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Re-sampling of Vegetation Survey Sites within Cape Sable seaside sparrow Habitat

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General Background

The Cape Sable seaside sparrow (CSSS) as well as the vegetation within its habitat are highly sensitive to natural and management-caused changes in both hydrologic and fire regimes. With a broad goal of assessing the response of marl prairie ecosystems to the Everglades restoration efforts, a study intended to characterize marl prairie vegetation and monitor its responses to hydrologic alterations and fire within CSSS habitat was conducted between 2003 and 2010 with funding from U.S. Army Corps of Engineers (USACE). In the first three years of the project (2003-2005), a detailed account of vegetation composition and structure was documented. Subsequently, during 2006-2010, sub-sets of sites in six sparrow sub-populations (A-F) were re-visited annually to assess vegetation dynamics over space and time. The sub-set sampled each year included both unburned and burned sites. Burned sites were sampled to assess the vegetation recovery process following fire, with surveys repeated 1, 2 and 4 years after fire. However, when extraordinary events such as hurricane-caused post-fire flooding provided an opportunity to learn more about vegetation response to fire-hydrology interactions, sites burned in 2005 were sampled annually up to five years after fire. In contrast, sites burned in 2008 were re-sampled for only two years after fire, primarily because funding for vegetation monitoring within the habitat was temporarily discontinued. In order for monitoring of vegetation response from the 2008 fires to be equivalent to those of 2005, and to study the impact of the fire-hydrology interaction on vegetation along wide range of hydrologic conditions, sites burned in 2008 were resampled in 2014, 6 years after fire, with funding from Everglades National Park.

This document summarizes the vegetation change pattern observed at the sites burned between 2003 and 2008, emphasizing the work accomplished in FY 2014 (Task Agreement # P13AC01271, Cooperative Agreement # H5000-06-0104). In FY 2014, the major activities included field work and data analysis. Field sampling was accomplished between May 3 and June 7, 2014.

The report is organized in two sections, each describing vegetation responses to changes in hydrologic and fire regimes within CSSS habitat. The first section describes post-fire vegetation recovery pattern at sites that were burned in 2003, 2005 and 2008. The sites experienced a wide range of hydrologic conditions in post-fire years, and were sampled annually for four to six years after fire. Data were examined with trajectory analysis, a method that allows quantification of the direction and rate of vegetation shift toward pre-defined targets or along the hydrology gradient, thus addressing the question of whether post-fire hydrology affects vegetation recovery. The second section highlights the decadal (2004-2014) vegetation change in Cape Sable coastal prairies, formerly habitat to the CSSS, and where Howell first discovered the sub-species in 1918; sampling sites are surrounded by wetlands subject to tidal waters.

1 Fire and flooding interactions: vegetation dynamics trajectories in the southern Everglades marl prairies, Florida, USA

1.1 Introduction

Disturbances are important processes that affect biological community characters and development. A post-disturbance community pattern is largely determined by interactions between pre-disturbance community characters and the nature of the specific disturbance (Foster et al. 1998). A single disturbance event may or may not cause a long-term change in the fundamental community characters (Stone 1998). However, a recurrence of the same type of disturbance or multiple interacting disturbances of different physical and/or biological forms often result in changes in community character different from the independent effects of each disturbance (Walker and Chapin III 1987; Paine et al. 1998; Collins and Smith 2006; Li et al. 2007; Donato et al. 2009; Pidgen and Mallik 2013). A community in the recovery process from disturbance is particularly sensitive to subsequent disturbances, which in some cases can result in a regime shift, i.e. a change in community state (Folke et al. 2004). In general, effects of compounding multiple perturbations are controlled by their sequential order, intensity, and spatio-temporal variability (Foster et al. 1998; Fukami 2001). The length of the intervals between disturbances may be determinative. For instance, disturbances recurring at short intervals may have deleterious effects on community composition and development through their synergistic influences on the process of vegetation recovery, often resulting in a distinct species composition (Ross et al. 2004; Donato et al. 2009).

In several ecosystems, both fire and flooding are natural perturbations that normally occur in sequence, with some interval between them. These disturbances are common in floodplains, coastal prairies, and seasonally flooded grasslands (Timoney et al. 1997; McKee and Baldwin 1999; Lockwood et al. 2003; Heinl et al. 2006). Specifically, in areas where the probability of wildfire is high at the onset of the rainy season, there is a likelihood that a wildfire will be closely followed by flooding, thus affecting the trajectories of post-fire vegetation recovery. The chances of such events are high in South Florida, where wildfires caused by natural lightning are frequent early in the rainy season (Slocum et al. 2007). Wetland plant communities of South Florida have evolved in response to the interplay of both fire and hydrologic regimes (Gunderson 1994; Lockwood et al. 2003; Duever and Roberts 2013). While the synergistic effects of fire and flooding on woody plant dynamics have recently been described (Ruiz et al. 2013), studies of the response of South Florida graminoid/herbaceous vegetation to disturbance have generally focused on flooding or fire separately (Forthman 1973; Busch et al. 1998; Armentano et al. 2006; LaPuma et al. 2007). Studies that did address fire and flooding together examined the response of individual species, such as sawgrass (*Cladium jamaicense* Crantz) (Herndon et al. 1991), muhly grass (*Muhlenbergia capillaris* (Lam.) Trin. var. *filipes* (M.A.Curtis) Chapm. ex Beal) (Snyder et al. 2004), or of two species together, sawgrass (*C. jamaicense*) and cattail (*Typha domingensis* Pers.) (Ponzio et al. 2004). In this study, we examined how post-fire plant community dynamics in seasonally-flooded grasslands were influenced by various levels of flooding, occurring at varying intervals after fire.

In a seasonally-flooded wetland, the rate and extent of post-fire vegetation recovery vary with vegetation type, soil characteristics, fire intensity, and pre- and post-fire hydrologic conditions (**Figure 1.1**). Vegetation after a single burn event in some wetlands returns to a pre-

burn state within 3-4 years (Pahl et al. 2003; LaPuma et al. 2007). However, in an area where vegetation is denuded either due to a single intense disturbance or multiple sequential disturbances, such as fire followed by flooding, the vegetation succession may result in changes in community characters by removing dominant species and facilitating the growth of opportunistic species. Sudden dieback of dominant species may occur when the aerial shoots are burned-off in a fire and are submerged by post-fire flooding, thus cutting off the oxygen supply to the rhizomes and resulting in death of the plants (Ball 1990; Herndon et al. 1991; Kirkman and Sharitz 1994; Ponzio et al. 2004). Plants can avoid such a drowning effect by growing enough to maintain their apices above the water level. The interval between subsequent disturbances, such as fire followed by flooding, may affect their ability to regrow, and thus determine the fate of the plants, ultimately affecting the trajectory of post- disturbance vegetation dynamics.

An understanding of plant community responses to the interacting multiple disturbances, natural or anthropogenic, is important for ecosystem management. In the Everglades, both prescribed fire and hydrologic operations are commonly used for managing different ecosystems. This is especially the case in the marl prairie landscape, a mosaic of seasonally flooded, short hydroperiod wetlands with calcitic marl substrates in the Southern Everglades, and the habitat of the federally listed endangered species, Cape Sable seaside sparrow (CSSS; *Ammodramus maritimus mirabilis*). CSSS populations are sensitive to frequency and intensity of both of these stressors, as they respond to changes in hydrology and fire regime directly, through their nesting success or failure (Pimm et al. 2002; Baiser et al. 2008), or indirectly, mediated through vegetation change in their habitat (Nott et al. 1998; Jenkins et al. 2003). Moreover, CSSS populations are also sensitive to the proportion of woody vegetation within its habitat (Pimm et al. 2002), which is determined by topography, soil characteristics, hydrology, fire, and their interactions (Hanan et al. 2010; Ruiz et al. 2013). This study will help natural resource managers to understand the nature of synergistic effects of both fire and flooding on CSSS habitat quality within marl prairie landscape and to avoid any undesirable consequences of management activities.

We examined plant community responses to multiple disturbances, i.e., fire followed by flooding, in Southern Everglades marl prairies. The specific questions were, i) Is the rate of post-fire vegetation recovery influenced by the hydrologic conditions present after fire?, and ii) Does the species composition in the post-fire community differ from pre-burn community composition more in flooded than not-flooded sites? We used trajectory analysis (Minchin et al. 2005; Sah et al. 2014) to examine the time course of post-fire vegetation recovery at the burned sites. Trajectory analysis allows researchers to test hypotheses about how community composition changes along an environmental gradient, or toward a pre-defined target, here pre-burn sites. We hypothesized that, i) vegetation recovery at sites that were flooded immediately after fire was slower than at sites that were not flooded, and ii) at a given time, dissimilarity in vegetation composition between pre- and post-fire communities was higher at the flooded than non-flooded sites. A deviation in the course of post-fire vegetation recovery will increase heterogeneity in the marl prairie landscape mosaic, and may affect habitat quality for the CSSS.

1.2 Methods

1.2.1 Study Area

The study area was located in short-hydroperiod marl prairies that flank Shark River Slough in the southern Everglades (**Figure 1.2**). In the marl prairies, soils are usually calcitic marls <15 cm thick, underlain by limestone of the highly porous Miami oolite formation (Randazzo and Jones 1997). Hydrologic conditions are functions of both climatic drivers and water management activities allied with the operations of a network of canals and water control structures. In the marl prairies, west of Shark River Slough, management-induced hydrologic regime is controlled through the operations of the S12 structures along the Tamiami Trail, and in the eastern Everglades, through the operations of a series of detention ponds and water pumps constructed along the L31W and L31N canals (Light and Dineen 1994; Kotun and Renshaw 2014). In the pre-drainage era, the marl prairies were flooded from two to nine months depending on topography and annual precipitation (Van Lent et al. 1993). In recent decades, water management practices in this area are considered the culprits for major ecological stressors, including shortened hydroperiod with increased drought severity in some locations, and extended hydroperiod with drying pattern reversals in others (Van Lent et al. 1993).

In South Florida, mean daily temperature ranges from 17 to 25ºC, and mean annual precipitation ranges from 120 to 160 cm, 60% of which falls during the core wet season that extends from June through September (Obeysekera et al. 1999). May and October are transitional months in which the mean precipitation varies across the years, depending on the start and end of the wet season (Duever et al. 1994). The marl prairie landscape supports a diverse vegetation mosaic comprised of wet prairies, sawgrass marsh and tree islands that are maintained by relatively short hydroperiods. The wet prairie vegetation is dominated or co-dominated by muhly grass, sawgrass, bluestem (*Schizachyrium rhizomatum* (Swallen) Gould) and black-top sedge (*Schoenus nigricans* L.), whereas sawgrass marsh is primarily dominated by sawgrass, but occasionally dominance is shared with beak rush (*Rhynchospora tracyi* Britton) (Ross et al. 2006). This marl prairie landscape within the Everglades National Park (ENP), Southern Glades Wildlife and Environment Area (SGWEA) and south-eastern part of Big Cypress National Preserve (BICY) supports the six sub-populations (A-F) of Cape Sable seaside sparrow (CSSS) (**Figure 1.2**).

Marl prairies are frequently burned, mostly as the result of natural lightning, but also due to anthropogenic fires in form of prescribed burns. Lightning tends to spark early in the wet season. In the eastern marl prairies that are most accessible to humans, incendiary fires are also common (Slocum et al. 2007). In recent years, seven major fires, three in 2005 and four in 2008, burned large sections of marl prairie within CSSS habitat (**Table 1.1**). In May 2005, the "Aerojet" fire (May_05) burned 76.7 ha in sub-population D, east of Everglades National Park, and in August of the same year the "Keyhole" and "Sisal" fires (Aug_05) burned 611 and 660 hectares, respectively, in sub-population B (**Figure 1.2**). The area burned in May remained unflooded for >1 month after fire, and for two months thereafter experienced a gradual increase in water level, while the areas burned in August were flooded by more than a foot (30 cm) of water by Hurricane Katrina (landfall in South Florida on Aug 25, 2015) within 7-15 days of fire. In 2008, Mustang Corner fire (MC_08) burned 15,941 ha, mostly marl prairie within CSSS sub-population F and E (Ruiz et al. 2013). The fire burned for one month, from May 14 to June 14, 2008. In late June of

the same year, the West Camp (WC_08) and Lime Tree (LT_08) fires burned 997 and 921 ha, respectively in sub-population A, and in the last week of July, 2008 the Radius Rod fire (RR_08) burned 387 ha in sub-population E.

Table 1.1: Major fires and number of Cape Sable seaside sparrow (CSSS) habitat vegetation sampling sites sampled before and after fires. Vegetation monitoring sites burned in Keyhole and Sisal fires were grouped together, as both fire occurred in August, and the sites burned in these fires were short-hydroperiod prairie sites and they were flooded within <1 month after fire.

Fire	Month/ Year	Fire/Sites name	Area (ha)	CSSS sub-populations						
				A	B	C	D	E	F	Total
Aerojet	05/2005	May_05	77				8			8
Keyhole	08/2005	Aug_05	611		5					5
Sisal	08/2005		660		8					8
Mustang Corner	05/2008	MC_08	15,941					4	40	44
West Camp	06/2008	WC_08	997	7						7
Lime Tree	06/2008	LT_08	921	9						9
Radius Rod	07/2008	RR_08	387					3		3

1.2.2   Data Collection

Vegetation sampling

Sample sites reported on here are part of a larger vegetation survey network established and first sampled in 2003-2005. The full network includes plots arranged at the interstices of a 1 x 1 km census grid that includes most of the marl prairie landscape occupied by the CSSS in Everglades National Park (ENP), as well as six transects (2.5 to 11.2 km in length) in CSSS subpopulations A-F, with plots distributed at 100 meter intervals, Plots were marked by 1 m rebar covered with tagged EMT (electric metallic tube) to facilitate repeated monitoring of vegetation structure and composition (Ross et al. 2006). To determine the CSSS vegetation sites that were within the fire boundaries, we used ArcGIS layers of fire perimeters provided by National Park, Fire and Aviation Division for six fires (Keyhole, Sisal, Aerojet, Mustang Corner, West Camp and Radius Rod) and by Big Cypress National Preserve (BICY) for the Lime Tree fire. The number of plots burned in individual fires varied from 3 in the Radius Rod fire to 44 sites in Mustang Corner fire. Out of 84 sites burned in these fires, 30 were transects sites (7 and 23 in sub-population D and F, respectively) and 54 were census sites. Within the habitat of sparrow sub-population F, 23 of 33 transect sites burned, whereas nine sites at the eastern end of the transect (TF-0000 to TF-0800) and one site in the middle (TF-1800) did not burn.

Prior to the fires, all 2005-burned sites, and 32 of 63 2008-burned sites were sampled only once between 2003 and 2005. The other 31 sites burned in 2008 were sampled twice, the first time between 2003 and 2005, and then between 2006 and 2008. After the fires, 2005-burned sites were re-sampled annually, between March and May, for five consecutive years, and 2008-burned sites were re-sampled in Years 1 and 2, and then 6 years after fire. On Transect F, when burned sites were re-sampled, the 10 unburned sites were also re-sampled.

At each sampling site, vegetation was sampled in a N-S oriented, 1 x 60 m rectangular plot beginning 3 m south of a rebar established to permanently mark the sampling site. Nested within the plots were ten 0.25 m^2 (0.5 x 0.5 m) subplots, arrayed at 6-meter intervals along the baseline (east side) beginning at Meter 5. In each subplot, we recorded our ocular estimation of cover (live + dead) of each species. We also noted any additional species present in the 1 x 60 m plot, and assigned these species a mean cover of 0.01% for the plot as a whole. In addition, a suite of structural parameters was recorded in 30 0.25 m^2 (0.5 x 0.5 m) subplots arrayed at every alternate meters beginning at Meter 1. Structural sampling included the following attributes: 1) Canopy height, i.e., the tallest vegetation present within a cylinder of ~5 cm width, measured at 4 points in each quadrat; 2) Total vegetative cover, in %; and 3) live vegetation, expressed as a % of total cover.

Hydrology

Hydrological variables used in this study were based on elevations determined from either topographic survey (for transect sites) or water depths measured in the field (for census sites). For transect sites, mean plot elevation was obtained by surveying from the nearest vertical control benchmark to each subplot of the transect sites. At the census sites, first we measured water depth at three locations within each 1x 60 m plots in the wet season of 2009, when sites in the region were inundated with standing water. Later, using the water surface elevations provided by Everglades Depth Estimation Network (EDEN) for the specific date, we calculated ground elevation for each plot. EDEN daily water surface elevation data (http://sofia.usgs.gov/eden/models/watersurfacemod_download.php) were then used to calculate annual mean daily water depth and hydroperiod for each site. Hydroperiod of each year was defined as the discontinuous number of days in a year when water level was above the ground surface.

1.2.3   Data Analysis

Trajectory Analysis

Vegetation data was first summarized by a non-metric multidimensional scaling (NMS) ordination, and differences in vegetation composition between pre-burn and post-burn years were examined using analysis of similarity (ANOSIM). For NMS ordination, cover data were relativized by species maxima. Post-fire vegetation response at individual sites was analyzed with trajectory analysis (Minchin et al. 2005), which was used to assess change in community composition along a vector representing time since last fire (TSLF) as well as along the vector toward a pre-defined target, here the pre-burn site response. To define the TSLF and hydrology vectors in the NMS ordination, we included the census and transect sites sampled between 2003 and 2005. The hydrology vector was derived by calculating plot level hydroperiod, using mean plot elevation, obtained by surveying from the nearest vertical control benchmark to each subplot of the transect sites and EDEN daily water surface elevation data. The TSLF vector was derived by calculating time since last fire using the fire database of the Everglades National Park.

In ordination space, the reference vectors for time since last fire and the hydrologic gradient were defined by the vector fitting technique in DECODA (Minchin 1998). In this method, a

gradient is defined in the direction through ordination which produces maximum correlation between the measured environmental attribute and the scores of the sampling units along the vector. The statistical significance of such correlations is tested using a Monte-Carlo permutation test with 10,000 random permutations, as samples in the given ordination space are not independent (Minchin 1998). The orientation of the ordination is then rotated so that hydroperiod has a perfect correlation (r = 1.0) with axis-1, the ordination's principal axis.

Two statistics, delta (Δ) and slope, were calculated to quantify the degree and rate of change in vegetation composition along the reference vectors (Minchin et al. 2005). Delta (Δ) measures the total amount of change in the target direction. Slope measures the mean rate of change in community composition along the target vector. In our analysis, the slope was calculated as the linear regression coefficient of projected scores on the target vector in sampling years since the sites were burned. The statistical significance of both delta (Δ) and slope was tested using Monte Carlo simulations with 10,000 permutations.

Change in species richness, evenness and biomass

Change in community characteristics (diversity and dominance) in response to fire and its interaction with post-fire hydrology was analyzed using rank-abundance plots in which both the species' relative abundance and richness are taken into account. The extent of dimensionality on the x-axis and slope of the rank-abundance curve represent the species richness and evenness, respectively. A curve with geometric shape and steep slope represents homogeneous community with only a few very dominant species. In contrast, an S-shaped (sigmoid curve) curve with lower slope represents a moderate level of heterogeneity in species composition (Whittaker 1965).

Additionally, vegetation structural measurements were summarized for each plot, and mean canopy height and total vegetative cover were used to estimate above ground plant biomass, using the allometric equation developed by Sah et al. (2007) for marl prairie vegetation within CSSS habitat. The equation for calculating biomass was as follows:

$$\sqrt{Biomass} = 6.708 + 15.607 * \text{arcsine} \sqrt{Cover/100} + 0.095 * \text{Ht}$$

where Biomass = Total plant biomass (g/m^2), Cover = Crown cover (%), and Ht = Mean crown height (cm).

Statistical analysis

Vegetation cover, biomass and species richness were measured repeatedly, once before fire and then three to five times after fire, in the same plots. Thus, to account for the variability caused by the repeated measures, General Linear Mixed Effect Models was used to investigate the impacts of explanatory variables, vegetation type (VEGTYPE: Marsh or Wet Prairie) and post-fire hydrologic conditions (WD_1M_Ave and WD_3M_Ave: water depth averaged over 1 and 3 months post-fire, respectively; and WD_3M_Max: maximum water depth in three months after fire) on cover and biomass over the sampling years. Generalized Linear Mixed Models was used to examine the effects of explanatory variables on species richness, a count variable. In both models, spatial autocorrelation among the sampling sites (SITE) within each fire was addressed

by incorporating nested random effect structures, in which SITE, as a random variable, was nested within FIRE, also a random variable on PREPOST (repeated sampling event). Repeated sampling event (PREPOST) was used as a fixed effect to examine the differences in biomass, cover and species richness between pre- and post-fire years. Models were run in R v.3.2.3 (R core team, 2015) using the *lmer* (for general linear mixed model) and *glmer* (for generalized linear mixed model) functions in the *lme4* package, and p-values for each effect were obtained using the Satterthwaite approximations to degrees of freedom (merModLmerTest) (Bates, D., 2014). In case the generalized linear mixed model failed to converge, the model was restarted from the previous fit and the number of iterations was doubled from 10,000 (default) to 20,000. Competing models were evaluated by comparing nested models with all possible subsets of predictors. The general linear mixed models were compared in the *pbkrtest* package with KRmodcomp function, whereas generalized linear mixed models were compared using *anova* function. When there was significant difference (F-test & $p < 0.05$) between two competing models, one nested in the other, Akaike Information Criteria (AIC) was used for model selection, and a model of higher rank order with all predictors that had significant effects was selected. Differences in biomass, cover and species richness between pre- and post-fire years were examined using *glht* function implemented in *multicomp* package.

The relationship between vegetation recovery five or six years after fire represented by delta ((Δ) and slope in trajectory analysis and post-fire hydrological conditions were examined using General Linear Model (GLM). In this case, however, the competing nested models were evaluated using the *anova* function.

Landsat TM image interpretation and vegetation recovery analysis

Changes in vegetation after fire were also examined using Normalized Difference Vegetation Index (NDVI). NDVI is widely used in ecological research, and is also considered a reliable estimator of vegetation change (Viedma et al. 1997; Abdel Malak and Pausas 2006; Hope et al. 2007). We calculated NDVI from Landsat 5 Thematic Mapper (TM) images (Path: 015, Row: 042). For sites burned in 2005, images were from pre-fire (2005) and five post-fire (2006 - 2010) years, while for the sites burned in 2008, images were from 2008 (pre-fire) and six post-fire (2009-2014) years. Images from all years except 2006, 2008 and 2010 were from the month of January or February. Since images with acceptable quality (cloud free) were not available for 2006 and 2008 for January-February, we used images from May 4, 2006 and April 13, 2008, respectively, as they were the first dry season cloud-free images for those years. However, for 2010, an image with acceptable quality was only available only for November 7, 2010, and thus the annual change in NDVI for the 2008-burned sites might have been affected. The spatial resolution of TM images was 30 m x 30 m. The images had been geo-referenced and atmospherically corrected for interference from atmospheric reflectance by the Remote Sensing and GIS Center of Florida International University. We used the corrected images and the computer program ArcGIS 10.2 to compute NDVI as: NDVI = (NIR-RED)/(NIR+RED).

For the 2005-burned sites, the difference between 2006 (Post-fire Year-1) and 2005 (Pre-fire) NDVI images was interpreted as the reduction in vegetation cover due to damage caused by the interaction of fire and flooding, while that damage due to fire for the 2008-burned sites was calculated as the difference between 2009 (Post-fire Year-1) and 2008 (Pre-fire) images. Likewise,

the NDVI difference between the first post-fire year and subsequent years was used to quantify vegetation recovery.

Tree islands are an integral component of marl prairie landscape. Since the objective of this research was to assess the response of only herbaceous vegetation to fire, while calculating NDVI within the perimeter of each fire, we used a tree island shape file for the respective area to mask the tree islands. Since, the minimum size of the tree island in the shape file was 36 m^2, any woody component having the patch size of <36 m^2 might have been included in the calculation and impacted the analysis. For the Mustang Corner fire, we used the tree island shape file developed by Ruiz et al. (2013). For the other fires, we developed the shape file from a set of 2004 1 m resolution color infrared (NIR, Red, and Green) images from US Geological Survey (USGS) National Aerial Photography Program (NAPP), and applying the method described in Ruiz et al. (2013).

1.3 Results

In the recently burned marl prairie landscape, vegetation composition varied spatio-temporally as a result of the variation in fire and hydrological regimes. Results of non-metric multidimensional scaling (NMS) ordination revealed that the sites burned in 2005 and 2008 fires were roughly arranged along gradients of hydrology and time since last fire in ordination space, (**Figure 1.3**). The pre-burn sites were noticeably arranged along the hydrologic gradient. Between two groups of sites burned in 2005, May_05 sites were towards the wetter end of the gradient than the Aug_05 wet prairie sites. Similarly, among the 2008-burned groups, the LT_05 sites were at the wettest end of the gradient, while MC_08 and WC_08 sites had wet prairie vegetation, but with wide range of hydrologic conditions. The sites burned in the Radius Rod fire in sub-population E had intermediate hydrologic conditions. In general, sites with vegetation adapted to relatively short hydroperiod had higher species richness than wetter sites, and plant species diversity was high at the most recently burned sites (**Figure 1.3**). Species diversity decreased as the total vegetation cover increased with time since the sites were burned.

Pre- and post-burn species composition

After the fire, vegetation composition in both May_05 and Aug_05 groups were very different from pre-burn vegetation. In both groups, vegetation composition even five years after fire differed significantly from pre-burn vegetation (ANOSIM: May_05 - R = 0.633, p = 0.002; Aug_05 - R = 0.437, p = 0.001) (**Table 1.2**), and mean total plant cover at the sites in these two groups were only 43.0% and 33.8% of the initial cover, respectively (**Figure 1.4 a-b**). Slow recovery of vegetation cover at these sites probably resulted from post-fire hydrologic conditions, as the majority of sites burned in 2005 experienced substantial flooding after fire. In contrast, within the majority of the 2008-burned groups, vegetation composition even two years after fire was not significantly different from pre-burn, suggesting rapid vegetation recovery (**Table 1.3**). Surprisingly, at sites burned in the Mustang Corner fire (MC_08), where post-fire conditions immediate after fire were not especially wet, vegetation composition even six years after fire was significantly different (ANOSIM: R = 0.177, p = 0.001) from pre-burn vegetation. In the same vicinity, the vegetation composition at the unburned sites on Transect F, did not differ among sampling years. As expected, total plant cover at the MC_08 and other 2008-burned sites recovered nearly to the pre-fire levels in six years after fire (**Figure 1.4 c-f**).

Fire usually impacts community composition by reducing the abundance of dominant species and facilitating the growth of light-demanding opportunistic species. This effect is well illustrated in rank-abundance plots that include both the relative abundance of species and evenness. At the sites burned in 2005, the relative cover of dominant species was considerably lower even five years after fire compared to pre-fire levels, resulting in large shifts in species rank abundance curve (**Figure 1.5a, b**). At those sites, especially in the Aug_08 group, the curve was significantly different from and less steep than the pre-burn curve, indicating that the community had become more heterogeneous. In contrast, in the 2008-burned groups, there was not much difference in curves between pre-burn and post-fire years, except, at the MC_08 sites which showed a significant shift one year after fire. However, by six years after the fire, the curve had returned to its pre-burn condition (**Figure 1.5c**).

Table 1.2: Global R and *p-values* from analysis of similarity (ANOSIM) testing for among-year differences in vegetation composition before and after fire for two 2005-burn groups, May_05 and Aug_05. (*p-value*: *** = 0.001, ** = 0.01, * = 0.05)

	May_05				
	Pre-burn	**Post_Yr-1**	**Post_Yr-2**	**Post_Yr-3**	**Post_Yr-4**
Post_Yr-1	**0.511***				
Post_Yr-2	**0.609***	**0.347**			
Post_Yr-3	**0.546***	**0.444***	0.027		
Post_Yr-4	**0.719***	**0.462***	0.031	-0.016	
Post_Yr-5	**0.633***	**0.473***	0.012	-0.107	0.006
	Aug_05				
Post_Yr-1	**0.732***				
Post_Yr-2	**0.686***	**0.287**			
Post_Yr-3	**0.516***	**0.281**	0.062		
Post_Yr-4	**0.444***	**0.322***	**0.140***	0.027	
Post_Yr-5	**0.437***	**0.371***	**0.119***	-0.008	-0.027

Table 1.3: Global R and *p-values* from analysis of similarity (ANOSIM) testing for among-year differences in vegetation composition before and after fire for two 2008-burn groups, MC_08, WC_08 and LT_08. RR_08 group had very low number (n=3) of sites. NB group includes the sites that did not burn, but were sampled with other 2008-burn sites (*p-values*: *** = 0.001, ** = 0.01, * = 0.05)

MC_08 (n=44)				**LT_08** (n=9)		
	Pre-burn	**Post_Yr-1**	**Post_Yr-2**		**Pre-burn (Yr-0)**	**Post_Yr-1**	**Post_Yr-2**
Post_Yr-1	**0.292***			**Post_Yr-1**	**0.138***		
Post_Yr-2	**0.201***	**0.125***		**Post_Yr-2**	0.085	0.027	
Post_Yr-6	**0.140***	**0.192***	**0.177***	**Post_Yr-6**	0.012	**0.118***	0.088
WC_08 (n=7)				**NB** (n=10; Pop-F)		
Post_Yr-1	**0.259***			**Yr-1**	-0.028		
Post_Yr-2	0.011	-0.006		**Yr-2**	-0.056	-0.089	
Post_Yr-6	-0.090	0.017	-0.155	**Yr-6**	0.012	-0.069	-0.031

The change pattern in relative abundance of species differed among different groups of sites. At Aug_05 sites, relative cover of four dominant species, i.e., sawgrass (*Cladium jamaicense*), bluestem (*Schizachyrium rhizomatum*), muhly grass (*Muhlenbergia capillaris* var. *filipes*) and black-top sedge (*Schoenus nigricans*) decreased significantly immediately after fire followed by flooding, and remained much lower than before the fire even five years later (**Figure 1.6b**). Persistence of the relatively low cover of these dominant species in post-fire years at the Aug_05 sites also facilitated the growth of other species. Relative cover of several minor species, such as spadeleaf (*Centella asiatica* (L.) Urb.)*,* southern beakrush (*Rhynchospora microcarpa* Baldwin ex A. Gray)*,* gulfdune paspalum (*Paspalum monostachyum* Vasey ex Chapm.) and bluejoint panicgrass (*Panicum tenerum* Beyrich ex Trin.) was higher in the fifth year after fire than in pre-burn samples. Interestingly, at May_05 sites also, where water level increased gradually, providing ample opportunity for the re-growth of plants after fire, a large decrease in the relative cover of sawgrass (*C. jamaicense*) was observed. Five years after the fire, the mean relative cover of sawgrass was only 55% in comparison to 90% one year before the fire. At these sites, the relative cover of beakrush *(Rhynchospora tracyi)*, black-top sedge *(S. nigricans),* spikerush (*Eleocharis cellulosa* Torr.), and southern beakrush (*R. microcarpa*) were significantly higher five years after fire than pre-fire. The relative cover of muhly grass (*M. capillaris* var. *filipes*), a C_4 grass, also increased immediately after fire, but decreased in the 4th and 5th post-fire years. In contrast to 2005-burned sites, dominant species in 2008-burned sites experienced only a minimal (<20%) decrease in cover (**Figure 1.7a-d**). In MC_08 and RR_08 group of sites, a decrease of 10 and 20% of pre-burn cover of *M. capillaris* var. *filipes* and *S. rhizomatum* was in response to the increased hydroperiod in some of those areas. Decrease in cover of these two dominant species that are indicators of relatively dry conditions was supplemented by an increase in cover of more hydric species, including *R. tracyi* and *Phyla nodiflora* (L.) Greene at MC_08 and *C. jamaicense* at RR_08 sites. Similarly, on unburned sites along Transect F, there was also a decrease in muhly grass over time (**Figure 1.8c**), suggesting an influence of hydrology on vegetation composition in that area. The same pattern was observed at LT_08 sites, located in western part of population A, but at these sites *C. jamaicense* cover also decreased, while the cover of other hydric species increased.

Biomass and species richness

In both 2005 and 2008-burned groups, the change in relative cover of dominant species, augmented by the growth of opportunistic species in post fire years, resulted in temporal variation in biomass and mean species richness. GLMM results indicated that biomass at the burned sites was significantly affected by post-fire water conditions, especially mean water depth in the first 1-3 months (**Table 1.4**). When biomass was averaged over all sites, the difference between pre-burn and 3+ years after fire was not significant, whereas mean post-fire three-month average water depth had a significant effect on biomass recovery (p = 0.006). Although, biomass averaged over all sites appeared to have returned to the pre-burn level in four to six years after fire, biomass at the 2005-burned sites was significantly lower even five years after fire (**Figure 1.9 a, b**). In contrast, aboveground biomass returned to pre-burn levels within 2-6 years after all of 2008 fires except MC_08 (**Figure 1.9 c-f**).

Table 1.4: Estimate/direction and standard error for each fixed effect from General Linear Mixed Effect modeling of biomass at 85 sites with Preborn and 3-5 sampling events within 5-6 years after fire. Estimates and SE are reported as biomass in g m^{-2}. Intercept represents the estimate for preborn, and values for different post-burn years represent the magnitude and direction of deviation in estimate from preborn. WD_3M_Ave, a continuous predictor, is the relative water level averaged over three months after fire.

Fixed effects:	Estimate/ Direction	SE	p-value
(Intercept)	539.5	21.7	**0.000**
Pre_post_Yr1	-294.5	39.3	**0.000**
Pre_post_Yr2	-204.6	61.7	**0.013**
Pre_post_Yr3	-225.4	37.5	0.086
Pre_post_Yr4	-169.7	50.9	0.103
Pre_post_Yr5	-152.7	50.9	0.142
Pre_post_Yr6	-120.4	29.5	0.060
WD_3M_Ave	-1.1	0.4	**0.006**

Prior to the fire, species richness varied among sites depending on the wetness. For instance, the May_05 sites were relatively wet sites (marsh), and in pre-fire year they had 12 species plot^{-1}, less than half of the number of species (27 species plot^{-1}) present at Aug_05 sites, which experienced drier water conditions (mostly wet prairie sites) (**Figure 1.10 a, b**). The richness declined by 25% in 2005 burn sites, and such decline was more obvious at wet prairie sites than marsh sites (**Table 1.5**). In contrast, when averaged over all 2008-burn sites, richness increased by 4% in the first year after fire, and did not differ between wet prairie and marsh sites.

Table 1.5: Estimate/direction and standard error for each fixed effect from Generalized Linear Mixed Effect modeling of species richness at the sites burn in 2005 and 2008 with Preborn and 3-5 sampling events within 5-6 years after fire. Intercept represents the estimate for preborn, and values for different post-burn years represent the magnitude and direction of deviation (in odd ratio) in estimate from preborn. M_WP represent marsh and wet prairie sites. WD_3M_Ave, a continuous predictor, is the relative water level averaged over three months after fire.

Fixed effects:	2005-Burn					2008-Burn				
	Estimate/ Direction	SE	Z-Value	p-value	exp (Est)	Estimate/ Direction	SE	Z-Value	p-value	exp (Est
(Intercept)	3.547	0.302	11.8	**<0.001**	34.69	2.912	0.166	17.5	**<0.001**	18.40
Pre_post_Yr1	-0.285	0.104	-2.7	**0.006**	0.75	0.041	0.088	0.5	0.646	1.04
Pre_post_Yr2	-0.122	0.092	-1.3	0.185	0.89	0.035	0.062	0.6	0.565	1.04
Pre_post_Yr3	-0.037	0.076	-0.5	0.625	0.96					
Pre_post_Yr4	0.008	0.088	0.1	0.928	1.01					
Pre_post_Yr5	-0.035	0.085	-0.4	0.684	0.97					
Pre_post_Yr6						0.077	0.057	1.3	0.178	1.08
M_WP (WP)	-0.287	0.274	-1.0	0.295	0.75					
WD_3M_Ave	-0.053	0.016	-3.3	**0.001**	0.95					
M_WP(WP): WD_3M_Ave	0.043	0.019	2.3	**0.022**	1.04	-0.015	0.002	-6.2	**<0.001**	0.99

The increase in species richness was mainly in two groups MC_08 and RR_08 (**Figure 1.10 c, f**), in which richness increased significantly in the first year after fire, then declined over time to the pre-burn level. At sites in which mean species richness declined after fire, the time required for recovery to the pre-burn level depended on the pre-burn site conditions (marsh or wet prairie sites) and post-fire hydrology. This was especially evident in 2005 burn sites, which exemplified the significant interaction between site type and post-fire hydrology.

Trajectories of vegetation change

The post-fire vegetation change pattern was also analyzed using trajectory analysis. In the analysis, the pre-burn samples were positioned in ordination space near the high end of the TSLF vector, and the burned sites that approached the pre-burn condition were likely to show a significant shift along individual vectors towards the respective pre-burn sites. The two sites burned in 2003 and sampled annually for four post-burn years resembled their pre-burn state by the 4th year after fire (**Figure 1.11a**), whereas the degree of a shift in position of sites in 2005- and 2008- burned groups varied (**Figure 1.11b**). The shift in positon of the sites back toward their reference position, expressed as the amount (Δ) and rate (slope) of change in vegetation composition, was significant for >30% sites in MC_08 and WC_08 groups (**Figure 1.12; Appendix 1**). In contrast, none of the sites in May_05 and Aug_05 groups showed a significant shift toward the target, which was the pre-burn site within the ordination space. However, five years after the fires, both mean degree (delta) and rate (slope) of change in vegetation composition were higher in the May_05 than in Aug_05 sites.

The rate of post-fire vegetation change at individual sites flooded immediately after fire was influenced by post-fire hydrologic conditions. For both the 2005 and 2008-burned sites, we used EDEN water surface elevation data to quantify the real time water depth when sites were burned, and during post-fire vegetation regrowth. Real time mean relative water level (RWL, water elevation in relation to ground elevation) was the average of daily mean water level for three consecutive days, starting from one-day before the burn date. Using the daily mean water level at each site, we then also calculated mean and maximum RWL for various post-fire periods: 1, 3 and 6 months, 1, 2, 3, 4 & 5 years after burn date. Vegetation recovery at the burned sites was negatively affected by water conditions at the time of burn, and the effects were much more pronounced at the prairie sites. The sites which had the water level near or above the ground level showed slow recovery (**Figure 1.13 a, b**). Post-fire water conditions were likewise very influential. The recovery process at the prairie sites was impeded when mean post-fire water depth was 20 cm or more over one to three months after fire, and maximum water level exceeded 30 cm during the same period (**Figure 1.14a, f**). Within three months, when RWL exceeded >50 cm, it had deleterious effect on vegetation recovery even at the marsh sites (**Figure 1.14g**), where mean RWL was <20 cm for the first month after fire (**Figure 1.14a**), but later increased and the sites remained flooded with more than 30 cm of water for almost 6 months (**Figure 1.14c**). The relationship between vegetation trajectory parameter and RWL, averaged over 2 years (**Figure 1.14e, j**) and beyond (not shown) was non-significant, suggesting that the water conditions for the first post-fire year were more important for vegetation recovery than the later years.

At several sites, the slope in trajectory analysis was negative (**Appendix 1**), suggesting that vegetation composition at those sites were currently on an opposite trajectory than normal, which

might have led to a vegetation state different from that which predominated prior to burn. At many of those sites BC similarity in vegetation composition was 40% or lower even after 5-6 years after fire (**Figure 1.15**).

Change in NDVI

The analysis of NDVI change revealed that vegetation recovery pattern varied over the 5-6 year post-fire period in both 2005 and 2008 fires (**Figure 1.16**). While the mean NDVI in Year-1 after fire was relatively low in both 2005 fires (0.219 and 0.200 in May_05 and Aug_05, respectively), the decrease in NDVI from pre-fire to 1st post-fire year was significantly greater in Aug_05 (47.7%) than May_05 sites (38.7%), suggesting that damage to vegetation from fire was more severe in the area, which was burned and immediately flooded (**Figure 1.17**). In MC_08 the decrease was only 23.5%, while in RR_08, no change in NDVI was observed. In subsequent years, vegetation recovery was relatively fast in all groups, but Aug_05. In all other groups, NDVI reached the pre-burn level within 1-2 post-fire years. In contrast, mean NDVI in Aug_05 sites increased linearly and reached the pre-fire level only in the third post-fire year (**Figure 1.18**). Change in NDVI analysis for WC_08 and LT_08 are not yet complete.

1.4 Discussion

Seasonally-flooded wetlands in South Florida are prone to frequent fire and flooding. This study indicated that the hydrologic conditions at the time of fire, the interval between fire and post-fire flooding, and the relative strength of the flooding event were all important in shaping the response of vegetation to the synergetic effects of the two disturbances. This study illustrated the sensitivity of prairie vegetation recovery to temporal variation in hydrologic regime. Differences of only a few cm in mean annual water depth could offset the recovery trajectories of vegetation that has not reached a stable state.

The observed pattern of post-fire vegetation dynamics in marl prairies burned in 2005 and in some fires of 2008 differs from results reported for other fires. Several authors reported that vegetation returns to pre-burn conditions within 3-5 years of fire after a single burn in seasonally-flooded wetlands (Werner 1975; Pahl et al. 2003; La Puma et al. 2007). A similar pattern of vegetation recovery was also reported at two wet prairie sites burned in spring 2003, and sampled annually for four years thereafter (Sah et al. 2008, 2009). The discrepancy between the results of the present study and earlier research is probably due to differences in post-fire hydrologic conditions, as the majority of sites burned in 2005, particularly Aug_05 sites, were flooded after fire. The high water conditions 1-2 months after fire in Population D (May_05) also had a significant impact on vegetation composition. In both 2005 fire groups and MC_08, not only did vegetation composition differ between pre-burn and 5-6 years post-burn, but cover and biomass also did not return to pre-burn levels within 6 years.

In two fires, west of Shark River Slough, there was not significant change in vegetation composition within observed period (Table 1.3). After both fires, vegetation cover and biomass also returned to the pre-fire level in 2-6 years. Even though the pre-fire site conditions differed between these two groups, vegetation recovery was at normal trajectory. However, recovery was faster after West Camp fire – in 2-3 years cover and biomass returned to pre-burn level, whereas

at the marsh sites (LM_08) it took longer. Since, sampling was not done annually, the time for recovery was somewhere between 2 and 6 years.

It was surprising to us that vegetation composition at the sites in MC_08 groups remained different from pre-burn even six years after the Mustang Corner fire even 6 years after the event. These sites were not flooded immediately after fire, but both burned and unburned sites in this area were under water in early spring of 2009, 9 months after fire, primarily due to seepage from the adjacent retention ponds. In contrast to the burned sites in MC_08 group, the unburned sites in the same area showed no change in composition over the same sampling period (**Figure 11a**). Moreover, when the MC_08 sites were analyzed by dividing them into subgroups based on their positions in relation to retention ponds, two distinct patterns were revealed. At sites located to the north, outside the range of influence of the retention ponds, vegetation recovery was faster than at southern sites impacted by the retention ponds. This dissimilarity in vegetation response pattern at burned sites with different post-fire hydrologic conditions, as well as differences in vegetation response to dry season high water conditions at burned and unburned sites, suggests that flooding in the dry season even one year after fire can severely affect marl prairie vegetation composition and impede its recovery.

In South Florida, where the likelihood of wildfire from lightning is much higher at the onset of the rainy season, flooding within 1-3 months of fire is common. However, what is important here is the rapidity and extent to which water rises after fire. The impact of post fire flooding on vegetation recovery becomes severe when there is a rapid rise in water level, and when water depth rises to elevations more than about 20-30 cm above the surface in the following three months after fire. In this study, seasonally-flooded wetlands were primarily graminoid-dominated. Most graminoids normally resprout and grow rapidly within a few weeks of fire, but when their aerial shoots are consumed and subsequently submerged by post-fire flooding, they may succumb to flooding-induced oxygen deficiency in their surviving belowground parts (Ball 1990; Kirkman and Sharitz 1994; Ponzio et al. 2004). Other Everglades studies have also reported that the synergistic effects of fire and flooding that submerge the remnant culms of plants can be locally detrimental to species such as sawgrass (*C. jamaicense*) and muhly (*M. capillaris* var. *filipes*) (Herndon et al. 1991; Snyder and Schaffer 2004). A steep decrease in the cover of dominant species usually provides conditions suitable for the growth of opportunistic species (Zedler and Krecher 2004). The relatively low cover of dominant species in post-fire years at the Aug_05 sites also seemed to facilitate the growth of other species. At May_05 sites, where water level increased gradually, providing ample opportunity for the re-growth of plants after fire, a large decrease in the relative cover of sawgrass was a surprise to us. When leaf meristems of sawgrass are not damaged by fire, the plants are known to grow rapidly, up to 20 to 40 cm in two weeks (Forthman 1973), which helps them to cope with the rising water level. In our study, lower relative cover of sawgrass in post-fire years than in pre-burn samples may also be a function of differences in dead material. Prior to the 2005 fires, the sites had not burned for 14 years, and retained a large component of dead sawgrass.

Both trajectory and NDVI change analyses contributed to our assessment of vegetation recovery dynamics in post-fire years. In the trajectory analysis, while many of 2008-burned sites demonstrated a significant shift in species composition towards pre-burn conditions, none of the May_05 and Aug_05 sites showed such a significant shift. Our expectation was that vegetation

recovery, i.e., the mean rate of change, would be faster in May_05 than in the Aug_05 group. Visual analysis of trajectories revealed that the trajectory of several May_05 sites had shifted roughly in the opposite direction of the TSLF vector during the 4th year after fire, suggesting that the vegetation recovery process at several May_05 sites changed in direction between Years 3 and 4. Within the group, a visible shift of sites in ordination space toward increasing hydroperiod suggested that vegetation in post-fire Year 4 was indicative of wetter conditions than in previous years. At the Aug_05 sites, vegetation recovery towards pre-burn composition was slow, and vegetation composition even five years after fire differed significantly from pre-burn surveys.

The change in NDVI following fire represented vegetation recovery trajectories that varied among the four burned areas, suggesting differences in the inherent resilience of pre-fire vegetation types, and in the post-fire environmental conditions, particularly hydrology. Pre-fire vegetation in the May_05 sites was dominated by sawgrass, which was indicative of relatively long hydroperiod, whereas vegetation in three groups Aug_05, MC_08 and WC_08, sites were typical of shorter hydroperiod wet prairies. In general, fire impacts on vegetation tend to be less severe at marsh sites than prairie sites, due to high plant and soil moisture content. Post-fire flooding killed most of the vegetation at the Aug_05 sites, while the vegetation at the May_05 sites had an opportunity to grow before the gradual onset of flooding two month after fire. The recovery of NDVI to the pre-fire level in just two and four years in May_05 and Aug_05 contrasted somewhat with field based estimates of plant cover, which took longer to return to pre-fire levels. NDVI is affected by several factors, including total plant cover, biomass, plant and soil moisture, and leaf area index. Since NDVI is sensitive to chlorophyll content, varying amounts of dead plant biomass in the ground cover may have effects on NDVI (van Leeuwen and Huete 1996). In general, dead or dry plant material produces spectral reflectance pattern similar to soil. In this study, total cover in the pre-fire year, was high, however live materials constituted only 24% and 37% at May_05 and Aug_05 sites, respectively. In contrast, the fraction of live biomass was >50% in post fire years.

In summary, fire, an integral part of marl prairie ecosystem, is likely to create a vegetation mosaic within the landscape, particularly when its effects on vegetation structure and composition are mediated through other disturbances, such as changing water levels. The interval between fire and post-fire hydrologic events is important in shaping the response of vegetation to the synergistic effects of these two disturbances. However, it is the relative strength and duration of secondary disturbance that determines the course of post-fire vegetation recovery trajectories, which in turn shapes the vegetation mosaic pattern. Our study of vegetation response to fire and hydrology also reveals that prairie vegetation recovering from a single fire is especially sensitive to annual variation in hydrologic regime, and especially dry season water level. Differences of only a few cm in mean annual water depth could offset the recovery trajectories of vegetation that has not reached a stable state. Finally, it is recommended that the use of fire as a management tool for restoration of marl prairie habitat take into account likely post-burn hydrologic conditions, and when necessary be coupled with management of the post-fire hydrologic conditions, in order to produce the desired results.

2. Cape Sable coastal prairies: a decadal change in vegetation structure and composition

2.1 Introduction

South Florida coastal ecosystems are vulnerable to natural disturbances, such as hurricanes and fires, as well as hydrologic changes caused by both human-induced water flow alterations and climatic variations. Both natural and anthropogenic stressors have greatly altered South Florida ecosystems. Located at the southwestern tip of South Florida, the Cape Sable region also witnessed the wave of ecosystem degradation that occurred throughout the Everglades during the 20th century. In that region, the coastal prairies, where Arthur H. Howell first discovered Cape Sable seaside sparrow (CSSS) on February 18, 1918 (Howell, 1919), were primarily dominated by sand cordgrass (*Spartina bakeri* Merr.). In most of these prairies, where the only breeding population of CSSS was known in the 1930s (Nicholson 1928), the vegetation has shifted to mangroves, mud flats, and salt-tolerant forbs such as shoreline seapurslane (*Sesuvium portulacastrum* (L.) L.), saltwart (*Batis maritima* L.) and rushy seaside oxeye (*Borrichia frutescens* (L.) DC.) (Kushlan and Bass 1983). The remaining coastal prairies in Cape Sable are also in a continuous state of impact from sea level rise, which exerts a slow, continuous pressure, and causes lasting change. Other disturbances, such as fire and hurricanes, occur abruptly over short period of time, and can cause almost immediate changes in community structure and composition. In general, the community usually recovers in a relatively short period of time after these periodic pulse disturbances. However, two major hurricanes, Labor Day in 1935 and Donna in 1960, had long-lasting effects on Cape Sable landscapes (Wanless and Vlaswinkel, 2005). Altogether, these disturbances have caused substantial modifications of both historical and recent range of sparrow habitat in southern Everglades (Post and Greenlaw 2000).

In the past century, Cape Sable prairies have also been impacted by canals, including East Cape Extension and Homestead Canals, constructed during early 1900s to drain the cape's interior and facilitate agriculture and development (NPS/ENP 2009). These canals were dredged through the Cape Sable marl ridge that has historically served to retain freshwater in the interior wetlands. The canals caused the freshwater to drain rapidly from the interior marshes, and salt water to intrude further inland, resulting in degradation of freshwater and brackish marshes. Higher salinity in the interior marshes resulted in altered vegetation composition and reduced the quality of wildlife habitat. Management efforts to mitigate this damage to natural ecosystems included construction of earthen dams in the canals in the late 1950s and early 1960s, and sheet-piling dams in 1997 (NPS/ENP 2009). However, the benefits of such efforts were not fully realized, mainly due to repeated failure of dams leaving the interior wetlands and lakes vulnerable. In 2009, Everglades National Park (ENP) initiated a project under the guidelines of American Reinvestment and Recovery Act (ARRA) to restore the dams and plugs on the East Cape Extension and Homestead Canals (NPS/ENP 2009). The expectation was that the fully functional dams would improve wetland conditions and vegetation pattern north of the Cape Sable marl ridge through a reduction in saltwater intrusion and an increased capacity for freshwater retention.

With a broad goal of assessing the vegetation shift in the historical range of CSSS habitat, the remaining part of Cape Sable coastal prairies were also included as a part of vegetation monitoring sites established throughout the CSSS habitat (Ross et al. 2006). The specific objectives were to characterize recent vegetation structure and composition within the coastal prairies, and to assess the response of the coastal prairie vegetation to both natural and anthropogenic stressors.

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2.2 Methods

2.2.1   Study Area

In the Cape Sable costal prairies, six sites were established in 2004 as a part of the network of CSSS habitat vegetation monitoring sites (Ross et al. 2006) (**Figure 2.1: Table 2.1**). Those six sites nearly overlapped sites that were surveyed for CSSS population in 1981 within the same area (Kushlan and Bass 1983). At those sites, vegetation survey was first conducted in 2004, and then in 2008 and 2014. Between successive surveys, some of the sites were burned in prescribed fires in 2008 and 2011 (**Table 2.1**), and part of the region was probably inundated by storm surge caused by hurricanes Katrina and Wilma in 2005, one year after the first sampling.

Table 2.1: Vegetation monitoring sites in Cape Sable coastal prairies

Year of establishment	SiteID	Easting (NAD83)	Northing (NAD83)	ENP_LOCID	Elevation (cm)	Burned year(s)
2004	G-01-01	497375	2786013	lake-006	-28.8	2011
2004	G-01-02	496702	2786098	lake-005	-37.2	2011
2004	G-01-03	496593	2786975	lake-001	-12.5	2001, 2011
2004	G-01-04	497506	2787635	lake-002	-19.4	2001, 2008, 2011
2004	G-01-05	498658	2787031	lake-003	-17.6	2002, 2008, 2011
2004	G-01-06	499125	2787145	lake-004	-14.4	2002, 2011

2.2.2   Vegetation Sampling

At each site, the vegetation sampling followed the same protocol described in Section 1. In brief, at each sampling site, vegetation was sampled in a N-S oriented, 1 x 60 m rectangular plot. Nested within each plot were the 30 0.25 m^2 (0.5 x 0.5 m) structural plots, of which ten were also the compositional subplots. We estimated ocular cover of each species present in the compositional plot. Any additional species not present in these sub-plots, but in the 1 x 60 m plot were assigned a mean cover of 0.01% for the plot as a whole. A suite of structural parameters (canopy height, total percent vegetation cover, and live vegetation expressed as a percent of total cover) were recorded in structural plots.

2.2.3   Data Analysis

The sites were classified using a hierarchical agglomerative cluster analysis. We used Bray-Curtis dissimilarity as our distance measure, and the flexible beta method to calculate relatedness among groups and/or individual sites. Vegetation pattern was then illustrated using a non-metric multidimensional scaling (NMS) ordination, and differences in vegetation composition among years were examined using analysis of similarity (ANOSIM). For NMS ordination, cover data were relativized by species maxima.

The vegetation composition in the coastal prairies is also a manifestation of ground water salinity in the area. However, we did not have direct measurement of salinity at the vegetation survey sites. Thus, we indirectly estimated site salinity index based on qualitative information on salt tolerance of species occurring at each site. The qualitative information on species' salt tolerance was gathered from the literature and web-based databases (**Appendix 2.1**). Available information was very diverse, and the number of categories to express species' salt tolerance varied from 3 to 6. We settled on three categories: low, moderate and high tolerances that were used in most literature. We then assigned an incremental value of 1, 5 and 10 in concurrence with increasing tolerance, and finally we calculated a weighted salinity index by multiplying the importance value (IVI) of each species and its tolerance value. A change in vegetation-based site salinity index between successive samplings reflects the amount and direction of change in vegetation, such as toward freshwater or brackish vegetation, at the particular site.

Vegetation structural measurements were summarized for each plot. Mean canopy height and total vegetative cover were used to estimate above ground plant biomass, using the allometric equation developed by Sah et al. (2007) for the vegetation within CSSS habitat. Finally, pair-wise t-tests were used to test for differences in vegetation cover, biomass and species richness among sampling years.

2.3 Results and Discussion

Vegetation within the Cape Sable coastal prairies were broadly of two types: 1) *Spartina* prairie, and 2) *Cladium* marsh (**Figure 2.2**). An increasing abundance of *Spartina* along the coastal gradient, possibly with increasing salinity, was evident. The northwestern prairies were dominated or co-dominated by thick sawgrass, whereas those in eastern and southern prairies were dominated by *Spartina bakeri* (**Figure 2.3**).

The difference in vegetation composition between years was not statistically significant (ANOSIM: p-value > 0.05). However, it could be due to low power because of small sample size (n = 6), as some sites revealed a dramatic change in cover value of characteristic species (**Figure 2.3**). For instance, at one site, close to Middle Fox Lake, *Spartina bakeri* cover in 2014 was only one third of what it was in 2004. At the same site, the reduction in *S. bakeri* was accompanied by an increase in the cover of *Sesuvium portulacastrum,* a species with much higher salt tolerance, suggesting that the salinity in some parts of Cape Sable increased over time. This was also confirmed by a noticeable reduction in the cover of a freshwater species, spikerush (*Eleocharis cellulosa*) at the adjacent site. Both sites had relatively low ground elevation (**Table 2.2**), and they were also relatively close to the Homestead Canal that is known to have increased the saltwater intrusion in the interior Cape Sable (Wanless and Vlaswinkel 2005). At the other four sites, located in the north, there was mostly an increase in the relative cover of freshwater species, such as sawgrass, spikerush and swamp fern (*Blechnum serrulatum* Rich.) (**Figure 2.3**).

Changes in species composition are also reflected in sum of species abundance-weighted salt tolerance values for the sites (**Table 2.2**). Our expectation was that the abundance-weighted salt tolerance index during the 2nd sampling would be higher than before, mainly due to storm surge, and then over time it would decrease. However, an initial increase was only at site G-01-

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02. At this site, the community salinity index increased over 4 years (2004-2008) and then continued to increase. The value was the highest during 2014 sampling. Since this site has the lowest elevation (-37.2 cm,), 9.2 cm lower than 2nd nearest site, this is the one site most likely to be impacted. At the first site the values are upset by the reduction in total cover of moderately tolerant species, even though there was an increase in the cover of highly tolerant species. When the restoration project encompassing the filling up the canal is underway, it will be interesting to follow these two sites, which can serve as indicators of change in salinity in the area.

Table 2.2: Sum of the species' importance value (IVI) weighted salt tolerance index for CSSS vegetation monitoring sites.

Field ID	Sampling year		
	2004	2008	2014
G-01-01	999	803	1000
G-01-02	764	920	1000
G-01-03	735	636	701
G-01-04	466	433	432
G-01-05	781	763	
G-01-06	957	786	833

The vegetation shift in Cape Sable prairie over one decade was probably influenced by both fire and hurricane. At the time of the sampling in 2004, vegetation at five of six sites was recovering from the fire that had occurred at those sites 2-3 years prior to the sampling. However, the recovery process might have been affected by hurricanes in 2005, when the Cape Sable region received storm surge from hurricanes Katrina and Wilma. The storm surge from Katrina in that region was estimated as 19.87 inches (NPS/ENP 2009). The decline in the cover of cordgrass or spikerush, accompanied by an increase in salt tolerant species at the southernmost sites, relatively close to the canals, suggests an increase in salinity in the area. At those sites, the total vegetation cover also decreased by one third (**Figure 2.3**). In the 20th century also, at least two hurricanes, Labor Day (1935) and Donna (1960) brought extreme storm surge flooding that caused a significant shift in sedimentation and vegetation composition in the Cape Sable region (Wanless and Vlaswinkel, 2005). In contrast to these locations, vegetation recovery at the northern sites continued for the next few years, resulting in a significant increase (> 40%) in cover of major species between 2004 and 2008. Thereafter, the plant communities were probably affected by 2008 and 2011 prescribed burns that occurred at two and five sites, respectively.

In the Cape Sable region, prescribed fires have been used for fuel reduction and control of woody plant expansion. Our results revealed that mean aboveground biomass was the lowest (909 g m^{-2}) in 2014. Almost all sites have lower biomass in 2014 than in 2008 (**Table 2.3**). It is likely that repeated burnings between 2008 and 2014 at some of our study sites have caused a reduction in mean above ground biomass in six years. Moreover, one of the objectives of the prescribed fire program was to control the exotic species, including *Lygodium microphyllum* (Cav.) R.Br. In our study, we found *L. microphyllum* at one site in 2014, less than what were in 2008, when it was present at two sites. Since, mean cover of *L. microphyllum* was very low (<0.1%) in both years, we were unable to ascertain any causal relation relationship between *L. microphyllum* cover and

prescribed burning, especially when an integrated approach for treating exotic plants, including herbicide treatment, was practiced.

Table 2.3: Vegetation monitoring sites in Cape Sable coastal prairies (historical range of Cape Sable seaside sparrow)

Year of establishment	FIU_ID	*Biomass g m^{-2}		
		2004	2008	2014
2004	G-01-01	1160	784	737
2004	G-01-02	1338	824	848
2004	G-01-03	783	1070	838
2004	G-01-04	1240	1827	1291
2004	G-01-05	867	795	
2004	G-01-06	778	965	833
Mean (±SD) Biomass		**1,028[a] (±248)**	**1,044[a] (±400)**	**909[b] (±218)**

Using biomass allometric equation from Sah et al. (2007)

Fires
2001 (Cape Sable # 01054); 2002 (Cape Sable # 02055);
2008 (RX-58 # 8093) and 2011 (RX-48 # 1138)

Woody encroachment is a chronic environmental problem in coastal areas (Ross et al. 2000; Lunt et al. 2010; Smith et al. 2013). However, we did not find any significant increase in woody plants at our survey sites in ten years. The reason could be the repeated fires at most of the sites. Fire has been considered an effective tool to control the woody encroachment in coastal prairies (Grace et al. 2005; Duever and Roberts 2013), though a recent study has shown contradicting results, as the researchers found a burned area with no mangrove encroachment, while another frequently burned area experienced an increase in mangrove cover and a decrease in marsh (Smith et al. 2013). Their study was limited to two sites in southwestern coast of Florida. A detailed analysis of the woody plant data in all the prescribed fires in the region is in progress (*Meghan Gonzalez* – Grad Student; personal communication), and it is expected that the study would help to assess the effectiveness of prescribed burning in controlling woody encroachment in coastal prairies.

In summary, the coastal plant community in the Cape Sable area is dynamic, and the species composition there is a manifestation of several interacting forces, natural and anthropogenic, acting on varying spatial and temporal scale. While disturbance pulses like natural fire and hurricanes have important roles in shaping the coastal communities, sea live rise is acting as a press disturbance. Overlaid on these forces are the fire management and the canal modification activities. All these forces will continue to impact the Cape Sable vegetation.

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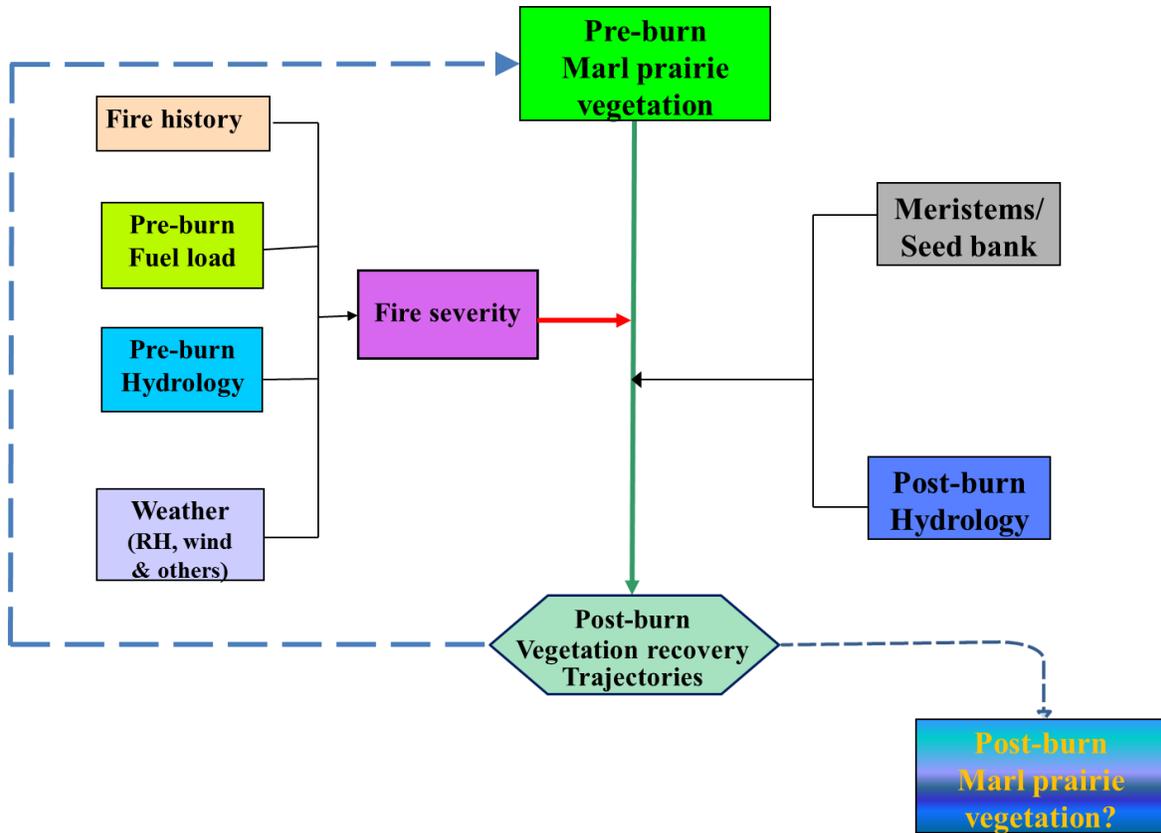
Figure 1.1: A conceptual model showing the factors, including post-fire hydrologic conditions affect post-burn vegetation recovery.
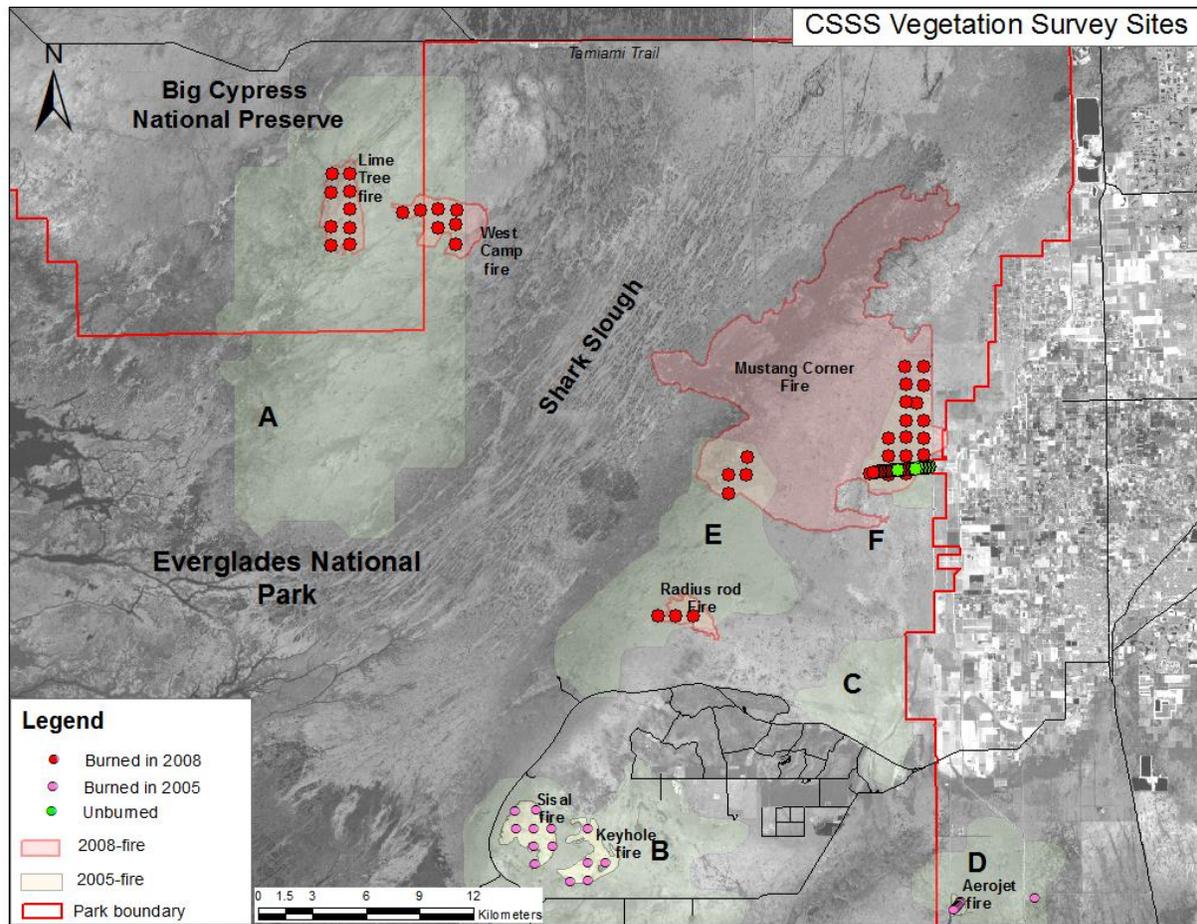
Figure 1.2: Study area map showing the Cape Sable seaside sparrow habitat vegetation study sites burned in 2005 and 2008. The sites were sampled before fire and several times after fire. A number of unburned sites on Transect F were also sampled several times.
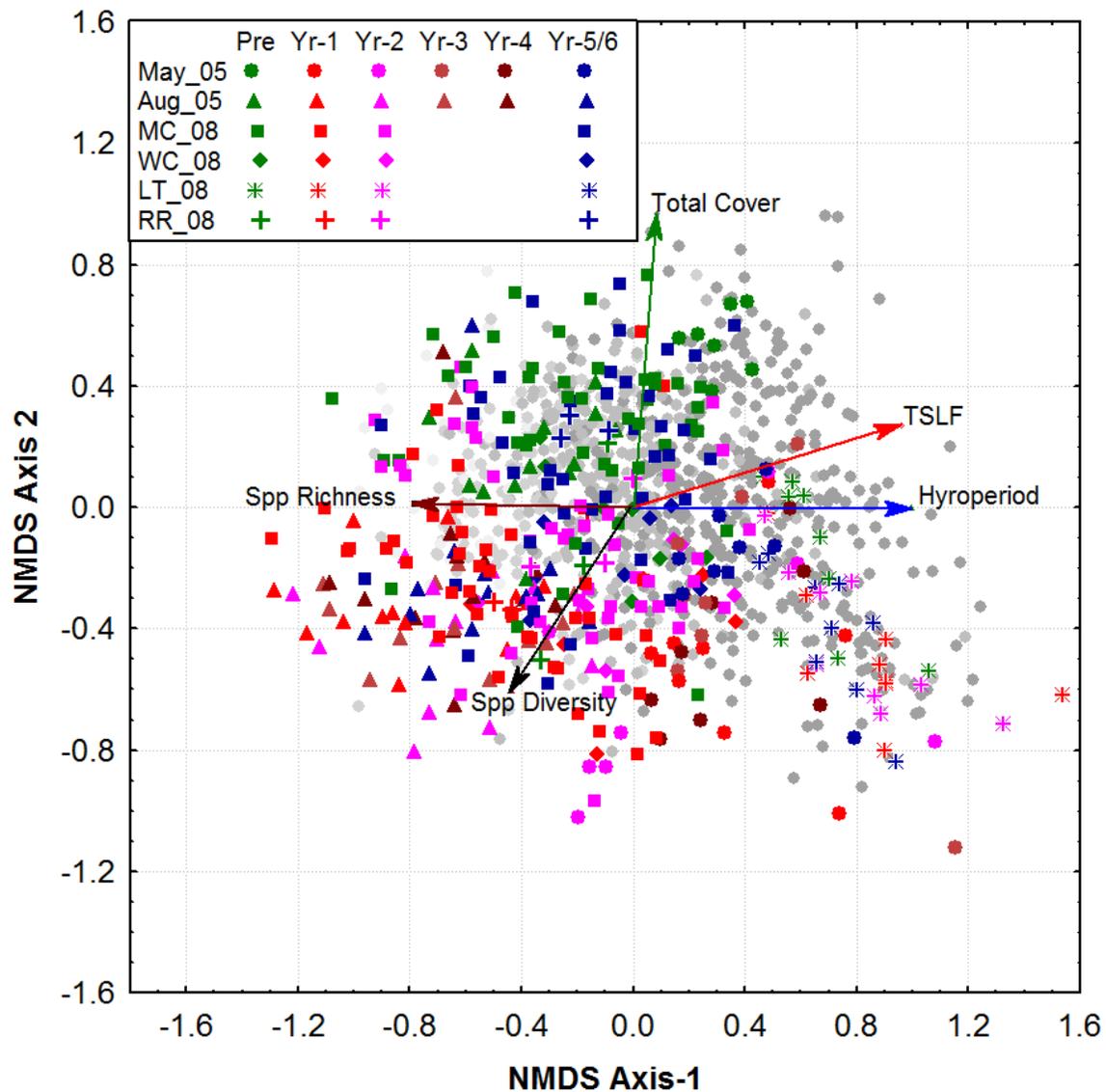
Figure 1.3: Non-metric multidimensional scaling (NMDS) ordination based on total cover at the reference sites and the sites burned in 2005 & 2008. Reference sites (gray) are shaded with increasing hydroperiod (≤120, 121-180, 181-240 & > 240 days). Environmental and community characteristic vectors were fitted within ordination space. TSLF = Time since last fire. Sites are grouped by fire and sampling events (pre- & post-burn years). May_05 and Aug_08 are the sites burned in May and August 2005, respectively. MC_08, WC_08, LT_08 and RR_08 are the sites burned in 2008 fires (Mustang Corner, West Camp, Lime tree and Radius Rod, respectively). Yr-1, Yr-2… are the number of years after fire.
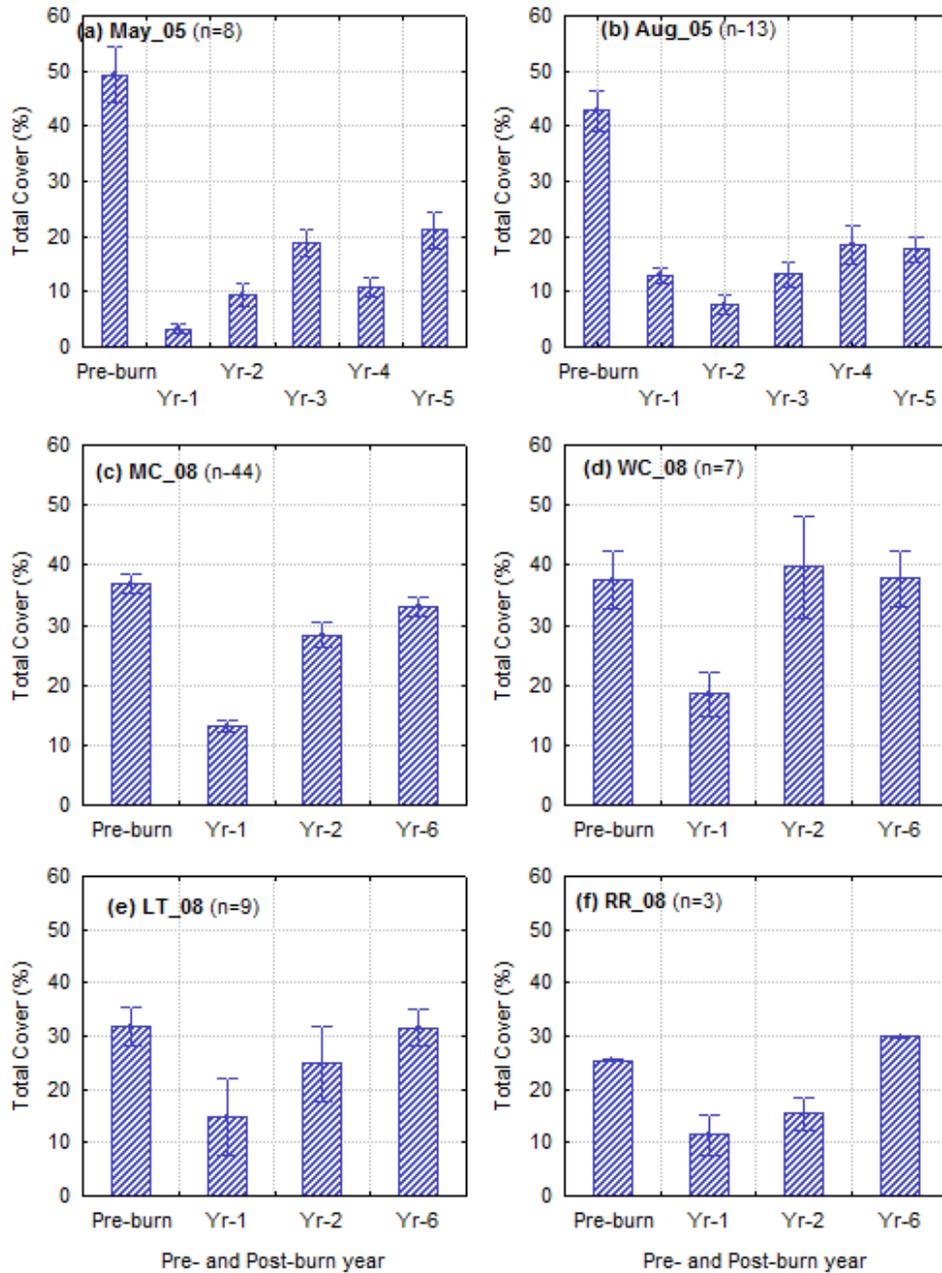
Figure 1.4: Mean (± SE) vegetation cover in pre- and post-burn years in 6 groups of sites burned in 2005 or 2008.
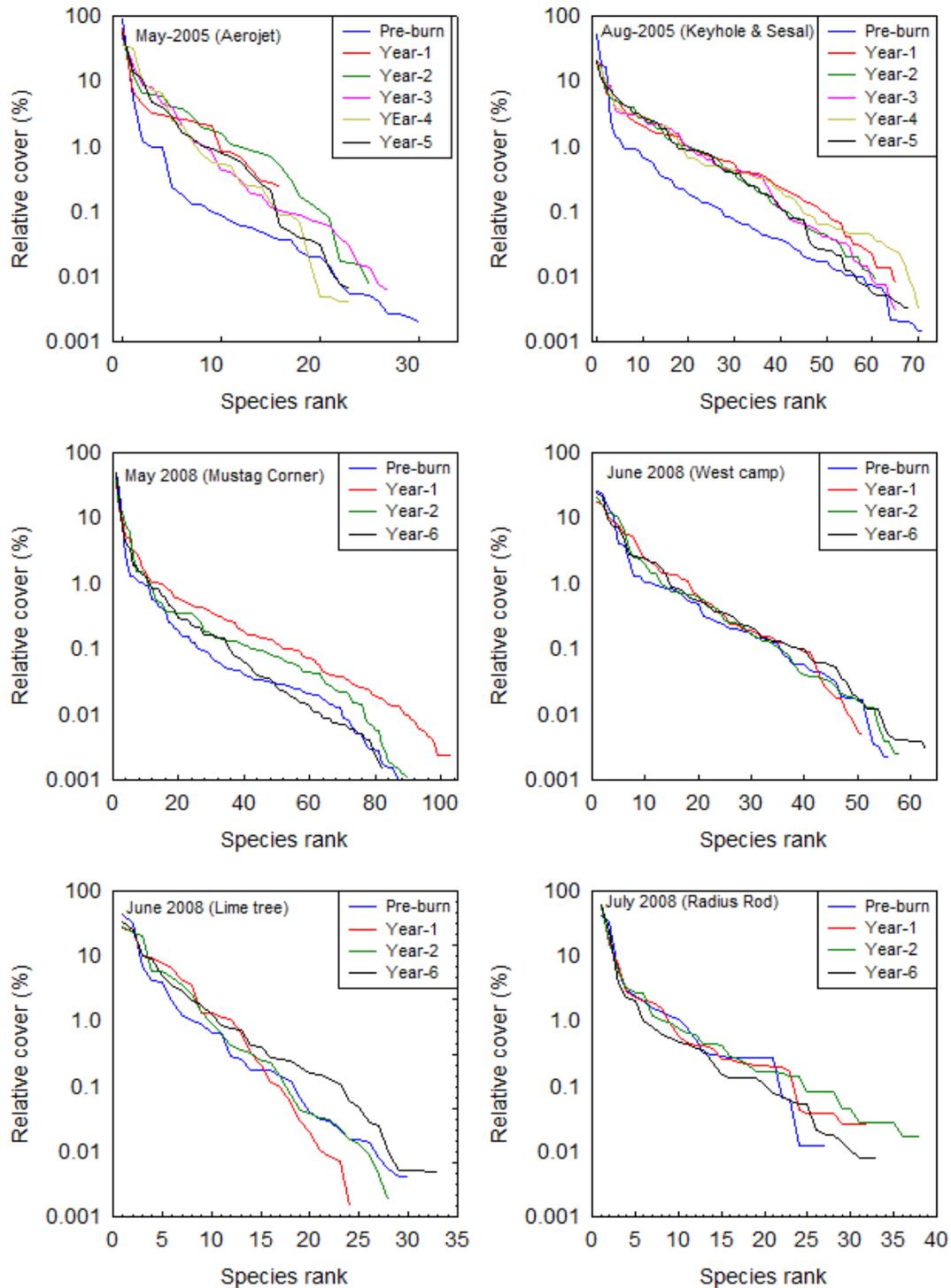
Figure 1.5: Species rank abundance at the sites burned in 2005 or 2008 fires. Sites from two fires, Keyhole and Sisal (2005) are lumped together, as sites after both fires were immediately flooded after fire. N = number of sites.
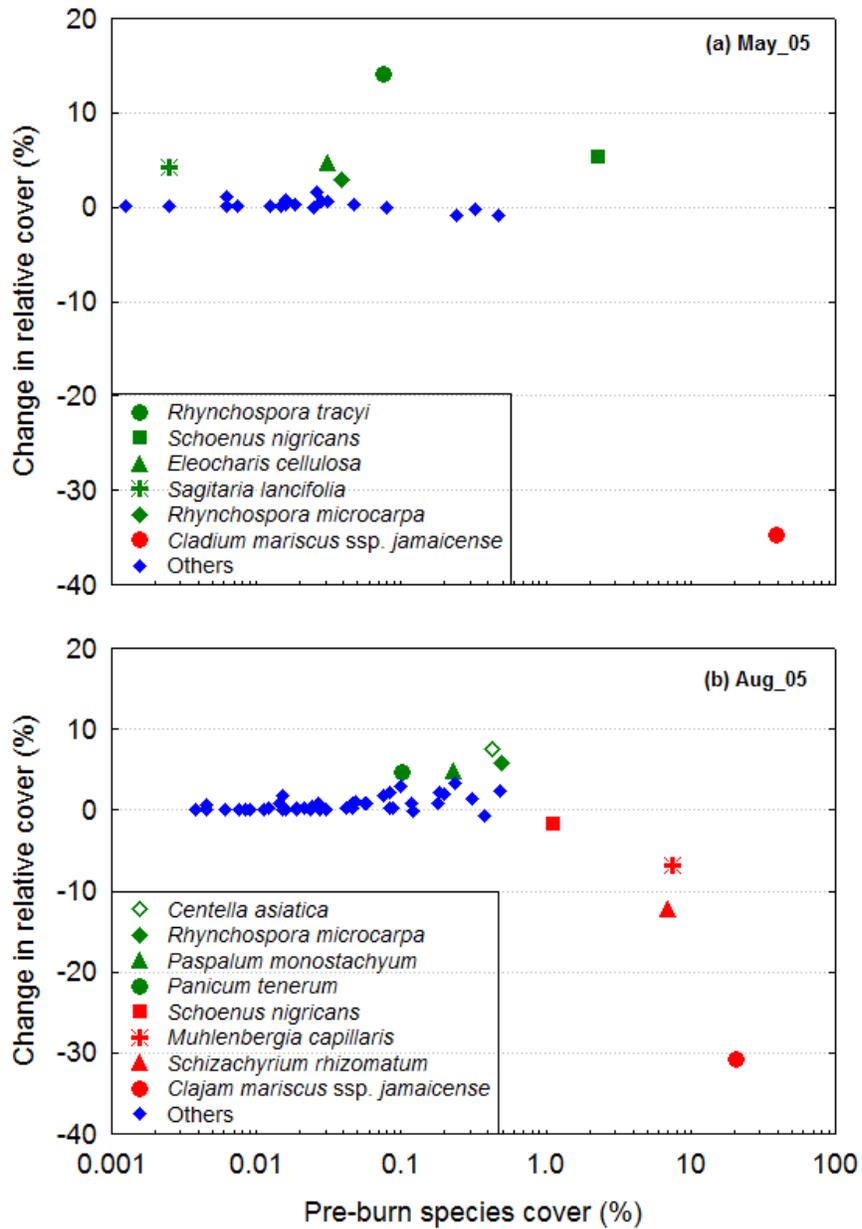
Figure 1.6: Change in the relative cover of species in five years after fire at the sites burned in (a) May 2005 (May_05) and (b) August 2005 (Aug_05) in CSSS sub-population D and B, respectively. The sites were sampled 0-3 years before fire, and re-sampled annually for five years after fire.
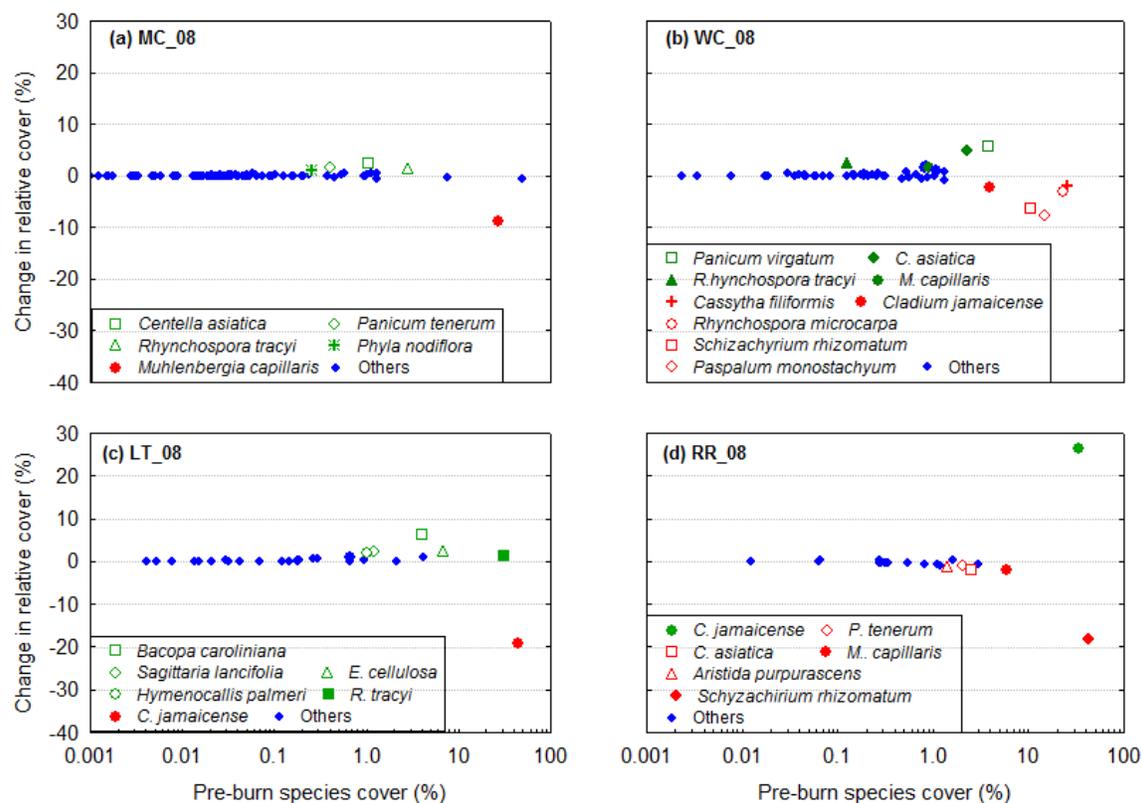
Figure 1.7: Change in the relative cover of species in six years after fire at the sites burned in four different fires (MC = Mustang Corner, WC = West Camp, LT = Lime Tree, RR = Radius Rod) in 2008 in CSSS sub-populations A, E and F. The sites were sampled 0-3 years before fire, and re-sampled in 1, 2 and 6 years after fire.
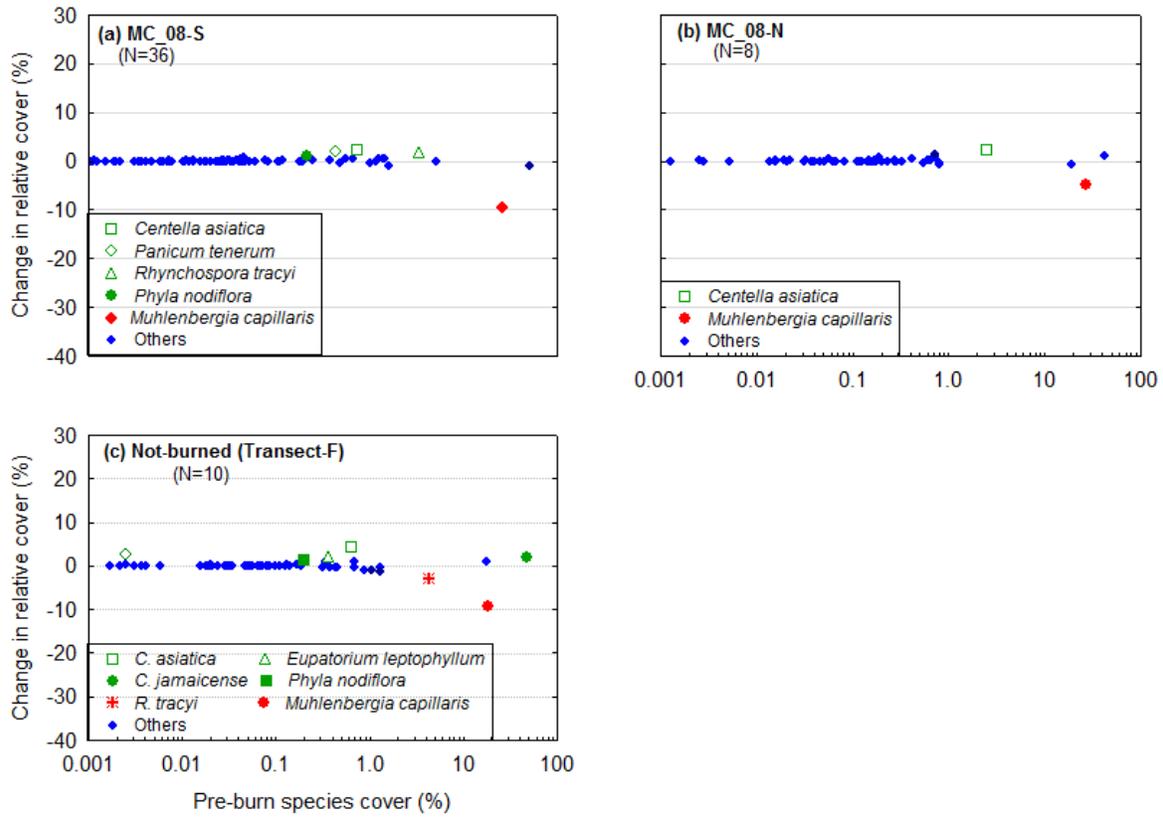
Figure 1.8: Change in the relative cover of species in six years after fire at the sub-population F vegetation sites burned in Mustang Corner fire in 2008. Sites are grouped in two groups: MC_08-S sites are west and south of the retention ponds, and MC_08-N sites are northwest of the retention ponds. The sites were sampled 0-3 years before fire, and re-sampled in 1, 2 and 6 years after fire. Ten sites on Transect F did not burn, but were also resampled simultaneously.
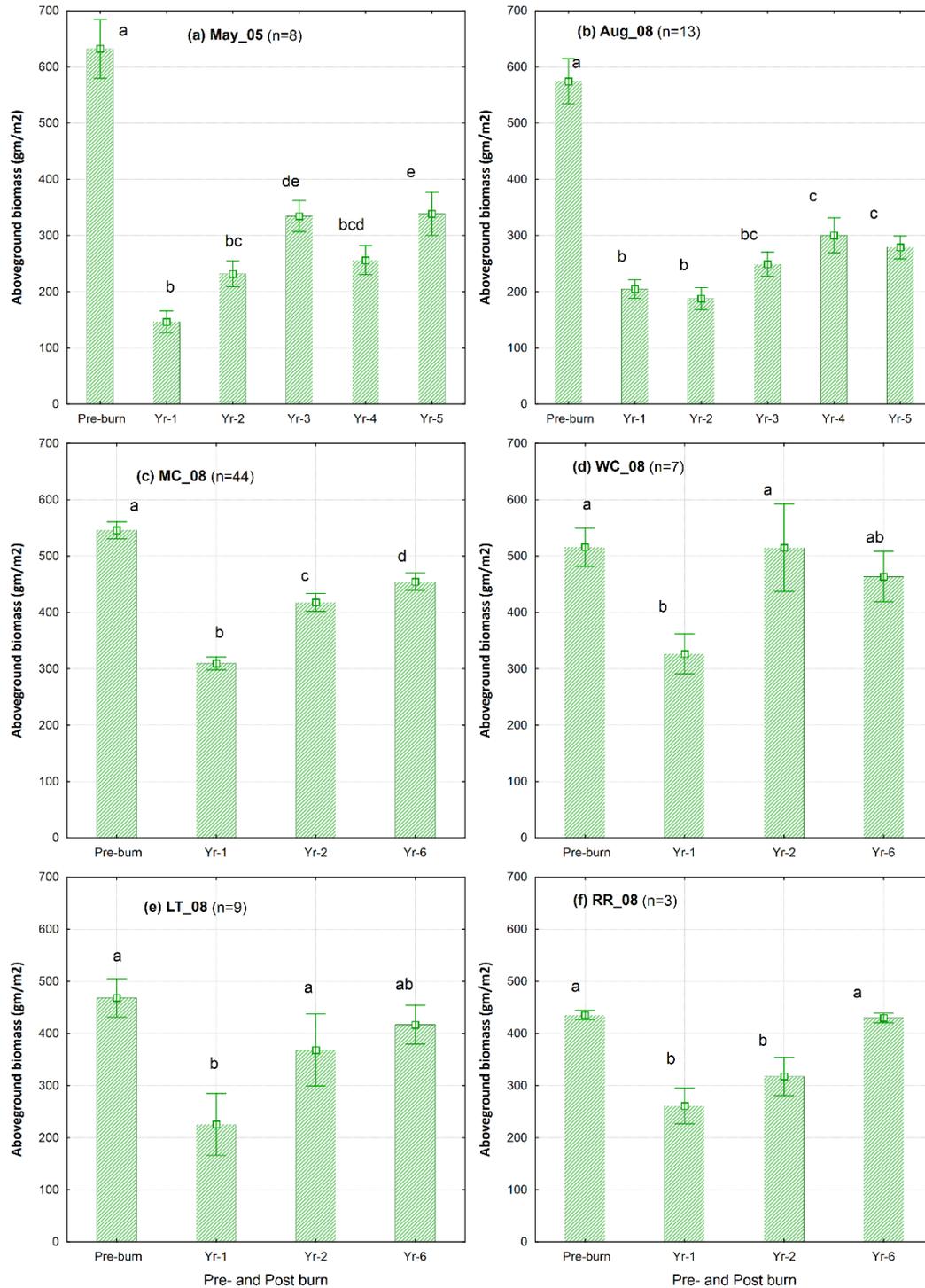
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Figure 1.9: Mean (± SE) above ground biomass in pre- and post-burn years in 6 groups of sites burned in 2005 or 2008.
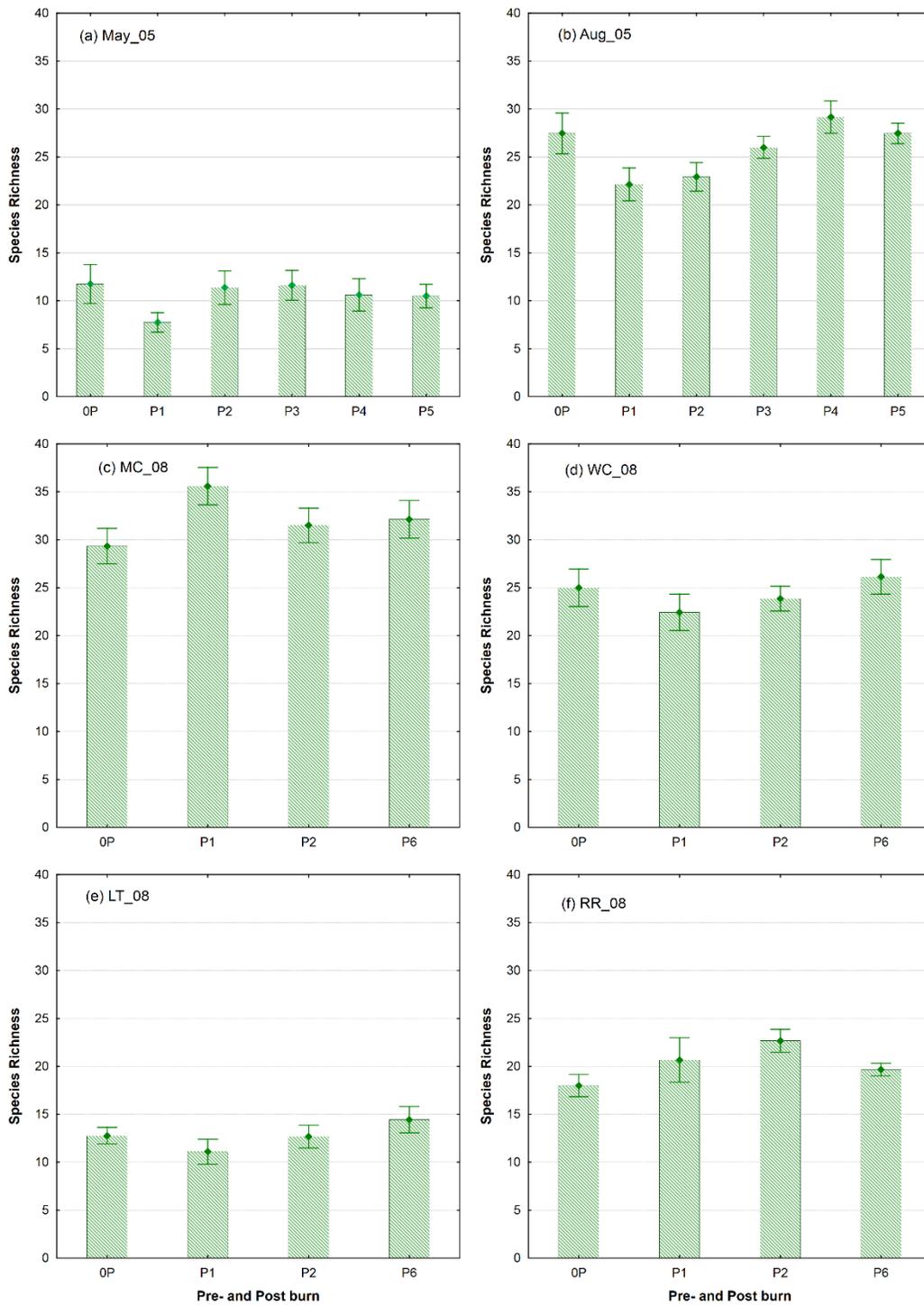
Figure 1.10: Mean (± SE) species richness in pre- and post-burn years in 6 groups of sites burned in 2005 or 2008.
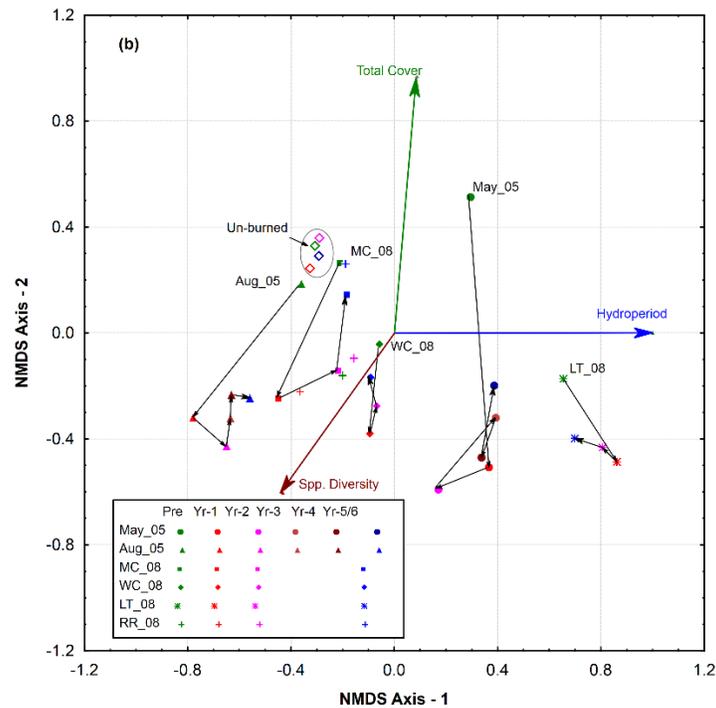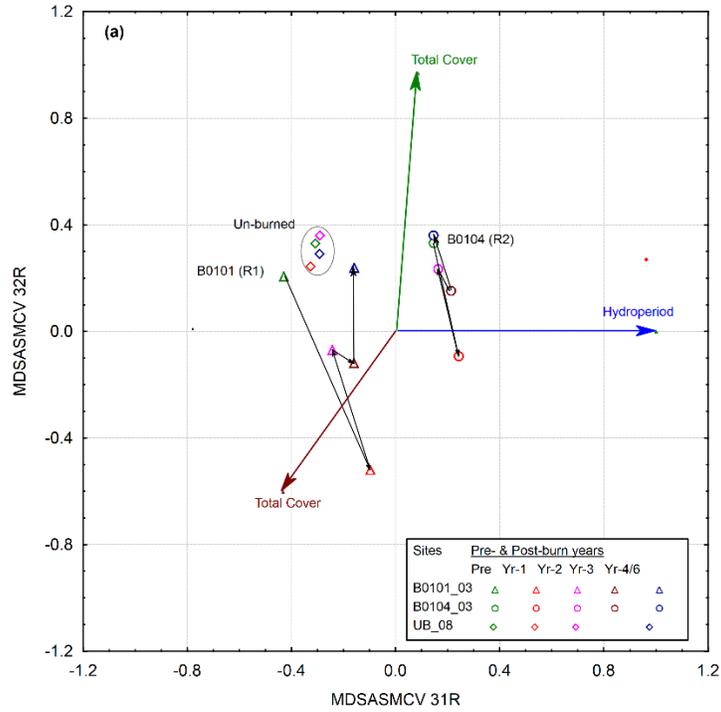
Figure 1.11: non-metric multidimensional scaling (NMS) ordination based on total cover at sites burned in 2003, 2005 or 2008 and sampled prior to burn and 1 to 5 or 6 years after fire. (A) Centroids of 2005 and 2008-burned sites grouped by burn year and fire. Sites burned in 2005 are sub-grouped in May-burned and Aug-burned sites. (B) Two sites burned in 2003 and sampled prior to burn and 1-4 years after fire. Unburned sites (UB_08) in both figures are the sties sampled at the same frequency as the sites burned in Mustang Corner (MC_08) fire were sampled.
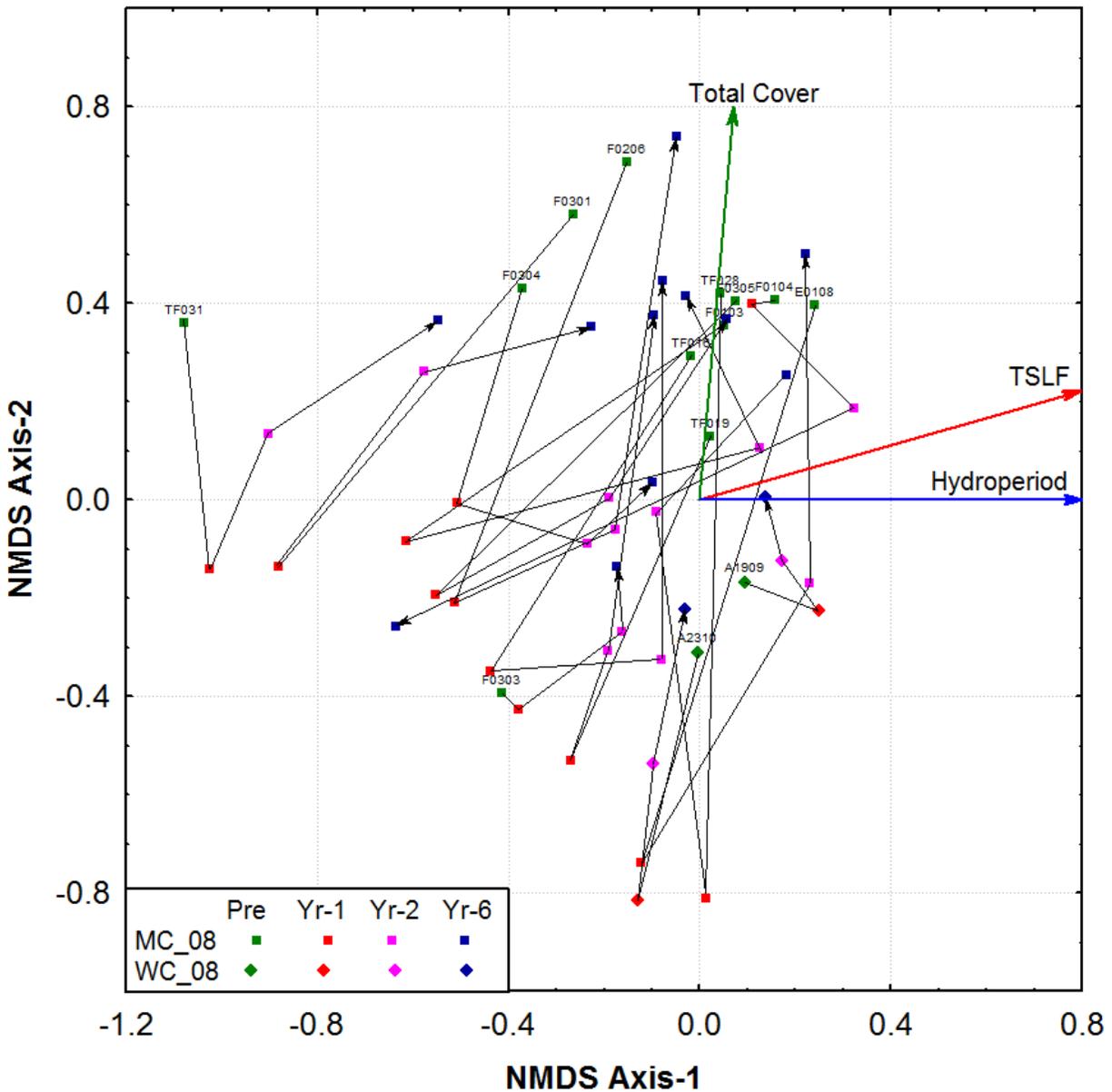
Figure 1.12: The NMDS ordination showing the trajectory of sites burned in 2005 or 2008. It includes only the sites that showed significant (p≤0.1) rate of change in species composition along the individual vectors between 1 year after fire and the initial (pre-burn) condition of the same site. Initial point and the end of arrows on each site trajectory represent the 2005 or 2008 and 5 (2010 sampling) or 6 (2014 sampling) years after fire, respectively.

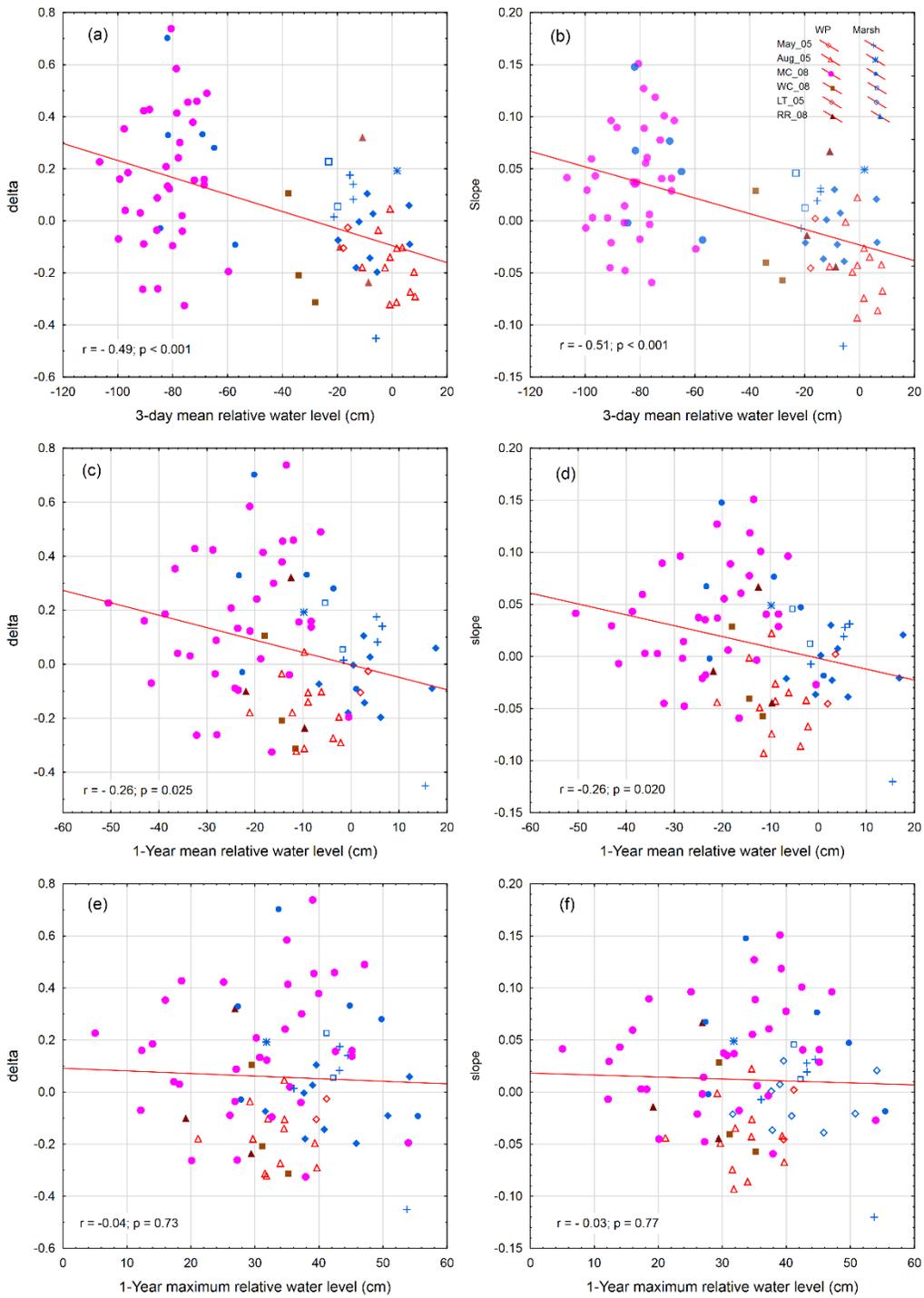**Figure 1.13:** Relationship between hydrologic conditions (water level) and a change in species composition at the sites, expressed as delta (Δ) and slope that quantify the degree and rate of change in vegetation composition along the reference vectors in the ordination space. The colored symbols represent different burn groups (May_05, Aug_05, MC_08, WC_08, LT_08 and RR_08) and two vegetation types (WP = Wet prairie, and M = Marsh).

Figure 1.14: a-i - Relationship between hydrologic conditions (mean & max relative water level, RWL) calculated for different periods (1, 3, 6, 12, and 24, months) and Slope (rate of vegetation change) calculated using trajectory analysis. The colored symbols represent different burn groups (May_05, Aug_05, MC_08, WC_08, LT_08 and RR_08) and two vegetation types (WP = Wet prairie, and M = Marsh). **j** – Change in coefficient of determination (R^2) between water conditions (mean and max RWL) and slope (rate of change in vegetation composition) over time.
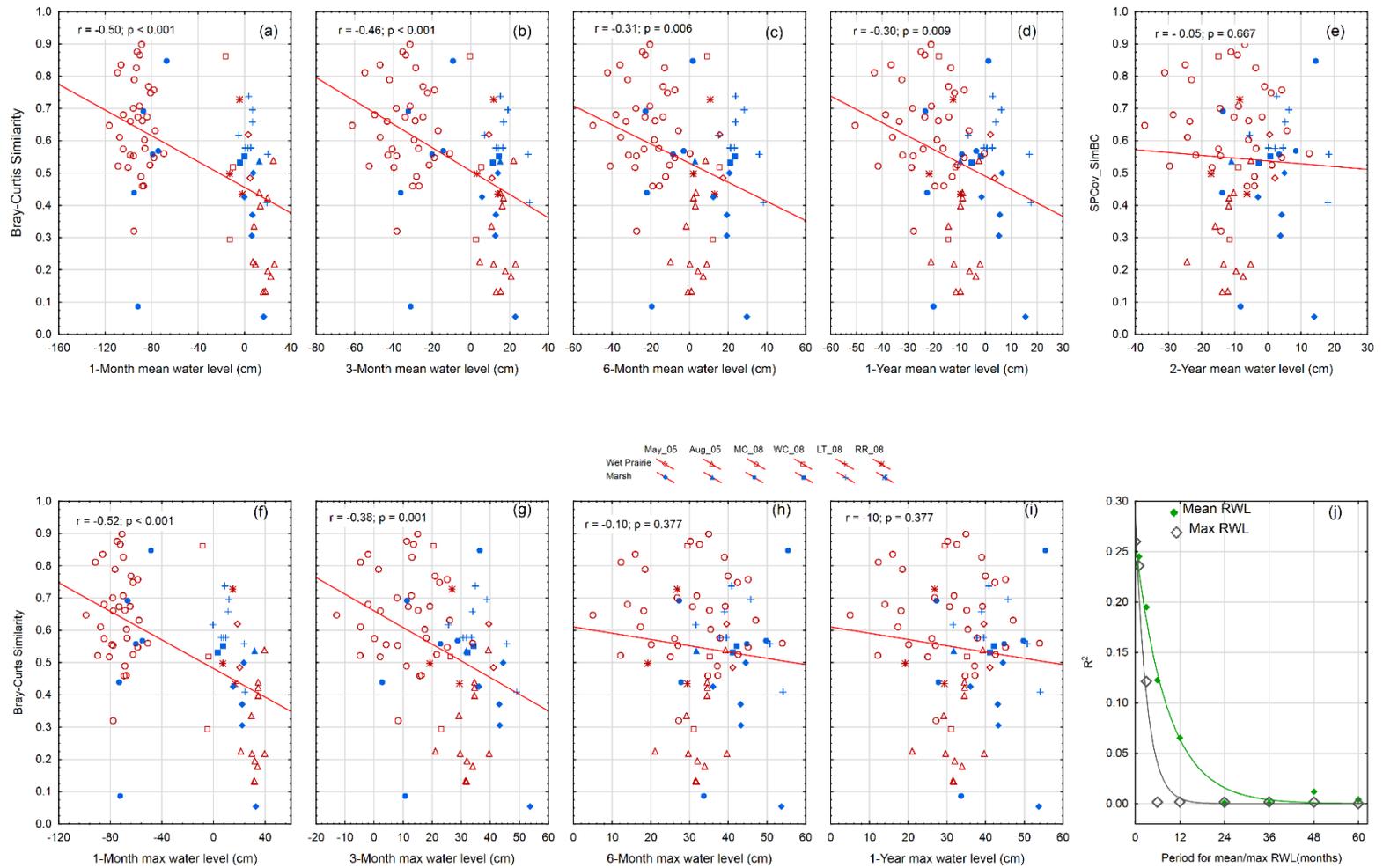
Figure 1.15: a-i - Relationship between hydrologic conditions (mean & max relative water level, RWL) calculated for different periods (1, 3, 6, 12 & 24 months) and Bray-Curtis similarity in species composition between pre-burn and at 5 to 6 years after burn. The colored symbols represent different burn groups (May_05, Aug_05, MC_08, WC_08, LT_08 and RR_08) and two vegetatio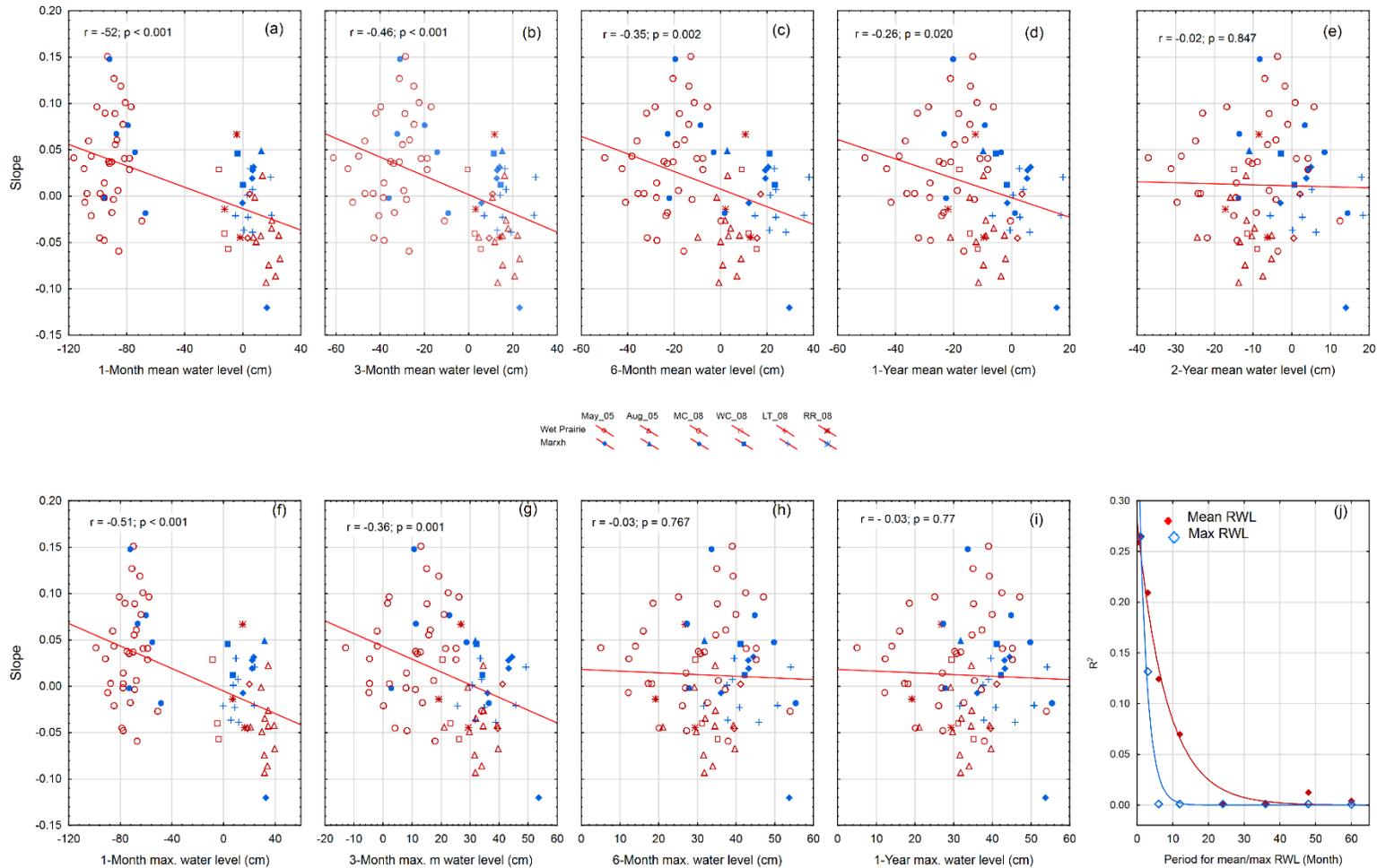n types (WP = Wet prairie, and M = Marsh). **j** – Change in coefficient of determination (R^2) between water conditions (mean and max relative water level, RWL) and Bray-Curtis similarity in species composition over time.
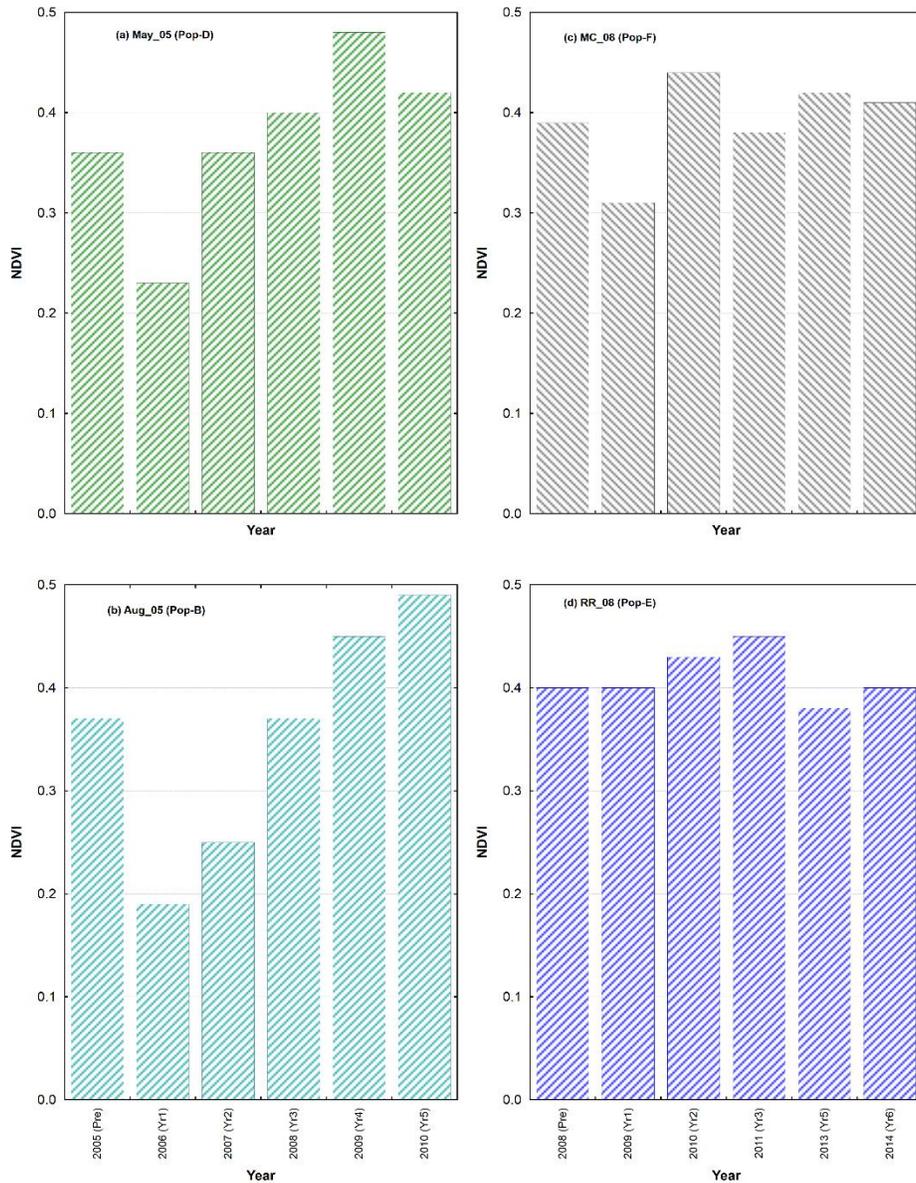
Figure 1.16: Mean Normalized Difference Vegetation Index (NDVI) in pre-burn and five to six years after fire. Sparrow habitat were burned by different fires in four different sub-populations (a) sub-population D burned in May 2005, (b) sub-population B burned in August 2005, 9c) sub-population F burned in May-June 2008, and (d) sub-population E burned in July 2008.

NDVI (2010)
(Departure from 2005 NDVI)

Sisal Fire

Keyhole Fire

Population B

0   0.5   1   2 Km

Aerojet Fire

Population D

% Change in NDVI

	0.46 - 0.63
	0.64 - 0.69
	0.7 - 0.74
	0.75 - 0.79
	0.8 - 0.85
	0.86 - 0.91
	0.92 - 0.97
	0.98 - 1.1
	1.2 - 1.2
	1.3 - 1.4

0   0.25   0.5   1 Km
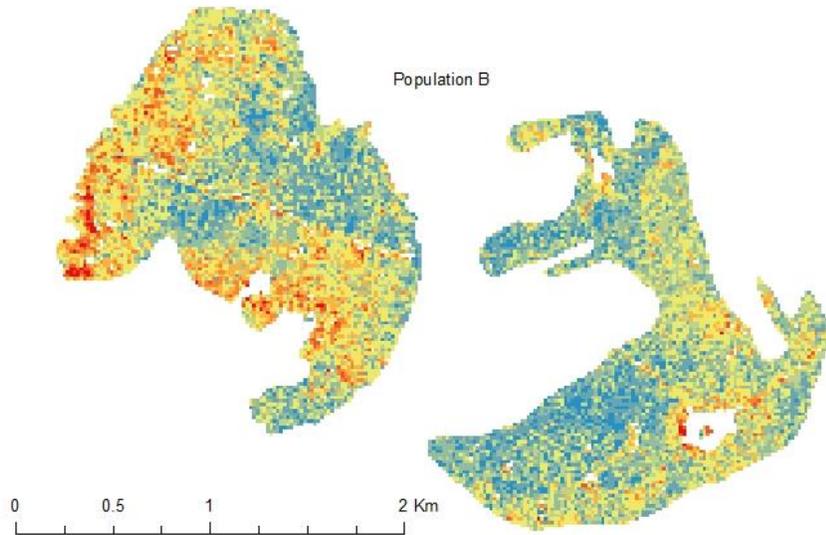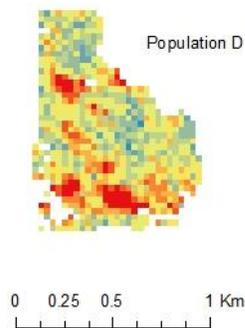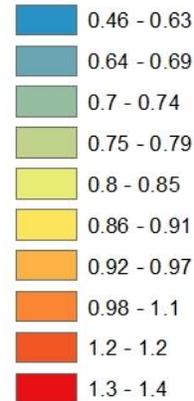
Figure 1.17: NDVI (Normalized Difference in Vegetation Index) values in 2010 (Post-fire Year-5) as a percentage of NDVI in 2005 (pre-fire) in two areas, one burned in May 2005 (Sub-population D) and the other in August 2005 (Sub-population B).
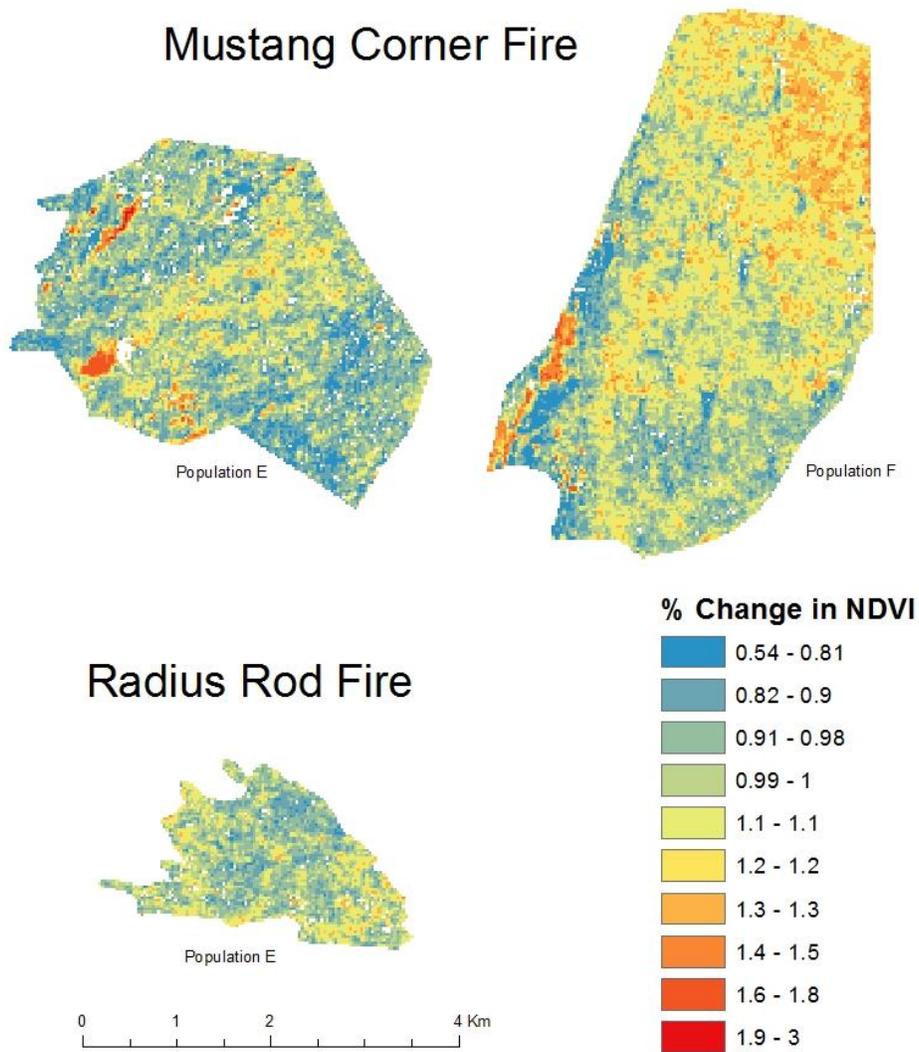
Figure 1.18: NDVI (Normalized Difference in Vegetation Index) values in 2014 (Post-fire Year-6) as a percentage of NDVI in 2008 (pre-fire) in two areas, one burned in May-June (Mustang Corner fire in Sub-population F & E by) and the other in July 2008 (Radius Rod fire in Sub-population E).

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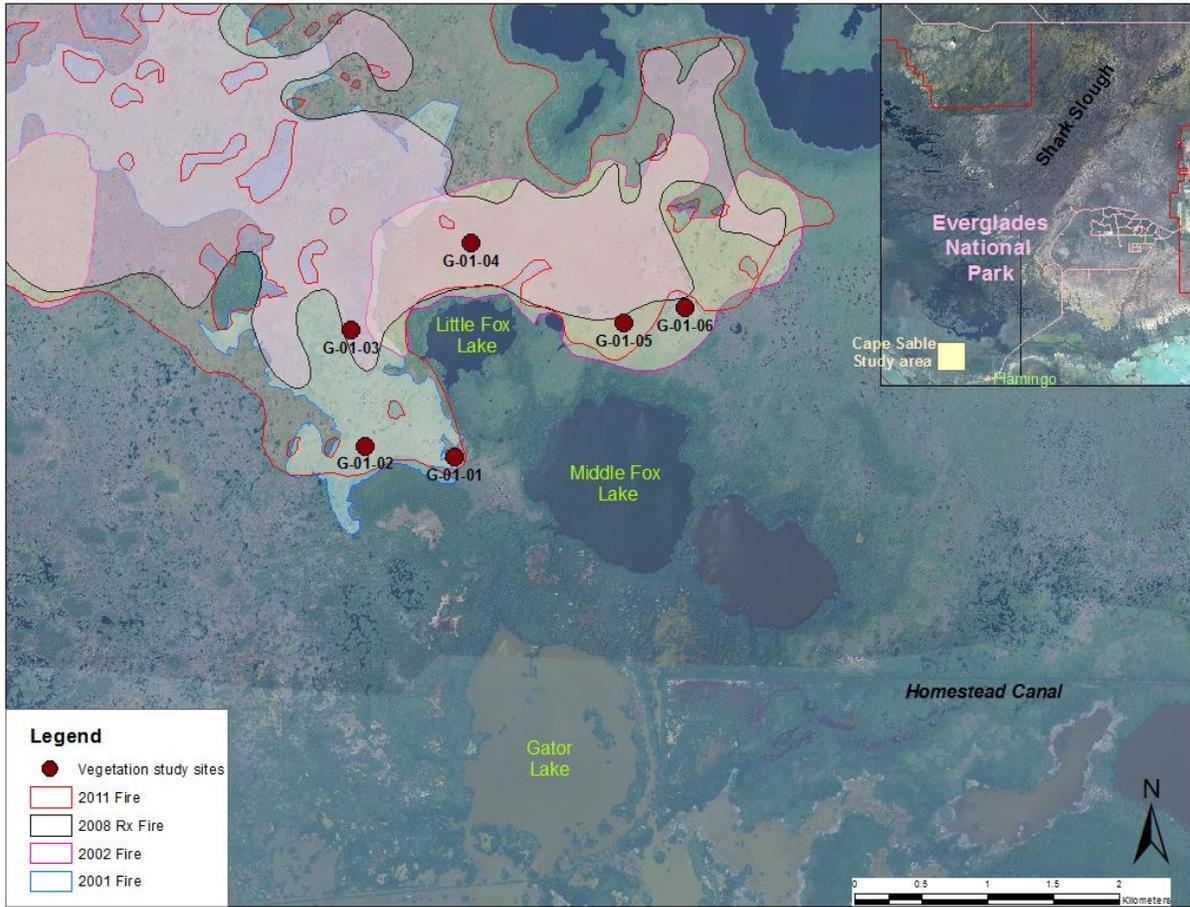
Figure 2.1: Map showing the Cape Sable coastal prairie sites sampled in 2004, 2008 and 2014.
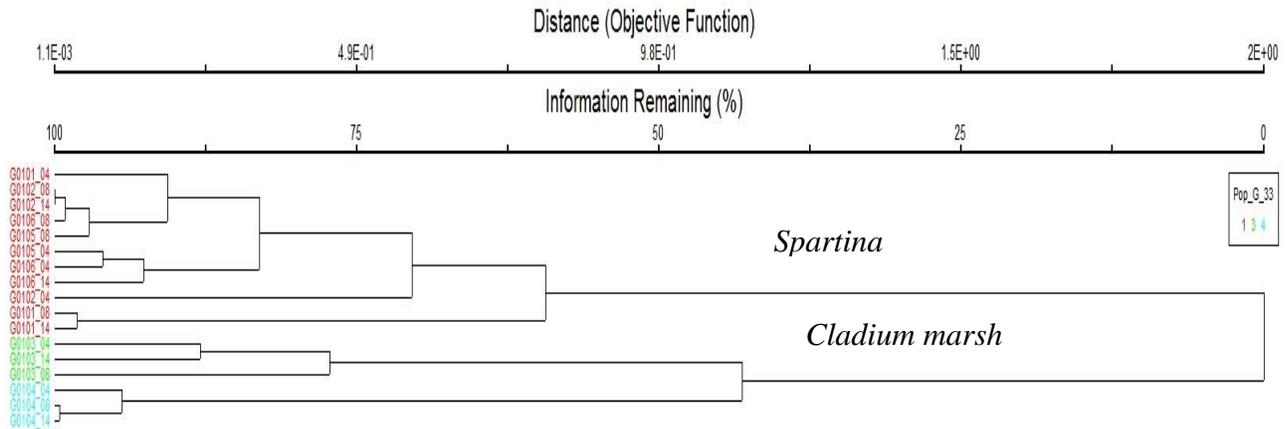
Figure 2.2: Vegetation types identified through cluster analysis of species cover values at 6 census sites sampled in in 2004, 2008 and 2014. Information remaining (%) is based on Wishart's objective function, following McCune and Grace (2002).
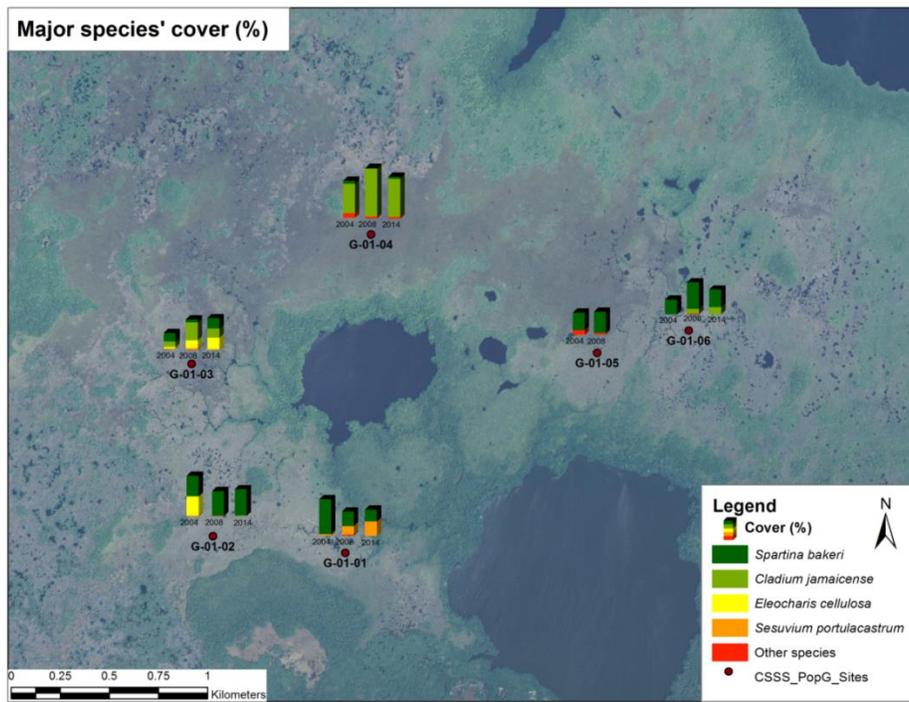
Figure 2.3: Major species' cover (%) at Cape Sable coastal prairie sites sampled in 2004, 2008 and 2014.

Appendices

Appendix 1: Delta and slope (amount and rate of change in the target direction, respectively) calculated for sites burned in 2003 (2), 2005 (21) and 2008 (63). The 2003 and 2005 burned sites were monitored for 4 and 5 years after fire, respectively. The 2008 burned sites were sampled in 1st, 2nd and 6th year after fire. The base year for change in vegetation was the 1st year after fire, and the vector from the base year to the individual pre-burn sites in the non-metric multidimensional scaling (NMDS) ordination was the target direction. Statistical significance ($p \leq 0.1$) of delta and slope was tested using Monte Carlo's simulations with 10,000 permutations.

| POP	FIUID	Burn Year	FIRE	Delta	Prob	Slope	Prob
A	A-09-04	2008	Lime Tree	0.059	0.416	0.021	0.378
A	A-09-06	2008	Lime Tree	-0.197	0.938	-0.039	0.937
A	A-11-05	2008	West Camp	0.040	0.362	0.012	0.312
A	A-16-01	2008	Lime Tree	-0.143	0.828	-0.023	0.773
A	A-17-01	2008	Lime Tree	-0.003	0.513	0.001	0.492
A	A-17-02	2008	Lime Tree	-0.073	0.623	-0.021	0.666
A	A-17-03	2008	Lime Tree	-0.180	0.719	-0.037	0.740
A	A-19-08	2008	West Camp	0.106	0.394	0.029	0.333
A	A-19-09	2008	West Camp	0.228	0.034	0.046	0.037
A	A-19-10	2008	West Camp	-0.312	0.979	-0.057	0.972
A	A-21-10	2008	Lime Tree	-0.090	0.753	-0.021	0.788
A	A-23-01	2008	Lime Tree	0.105	0.384	0.030	0.365
A	A-23-08	2008	West Camp	0.056	0.292	0.013	0.274
A	A-23-09	2008	West Camp	-0.207	0.920	-0.040	0.920
A	A-23-10	2008	West Camp	0.418	0.034	0.087	0.023
A	A-24-06	2008	Lime Tree	0.027	0.448	0.007	0.428
B	B-01-01	2003	Reference	0.285	0.096	0.075	0.144
B	B-01-04	2003	Reference	0.247	0.058	0.063	0.086
B	B-05-06	2005	Keyhole	-0.139	0.790	-0.043	0.855
B	B-05-07	2005	Keyhole	-0.035	0.576	-0.001	0.529
B	B-05-08	2005	Keyhole	-0.179	0.768	-0.049	0.785
B	B-06-05	2005	Sisal	-0.273	0.924	-0.086	0.969
B	B-06-07	2005	Sisal	-0.179	0.875	-0.044	0.894
B	B-06-08	2005	Sisal	-0.322	0.898	-0.093	0.939
B	B-10-03	2005	Sisal	-0.313	0.871	-0.074	0.871
B	B-10-05	2005	Keyhole	0.046	0.395	0.022	0.291
B	B-10-09	2005	Keyhole	0.193	0.295	0.049	0.260
B	B-11-03	2005	Sisal	-0.102	0.688	-0.035	0.756
B	B-11-04	2005	Sisal	-0.290	0.908	-0.067	0.906
B	B-11-05	2005	Sisal	-0.196	0.808	-0.042	0.805
B	B-13-10	2005	Sisal	-0.104	0.690	-0.026	0.683
D	D-01-10	2005	Aerojet	-0.268	0.851	-0.066	0.876

POP	FIUID	Burn Year	FIRE	Delta	Prob	Slope	Prob
E	E-01-07	2008	Mustang Corner	-0.028	0.516	-0.002	0.499
E	E-01-08	2008	Mustang Corner	0.738	0.051	0.151	0.042
E	E-03-02	2008	Radius Rod	-0.235	0.825	-0.044	0.807
E	E-03-07	2008	Radius Rod	0.322	0.180	0.067	0.168
E	E-03-09	2008	Radius Rod	-0.100	0.580	-0.014	0.561
E	E-04-01	2008	Mustang Corner	-0.088	0.645	-0.021	0.663
E	E-05-03	2008	Mustang Corner	-0.039	0.582	-0.003	0.543
F	F-01-01	2008	Mustang Corner	0.186	0.153	0.043	0.116
F	F-01-02	2008	Mustang Corner	0.227	0.126	0.042	0.139
F	F-01-03	2008	Mustang Corner	0.354	0.055	0.060	0.087
F	F-01-04	2008	Mustang Corner	0.703	0.038	0.148	0.023
F	F-02-02	2008	Mustang Corner	0.031	0.432	0.003	0.459
F	F-02-03	2008	Mustang Corner	-0.262	0.901	-0.045	0.865
F	F-02-04	2008	Mustang Corner	0.220	0.161	0.053	0.112
F	F-02-05	2008	Mustang Corner	-0.069	0.576	-0.007	0.523
F	F-02-06	2008	Mustang Corner	0.585	0.006	0.127	0.002
F	F-03-01	2008	Mustang Corner	0.330	0.080	0.068	0.070
F	F-03-02	2008	Mustang Corner	0.161	0.143	0.030	0.159
F	F-03-03	2008	Mustang Corner	0.021	0.472	0.006	0.452
F	F-03-04	2008	Mustang Corner	0.123	0.301	0.037	0.213
F	F-03-05	2008	Mustang Corner	0.332	0.104	0.077	0.078
F	F-04-03	2008	Mustang Corner	0.428	0.013	0.090	0.011
F	F-04-04	2008	Mustang Corner	0.446	0.007	0.090	0.005
F	F-04-05	2008	Mustang Corner	0.469	0.007	0.094	0.008
D	TD-1900	2005	Aerojet	-0.105	0.574	-0.045	0.675
D	TD-2000	2005	Aerojet	0.015	0.478	-0.007	0.513
D	TD-2100	2005	Aerojet	0.083	0.405	0.028	0.378
D	TD-2200	2005	Aerojet	-0.026	0.523	0.002	0.471
D	TD-2300	2005	Aerojet	0.141	0.301	0.031	0.326
D	TD-2400	2005	Aerojet	0.175	0.373	0.019	0.446
D	TD-2500	2005	Aerojet	-0.450	0.684	-0.120	0.725
F	TF-0900	2008	Mustang Corner	-0.095	0.657	-0.017	0.644
F	TF-1000	2008	Mustang Corner	0.089	0.379	0.014	0.405
F	TF-1100	2008	Mustang Corner	-0.035	0.560	-0.002	0.511
F	TF-1200	2008	Mustang Corner	-0.260	0.917	-0.048	0.900
F	TF-1300	2008	Mustang Corner	0.209	0.229	0.038	0.241
F	TF-1400	2008	Mustang Corner	0.242	0.266	0.056	0.249
F	TF-1500	2008	Mustang Corner	-0.091	0.681	-0.018	0.678
F	TF-1600	2008	Mustang Corner	0.379	0.096	0.078	0.089
F	TF-1700	2008	Mustang Corner	0.134	0.315	0.035	0.258
F	TF-1900	2008	Mustang Corner	0.460	0.034	0.101	0.017

POP	FIUID	Burn Year	FIRE	Delta	Prob	Slope	Prob
F	TF-2000	2008	Mustang Corner	-0.195	0.675	-0.027	0.608
F	TF-2100	2008	Mustang Corner	-0.325	0.921	-0.059	0.900
F	TF-2200	2008	Mustang Corner	0.456	0.455	0.119	0.370
F	TF-2300	2008	Mustang Corner	0.160	0.396	0.041	0.379
F	TF-2400	2008	Mustang Corner	0.414	0.131	0.089	0.105
F	TF-2500	2008	Mustang Corner	0.139	0.376	0.029	0.376
F	TF-2600	2008	Mustang Corner	0.157	0.334	0.041	0.288
F	TF-2700	2008	Mustang Corner	0.301	0.134	0.061	0.127
F	TF-2800	2008	Mustang Corner	0.490	0.095	0.097	0.087
F	TF-2900	2008	Mustang Corner	0.281	0.220	0.048	0.244
F	TF-3000	2008	Mustang Corner	0.041	0.483	0.003	0.506
F	TF-3100	2008	Mustang Corner	0.424	0.049	0.097	0.026
F	TF-3200	2008	Mustang Corner	-0.126	0.659	-0.007	0.541

Appendix 2: Qualitative information on species' salt tolerance gathered from the literature and web-based databases

Species	Salt tolerance (level)	Salt tolerance (value)	Source
Acrostichum aureum	*Moderate*	5	IRC, 2014
Acrostichum danaeifolium	*Moderate*	5	IRC, 2014
Agalinis maritima	*High*	10	Go Botany, New England Wild Flora Society
Amaranthus australis	NA	0	
Blechnum serrulatum	*Low*	1	IRC, 2014
Cladium jamaicense	*Moderate*	5	IRC, 2014; Rockledge Garden
Cynanchum angustifolium	*Moderate*	5	IRC, 2014
Cynanchum blodgettii	*Low*	1	IRC, 2014
Cyperus haspan	*Low*	1	Eleuterius and McDonald, 1978
Cyperus polystachyos	*Moderate*	5	
Distichlis spicata	*High*	10	IRC, 2014; Miyamoto et al. 2004
Eleocharis cellulosa	*Moderate*	5	IRC, 2014; http://www.fnps.org/
Hydrolea corymbosa	*Low*	1	IRC, 2014
Hydrocotyle umbellata	*Low*	1	http://www.fnps.org/
Kosteletzkya virginica	*Moderate*	5	IRC, 2014
Laguncularia racemosa	*High*	10	IRC, 2014; http://www.fnps.org/
Ludwigia alata	*Moderate*	5	Eleuterius and McDonald, 1978
Lygodium microphyllum	NA	0	
Mikania scandens	*Low*	1	IRC, 2014
Myrica cerifera	*Moderate*	5	Black 2003; Ferguson 1952
Pluchea odorata	*Low*	1	http://www.fnps.org/
Pluchea rosea	*Low*	1	http://www.fnps.org/
Sabal palmetto	*Moderate*	5	IRC, 2014
Sarcostemma clausum	NA	0	
Schinus terebinthifolius	*Moderate*	5	Ferguson 1952
Sesuvium portulacastrum	*High*	10	IRC, 2014; Rockledge Garden
Spartina bakeri	*High*	10	http://www.fnps.org/
Vicia acutifolia	NA	0	