curlytop knotweed	x	x		x
Persicaria pensylvanica	Pennsylvania smartweed	x	x	x	x
Peltandra virginica	Green arrow arum	x			
Phyla lanceolata	Lanceleaf fogfruit	x	x	x	x
Physalis angulata	Cutleaf groundcherry	x	x	x	x
Phytolacca Americana	American pokeweed	x			
Plantago rhodosperma	Redseed plantain		x		
Fallopia convolvulus	Black bindweed		x		
Polygonum ramosissimum	Bushy knotweed	x	x	x	x
Populus spp.	Cottonwood spp.	x			
Prunus spp.	Plum spp.	x			
Pyrrhopappus spp.	Chicory spp.	x			
Ranunculus sceleratus	Cursed buttercup	x	x	x	x
Rhus spp.	Sumac spp.	x			
Rorippa palustris	Bog yellowcress	x	x	x	x
Rudbeckia amplexicaulis	Clasping coneflower	x	x	x	
Rumex crispus	Curly dock	x	x	x	x
Sagittaria ambigua	Kansas arrowhead				x
Salix nigra	Black willow	x		x	x
Schizachyrium scoparium	Little bluestem	x			
Setaria pumila	Yellow bristlegrass	x			
Solanum physalifolium	Hairy nightshade	x			
Solidago canadensis	Canada goldenrod	x	x	x	

Scientific Name	Common Name	Research Site 1	Research Site 2	Research Site 3	Research Site 4
Sonchus asper	Spiny sowthistle		x		
Sorghum halepense	Johnsongrass	x	x		
Stellaria media	Chickweed	x			
Symphyotrichum subulatum	Eastern annual saltmarsh aster	x	x	x	x
Teucrium canadense	American germander	x		x	
Torilis arvensis	Spreading hedgeparsley	x		x	x
Toxicodendron radicans	Poison ivy	x		x	
Triticum aestivum	Common wheat		x		
Ulmus spp.	Elm spp.	x	x	x	
Utricularia spp.	Bladderwort spp.				x
Valerianella radiata	Beaked cornsalad	x	x		
Verbena urticifolia	White vervain			x	
Veronica peregrina	Neckweed	x	x	x	x
Vicia ludoviciana	Louisiana vetch			x	
Xanthium strumarium	Cocklebur		x		x
Zizaniopsis miliacea	Giant cutgrass		x		

APPENDIX D

Figure 1. LANDFIRE Fire Regime Group (FRG) classification on Aransas and Lamar.

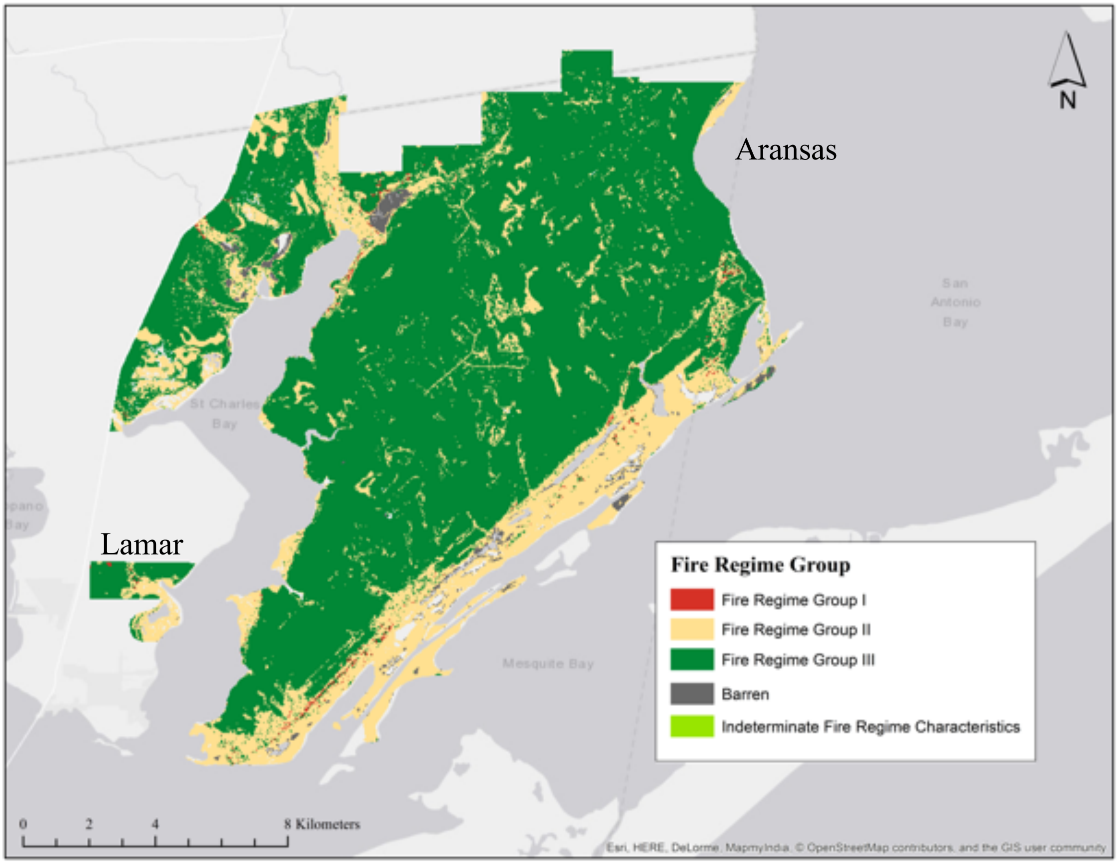


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Figure 2. Matagorda Island and the Myrtle Foester Whitmire Unit LANDFIRE Fire Regime Group (FRG) classification.

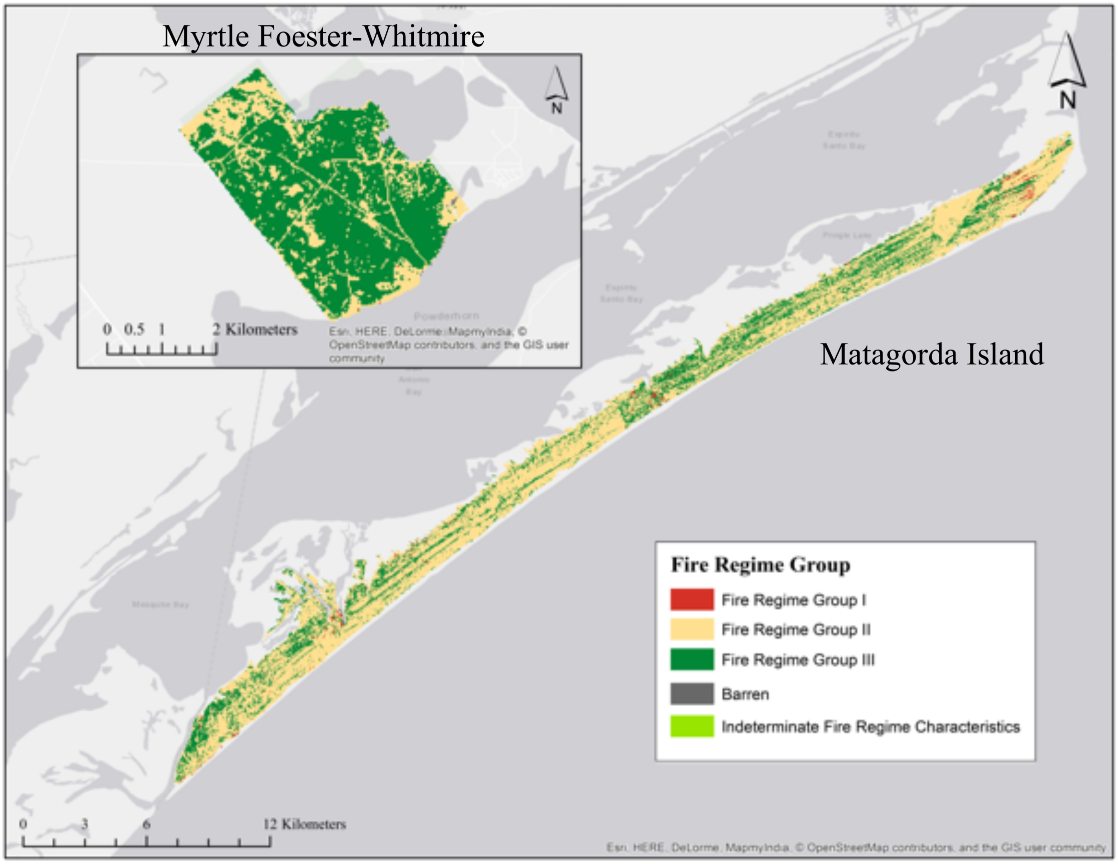


Figure 3. LANDFIRE Mean Fire Return Interval (MFRI) of historic wildfires on Aransas and Lamar.

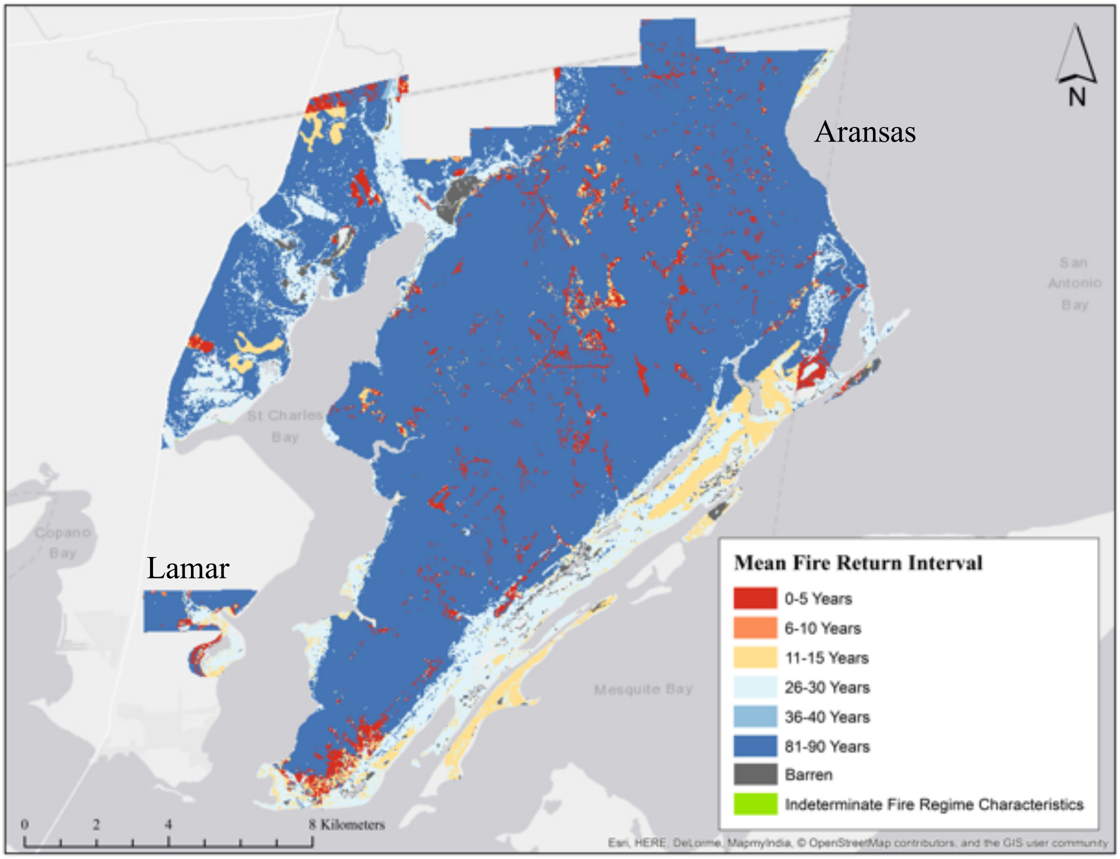
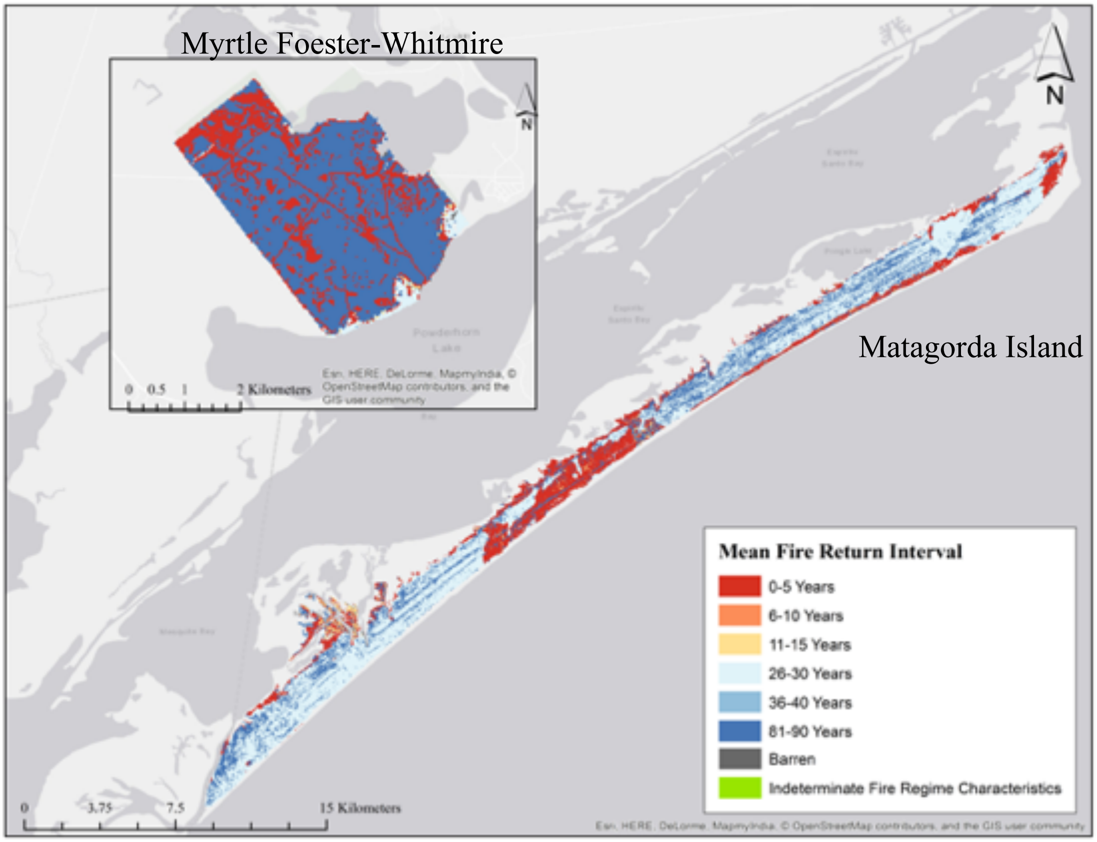


Figure 4. Matagorda Island and the Myrtle Foester Whitmire Unit LANDFIRE Mean Fire Return Interval (MFRI) of historic wildfires.



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APPENDIX E

Table 1. Habitat types grouped to created broad landcover variables based on vegetation types classified by NatureServe and MoRAP.

Habitat Type Category	Vegetation Types Classified by NatureServe and MoRAP
Bare ground	Active Sand Dune Barren Central and Lower Coastal: Beach
Water	Open Water
Woody Shrubland	Coastal and Sandsheet: Deep Sand Live Oak - Mesquite Woodland Coastal and Sandsheet: Deep Sand Live Oak Forest and Woodland Coastal and Sandsheet: Deep Sand Live Oak Shrubland Coastal and Sandsheet: Deep Sand Shrubland Coastal Bend: Floodplain Evergreen Shrubland Coastal Bend: Floodplain Hardwood Forest Coastal Bend: Floodplain Deciduous Shrubland Coastal Bend: Floodplain Live Oak - Hardwood Forest Coastal Bend: Floodplain Live Oak Forest Coastal Bend: Riparian Deciduous Shrubland Coastal Bend: Riparian Evergreen Shrubland Coastal Bend: Riparian Hardwood Forest Coastal Bend: Riparian Live Oak - Hardwood Forest Coastal Bend: Riparian Live Oak Forest Gulf Coast: Salty Prairie Shrubland Native Invasive: Deciduous Woodland Native Invasive: Huisache Woodland or Shrubland Native Invasive: Mesquite Shrubland Non-Native Invasive: Chinese Tallow Forest, Woodland, or Shrubland Non-native Invasive: Saltcedar Shrubland Post Oak Savanna: Live Oak Motte and Woodland Post Oak Savanna: Live Oak Shrubland Invasive: Evergreen Shrubland Native Invasive: Baccharis Shrubland South Texas: Clayey Blackbrush Mixed Shrubland South Texas: Clayey Live Oak Motte and Woodland South Texas: Clayey Mesquite Mixed Shrubland South Texas: Sandy Live Oak Motte and Woodland South Texas: Sandy Mesquite - Evergreen Woodland South Texas: Sandy Mesquite Dense Shrubland South Texas: Sandy Mesquite Woodland and Shrubland

Habitat Type Category	Vegetation Types Classified by NatureServe and MoRAP
Woody-shrub Wetland	Coastal and Sandsheet: Deep Sand Live Oak Swale Marsh Coastal: Mangrove Shrubland Coastal: Salt and Brackish High Tidal Shrub Wetland
Grassland	Coastal and Sandsheet: Deep Sand Grassland Coastal Bend: Floodplain Grassland Coastal Bend: Riparian Grassland Gulf Coast: Coastal Prairie Gulf Coast: Salty Prairie Native Invasive: Common Reed
Herbaceous-Grassland Wetland	Coastal and Sandsheet: Deep Sand Grassland Swale Marsh Coastal Bend: Floodplain Herbaceous Wetland Coastal Bend: Riparian Herbaceous Wetland Coastal: Salt and Brackish High Tidal Marsh Coastal: Salt and Brackish Low Tidal Marsh Gulf Coast: Coastal Prairie Pondshore
Tidal Flat	Coastal: Tidal Flat South Texas: Algal Flats Coastal: Sea Ox-eye Daisy Flats South Texas: Wind Tidal Flats
Row Crops	Row Crops
Urban	Urban High Intensity Urban Low Intensity

Appendix F

Cranes have been documented using upland areas post fire (Chavez-Ramirez et al. 1996) and we therefore wanted to include fire as a variable that could potentially affect habitat use by whooping cranes. ANWR has implemented a prescribed burn program on the refuge over the past three decades for the benefits of the cranes. We received a list of prescribed burns by date that were conducted on Aransas, Matagorda Island, and the Myrtle-Foester Whitmire Unit between 1985-2013. We used remotely sensed satellite imagery collected before and after each burn date to develop GIS layers of each individual burn on the refuge. We first had to obtain and correct the satellite imagery. We downloaded Landsat imagery and the raw Landsat imagery was calibrated from its raw brightness pixel values to top of atmosphere reflectance (Key and Benson 2005) to correct for sunlight and topographic illumination errors. The effects of atmospheric transmittance can then be corrected after conversion to top of atmosphere reflection of the images. As light enters the atmosphere, it can be scattered by aerosols and alter the reflectance (Jensen 2005). Dark object subtraction was performed to reduce and correct for atmospheric effects found in satellite imagery (Chavez 1988) and it does not rely on spectral ground readings, which are impractical to obtain for historical datasets or remote locations.

Normalized Burn Ratio

The Normalized Burn Ratio (NBR) is a spectral index used to delineate burned areas using Landsat Tm/ETM+ bands 4 (near-infrared) and 7 (short wave infrared). We used the method of determining the NBR to spectrally find the locations and sizes of the burns on the refuge. Healthy vegetation responds strongly in the near-infrared (NIR) portion of the electromagnetic spectrum. Destruction of plant cell structure by fire and reduced chlorophyll production results in decreased reflectance at this wavelength (Jensen 2005). The shortwave infrared (SWIR) portion of the electromagnetic spectrum also responds significantly to the changes in vegetation post burn. Fire-altered landscapes display reduced absorption in the SWIR due to the decrease of water content in the vegetation (Rogan and Yool 2001). These attributes of vegetation response to fire make infrared reflectance ideally suited for burn severity studies. The normalized burn ratio is defined as: $NBR = (B4 - B7) / (B4 + B7)$, where B4 and B7 are the reflectance values of bands 4 and 7 (Key and Benson 2005).

Differenced Normalized Burn Ratio

A differenced normalized burn ratio (dNBR) can be calculated to isolate pixels that indicate a burn from those that do not. The dNBR better differentiates low severity pixels from those that represent non-vegetated areas (Escuin et al. 2005). The differenced normalized burn ratio is calculated as: $\text{dNBR} = \text{NBR}_{\text{pre-fire}} - \text{NBR}_{\text{post-fire}}$. Resulting values range from -2.0 to 2.0, with positive values indicating pixels that represent burned areas. For our analysis, pixels with values greater than 0.15 were classified as burned. Biophysical reactions of vegetation to fire have been well documented in spectral responses of Landsat TM data (Jakubauskas et al. 1990); specifically, the spectral response to fire in the mid and near-infrared ranges has been successfully employed to assess fire severity (Patterson and Yool 1998). All of the burn map layers were created using ArcMap v10.4.1 (ESRI 2016) and ENVI 5.3 (Harris Geospatial Solutions, Inc. 2019).

VITA

Katherine Elizabeth Golden

Candidate for the Degree of

Doctor of Philosophy

Thesis: THE EFFECTS OF PRESCRIBED FIRE ON WETLAND ECOLOGICAL COMMUNITIES

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