Cape Sable seaside sparrow habitat Monitoring and Assessment - 2010

(Cooperative Agreement # W912HZ-10-2-0025)

Final Report

Submitted to:
Dr. Al F. Cofrancesco
U. S. Army Engineer Research and Development Center (U.S. Army – ERDC)
3909 Halls Ferry Road, Vicksburg, MS 39081-6199
Email: Al.F.Cofrancesco@usace.army.mil

Jay P. Sah[1], Michael S. Ross[1,2], Pablo L. Ruiz[1], James R. Snyder[3], Diana Rodriguez[1], W. T. Hilton[3]

1. Southeast Environmental Research Center, Florida International University, Miami, FL
2. Department of Earth and Environment, Florida International University, Miami, FL
3. USGS, Southeast Ecological Science Center, Big Cypress National Preserve, Ochopee, FL

April 30, 2011

Southeast Environmental Research Center
11200 SW 8th Street, OE 148 ◆ Miami, Fl 33199 ◆ Tel: 305.348.3095 ◆ Fax: 305.348 4096 ◆ www.fiu.edu

Cape Sable seaside sparrow: Habitat Monitoring and Assessment - 2010

Summary

For the last two decades, the Cape Sable seaside sparrow (CSSS), a federally endangered species, has been a pivot point for water management operations in the Everglades, primarily because a decline in sparrow population in the early 1990s was attributed in part to management-induced alterations in hydrologic regimes. With a goal of understanding the response of landscape-level processes to hydrological restoration and its interaction with fire, a study intended to monitor vegetation structure and composition throughout the marl prairie landscape has been conducted since 2003 with funding from U.S. Army Corps of Engineers (USACE). In the first three years (2003-2005), vegetation structure and composition was characterized in relation to the existing hydrologic regime and fire history. During 2006-2010, vegetation was re-sampled to assess vegetation change within the sparrow habitat. This document summarizes the vegetation change pattern observed between the two sampling periods in sub-population A, C, E and F, emphasizing the work accomplished in FY 2010.

We used Analysis of Similarity (ANOSIM) to test differences in vegetation composition, and vegetation-inferred hydroperiod to quantify the change in vegetation in response to hydrologic differences between the two sampling periods (2003-2005 and 2006-2009). We reasoned that a significant change in vegetation-inferred hydroperiod over time would support the hypothesis that vegetation at the sampled sites changed in response to hydrologic alterations. Additionally, we used trajectory analysis to quantify the rate of vegetation change in relation to hydrology at unburned sites, and to time since last fire at burned sites. Patterns of vegetation change within the sparrow habitat were spatially differentiated. In the sub-population A, west of the Shark Slough, vegetation changed considerably over the sampling period (2003-2009), but with distinct temporal and spatial trends. Between 2003 and 2006, the compositional change was relatively uniform, towards a wetter type. However, in subsequent years, vegetation change showed a distinct spatial pattern that was consistent with the water level records from the P-34 and NP-205 stage recorders. The eastern part of sub-population A, near NP-205, showed a drying trend, while vegetation in the extreme west, represented best by P-34, became wetter in the late 2000s than earlier in the decade. The wetting trend in the western part of sub-population A continued despite achievement of the mandated regulatory water levels at NP-205. In sub-populations C and F in the East Everglades, where reduction in sparrow numbers has been attributed to an increased fire frequency associated with excessive drying, the shift in plant species composition was indicative of relatively wet conditions, likely in response to increased wetness resulting from pumping of canal water into impoundments along the eastern boundary of Everglades National Park (ENP).

In summary, vegetation within the CSSS habitat tracked hydrologic changes, and at burned sites, shifted in conjunction with time since last fire and post fire hydrologic conditions. Moreover, while hydrating the Rocky Glades has helped to improve CSSS habitat along the eastern boundary of ENP, the wetting trend in the western portion of the sub-population A reflected deteriorating habitat conditions for the Cape Sable sparrow. Hence, formulation of a strategy that achieves desirable sparrow habitat conditions while satisfying the broader ecosystem restoration goals of the Comprehensive Everglades Restoration Plan (CERP) is needed.

Table of Contents

Cover photo: Marl prairie vegetation on a transect in an area burned in 2008 in the Sub-population F.

1. General Background

For last two decades, the Cape Sable seaside sparrow (CSSS), a federally endangered species, has remained a focus of several water management operations in the Everglades, primarily because a decline in CSSS population in the early 1990s was attributed to management-induced alterations in hydrologic regimes. Within the sparrow habitat, marl prairie vegetation is sensitive to changes in both hydrologic and fire regimes. With a broad goal of assessing the response of marl prairie ecosystems to water management efforts under Comprehensive Everglades Restoration Plan (CERP), a study intended to characterize marl prairie vegetation and monitor its responses to changes in hydrologic and fire regime within CSSS habitat was initiated 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, while in the following five years, between 2006 and 2010, sub-sets of sites in each of six sparrow sub-populations (A, B, C, D, E and F) were re-visited annually to assess spatio-temporal changes in vegetation in response to changes in hydrologic regimes and fire events. 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 in 2005 provided an opportunity to learn more about vegetation response to fire-hydrology interactions, sites were sampled annually.

This document summarizes the vegetation change pattern observed during 2003-2010, particularly in sub-populations A, C, E and F, emphasizing the work accomplished in FY 2010 (contract # W912HZ-10-2-0025). In FY 2010, the major activities included field work, data analysis, and presentations. Field sampling was accomplished between May 3 and June 7, 2010, and the data were processed in the remaining part of the year. Once the data were analyzed, Jay Sah (Co-PI) presented the results at the Greater Everglades Ecosystems Restoration (GEER)-2010 Meeting, July 12-16, Naples, FL, and at the Cape Sable seaside sparrow (CSSS) Fire Meeting 2010, held on December 7 at the Krome Center, Homestead, Florida.

The report is organized in three sections, describing vegetation responses to changes in hydrologic and fire regimes within CSSS habitat. Section 2 describes a general trend in vegetation responses to spatial differences in hydrologic changes within the sub-population A, located west of Shark River Slough. The section also includes results from in-depth analysis of vegetation change along a 5 km transect placed near stage recorder NP-205, which has served as an indicator of hydrologic conditions within sub-population A for water management purposes. Section 3 of the Report highlights the incipient effects of hydrologic changes due to operations of water structures B, S332-C and D and the adjoining detention ponds. Finally, Section 4 describes the effects of fire and flooding at sites that were burned and flooded within 2 to 8 weeks thereafter in 2005. The sites have been sampled annually for five years after fire, leading us to analyze the data using trajectory analysis. The analysis in the Section updates through 2010 an earlier examination of vegetation change in these plots that was described in our FY-2009 Report.

2. Linking vegetation dynamics to hydrologic changes in the habitat of Cape Sable seaside sparrow sub-population A

2.1 Introduction

Ecosystem management practices often direct the major drivers and stressors of a system to achieve various levels of desirable ecosystem services, while maintaining its characteristics in the natural conditions as closely as possible. Such alterations in the environmental drivers, however, frequently lead to changes in the plant community composition (Collins 2000; Chapin et al. 2006). Wetland plant community composition often is very sensitive to hydrologic change, the major driver of wetland ecosystem functions. However, severe anthropogenic modifications of hydrologic regime in wetland systems around the globe have resulted in deterioration of wildlife habitat. One example is the Southern Everglades marl prairie landscape inhabited by the Cape Sable seaside sparrow (CSSS), a federally listed endangered species that has gone through management-induced hydrologic changes. The question we address here is how closely vegetation changes in marl prairie landscape mimic changes in hydrologic regimes.

During the 20th century, the wetlands of the Florida Everglades experienced unprecedented interferences from human activities. Central among these activities was water management, accomplished through the construction and operation of canals, levees, spillways, pump stations, and various other water control structures (Light and Dineen 1994). These water control systems have been considered a major culprit in the degradation of numerous areas of the Greater Everglades, including the marl prairies west of Shark River Slough that provide habitat for Cape Sable seaside sparrow sub-population A. Before 1993, sub-population A was the largest of six CSSS sub-populations. However, the numbers of sparrows dropped drastically owing to unprecedented water release during 1993 breeding season (Pimm et al. 2002). Since then, this sub-population has been the focus of water management efforts to improve habitat conditions for the Cape Sable seaside sparrow. As a part of these efforts, regulatory schedules for the S-12 structures along Tamiami Trail have been followed under the operational objectives of Interim Structural and Operation Plan (ISOP)/Interim Operational Plan (IOP), whose objective is to maintain NP-205 stage $\leq$ 6 ft NGVD for a minimum of 60 consecutive days between March 1 and July 15, thus avoiding flooding during the sparrow breeding season. In concurrence with management efforts to regulate water deliveries from the S-12 structures, a consistently low water level has been maintained at NP-205 and nearby in the eastern part of this sub-population. However, the same conditions have not been observed at P-34 in the western part of the sub-population A, where relatively high water level has persisted in recent years. Since the western parts of the sub-population A also receive runoff from the water basin intersected by a section of Tamiami Trail between Forty-Mile Bend and Monroe Station (FMB-Monroe), and by the Loop Road, the hydrologic conditions in this area are influenced by the spatial and temporal variation in flows through the culvert and bridges on these two roads. In a recent analysis of the flow data in relation to rainfall, Kotun et al. (2009) showed that mean annual runoff per unit rainfall in the FMB-Monroe sub-basin increased by a factor of two during 1992-2008 in comparison to three earlier periods (1941-1952, 1953-1963 and 1964-1991). They attributed the increased runoff to high stage level in WCA-3A, which resulted in a backwater effect in Mullet Slough, causing water to flow southwest towards Big Cypress National Preserve, and ultimately ending up in increased flow across the Tamiami Trail. Moreover, the increase in flow was much greater at the bridges in the east, close to the L28 (Kotun et al. 2009), which apparently contributed to high water levels in the western part of sub-population A.

One of the goals of implementing regulatory schedules on S12 water deliveries and maintaining low water levels at NP-205 was to restore sub-population A habitat by reversing the trend in vegetation triggered by high local water levels during most of the 1990s (Nott et al. 1998; Pimm et al. 2002). The hypothesis that led to implementation of such efforts was that the regulations would result in a shift in vegetation composition towards a drier type that is more suitable for sparrows. We examined this hypothesis through two questions, (a) has the vegetation composition in the sub-population A changed monotonically along the hydrology gradient in response to the recent management efforts?, and (b) is a trend in temporal changes in vegetation pattern in concurrence with observed spatial differences in hydrologic conditions within the sub-population.

2.2 Method

2.2.1 Field Data

Between 2003 and 2005, a network of 901 sampling sites was established and vegetation was sampled for the first time. The network included 293 sites at 100 or 200 m intervals along six transects of 2.5 to 11 km, and at 608 sparrow census sites distributed throughout the recent range of six sparrow sub-populations, with the exception of six sites in coastal prairie at Cape Sable, where sparrows were present in the early part of the 20th century (**Figure 2.1**). Vegetation sampling at each sampling location included records of species cover in ten 0.25 m^2 sub-plots in a 60x1 m^2 plot, and structural measurement of vegetation in 30 sub-plots (Ross et al. 2006). At the transect sites, we determined elevations in the ten compositional sub-plots, and estimated hydroperiod for the previous six years, using water level data from nearby stage recorders.

In sub-population A, the sampling network included 51 sites along a 5-km transect and 280 census sites, of which 269 unburned sites were re-sampled for vegetation composition during 2006-2009. Sites that were not burned for four years prior to sampling were considered to be unburned. Initially sampled in 2003, the transect sites in this area were re-sampled in 2006 and 2010.

2.2.2 Analytical Methods

We summarized the data using Non-metric Multi-dimensional Scaling (NMS) ordination, and used Analysis of Similarity (ANOSIM) to quantitatively examine the differences in vegetation composition among years. In the analysis, we used the Bray-Curtis distance metric as a measure of dissimilarity. In ANOSIM, an R-statistic is generated based on the mean ranks in ecological distance among groups and within groups; an absolute value of the R-statistic close to 1 suggests a real difference in vegetation composition among groups (Clarke 1993). In our analysis, the groups were different sampling years.

Additionally, we used an approach similar to Armentano et al. (2006) to assess whether a change in vegetation composition between sampling years was a response to periodic differences in hydrology. We analyzed the differences in mean vegetation-inferred hydroperiod, i.e. the hydroperiod for a site predicted from vegetation composition using a Weighted Averaging Partial Least Square (WAPLS) regression model developed in 2005 (Ross et al. 2006). The model was validated using field water depth-based hydroperiods, calculated using Everglades Depth Estimation Network (EDEN) stage data. Our assumption was that a significant change in vegetation-inferred hydroperiod between sampling years supports the hypothesis that vegetation change between subsequent sampling events was a response to hydrological change.

Trajectory analysis: In sub-population A, west of Shark River Slough, vegetation change along Transect-A was analyzed using the data collected in 2003, 2006 and 2010. Change in vegetation composition along the hydrologic gradient in Transect was analyzed using trajectory analysis (Minchin et al. 2005). This analytical procedure allows researchers to test statistically the change in community composition along a target vector represented by an environmental gradient in NMDS ordination space. A vector representing the hydrologic gradient in the ordination was defined by using vector fitting technique in DECODA (Minchin 1998). In this method, a gradient is defined in the direction through the 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 was tested using a Monte-Carlo permutation test with 10,000 random permutations (Minchin 1998). The orientation of the ordination was then rotated so that 5-year average hydroperiod had a perfect correlation ($r = 1.0$) with Axis-1.

Two statistics, delta (Δ) and slope, were calculated to quantify the degree and rate of change in vegetation composition along the reference vector (Minchin et al. 2005). Delta, a measures of the total amount of change in the target direction, was calculated as the difference between projected scores of the final and initial time steps. Slope measures the mean rate of change in community composition along the target vector. Since NMS ordination was scaled in half-change units, the rate was mean half change per year, where one half-change is the distance between sampling units at which their mean similarity is 50% of the value between sampling units with similar species composition and identical coordinates in the ordination space. In our analysis, the slope was calculated as the linear regression coefficient of projected scores on target vector on sampling years after intervention. The statistical significance of both delta (Δ) and slope was tested using Monte Carlo simulations with 10,000 permutations.

Change in species abundance: A trend in species abundance along hydrology vector was analyzed by fitting curves to the species cover data for the species present in at least 25 samples among sites that showed significant trajectory shift along the hydroperiod vector. Species response curves were fitted using Generalized Linear Models (GLM) with Poisson error distribution and log link function (McCullagh and Nelder 1989). Poisson distribution assumes that variance equals to the mean, and is considered appropriate for count data, and we used it instead of a Gaussian (i.e. normal) distribution with an identity link, because the cover values for the majority of species were skewed. A model using the Poisson distribution consistently yielded lower deviance – a criterion used to select the best fit model – than the corresponding model with the normal distribution. In addition, the Poisson distribution with log link function also allowed us to constrain the predictions within a range of nonnegative values of species cover, the response variable in the model. Assuming unimodal, symmetric or skewed responses of species to the environmental gradient, we started with second-order polynomial model, and used a stepwise backward function to select the significant variables. A χ_2-test was performed to test the significance of coefficients of linear and quadratic terms, and terms with non-significant coefficients ($p<.05$) were removed from the final model. Finally, for the species which showed a significant trend in abundance along the hydrology vector, we calculated differences in mean species cover among sampling years for the whole transect. A pair-wise t-test was used to test the significance of mean differences.

2.3 Results

In sub-population A, vegetation changed considerably over the sampling period (2003-2009), with distinct temporal and spatial trends. In general, at un-burned sites sampled twice in seven years, the vegetation composition differed between two sampling periods, 2003-2005 and 2006-2009 (ANOSIM: 1000 permutations, Global R = 0.01, p-value = 0.004). The trajectory analysis results revealed that the rate (slope) of vegetation change along the hydrology vector was statistically significant at 26.8% of sites (**Appendix A-1**). However, the direction of change along the hydrology gradient was not consistent (**Figure 2.2**), and the wetting and drying trends were also geographically differentiated (**Figure 2.3**). Among the sites that showed a significant trajectory shift with increasing hydroperiod, the majority were first sampled in 2003 and again in 2006. The results suggested that the three-year period (2003-2006) was marked by a general shift in vegetation towards more hydrophytic type (**Figure 2.2**), a conclusion also supported by an increase in vegetation-inferred hydroperiod in those three years within the sub-population (**Figure 2.4a**). In contrast, at the sites first sampled in 2004 or 2005 and again re-sampled during the period 2007 and 2009, the direction of change in vegetation composition differed spatially among sites. In the western and south-eastern parts of the sub-population, vegetation inferred hydroperiod increased at several sites, whereas an opposite trend was observed at the sites in the northeastern and southern parts of the sub-population (**Figure 2.4b, c**).

Plant species in the Everglades marl prairie showed increasing, decreasing or unimodal trends along the hydroperiod gradient. In Sub-population A, the species that showed a significant increase in species cover with increasing wetness were *Bacopa caroliniana*, *Cladium mariscus* ssp. *jamaicense, Eleocharis cellulosa*, *Panicum hemitomon*, *Paspalidium geminatum*, *Rhynchospora tracyii and Sagittaria lancifolia*. These species are known to be the characteristics of long hydroperiod conditions in the Everglades. Cover of a few species that are usually present in dry sites decreased with an increase in wetness along the hydrology vector. They were *Cassytha filiformis*, *Centella asiatica*, and *Schizachyrium rhizomatum*. Several species, including *Crinum americanum*, *Panicum tenerum, Panicum virgatum, Paspalum monostachyum, Pluchea rosea,* and *Schoenus nigricans* showed unimodal distribution.

The vegetation change pattern observed at the census sites was also evident at sites along the Transect A in the eastern portion of the sub-population. The transect extends east (2 km) and west (3 km) from NP-205, and represents the hydrologic conditions within the sub-region well. Vegetation composition along the transect significantly differed between sampling years

(ANOSIM: Global R = 0.229, p-value <0.001). However, the differences in composition were stronger between 2006 and 2010 (R = 0.280) than between 2003 and 2006 (R=0.136). The vegetation change between 2003 and 2006 was marked by a significant reduction of total ground cover as well as the absolute cover of several major species. In that period, however, the change in species cover does not seem to be limited only to species that are indicative of either wetter or drier environments, as the change in vegetation-inferred hydroperiod between 2003 and 2006 was not statistically significant (**Figure 2.5**). In contrast, vegetation-inferred hydroperiod was significantly lower in 2010 than in both the 2003 and 2006 samples, suggesting that the compositional change through 2010 was marked by a significant increase in the cover of species indicative of dry conditions, and decrease in the cover of species characteristic of wet conditions. The change in vegetation-inferred hydroperiod on the transect coincided with parallel changes in hydroperiod referenced to stage level at NP-205 (**Figure 2.5**).

Between 2003 and 2010, 80% of the sites on Transect-A took on an opposite trajectory along the vector of increasing hydroperiod, suggesting a trend from wetter to drier conditions (**Appendix A-2**). At those sites, vegetation-inferred hydroperiod also decreased (**Figure 2.6**). Trajectory analysis results revealed that the amount (delta) and rate (slope) of vegetation change over the seven year period were statistically significant at 50% of the sites. At the sites that showed significant shift in trajectory along hydrologic vector, the mean change towards drier vegetation, represented by a shift along X-axis (**Figure 2.7**), was more prominent between 2006 and 2010 than between 2003 and 2006.

On Transect A, the mean cover of *Bacopa caroliniana*, *Cladium mariscus* ssp. *jamaicense*, *Rhynchospora tracyii* and *Schoenus nigricans* significantly decreased, whereas the mean cover of *Schizachyrium rhizomatum* increased by more than two-fold within the same period (**Table 1**). Mean cover of *C. mariscus* spp. *jamaicense* were 13.2% and 10.8% in 2003 and 2010, respectively. In contrast, mean cover of *S. rhiozomatum*, a dominant species in short-hydroperiod prairies, increased from 3.69% in 2003 to 7.73% in 2010. Other species whose mean cover significantly (pair-wise t-test; p<0.001) increased between 2003 and 2010 were *Centella asiatica*, *Crinum americanum* and *Panicum virgatum*.

Table 2.1: Mean cover (%) of major species on Transect A in 2003, 2006, & 2010. Different superscript letters indicate significant difference (Pair-wise t-test; p-value <0.05) in species' cover between years.

Species	2003	2006	2010
Bacopa caroliniana	0.36[a]	0.87[b]	0.20[c]
Centella asiatica	0.27[a]	0.18[b]	1.45[c]
Cladium mariscus ssp. *jamaicense*	13.20[a]	10.18[b]	10.83[b]
Crinum americanum	0.18[a]	0.35[b]	0.63[c]
Panicum tenerum	1.38[a]	0.28[b]	1.86[a]
Panicum virgatum	1.15[a]	1.07[a]	2.28[b]
Paspalum monostachyum	2.83[a]	2.02[b]	3.63[a]
Rhynchospora tracyi	1.36[a]	2.57[b]	0.24[c]
Schoenus nigricans	3.98[a]	2.03[b]	2.01[b]
Schizachyrium rhizomatum	3.69[a]	4.18[a]	7.73[b]

2.4 Discussion and Conclusions

In the southern Everglades marl prairies, particularly west of Shark River Slough which is also the habitat of CSSS sub-population A, hydrologic conditions have changed over eight years (2003-2010), mainly due to changes in water management activities. Such alterations in the hydrologic regimes have resulted in changes in vegetation composition that, in harmony with the hydrologic change pattern, showed distinct temporal and spatial patterning. In other studies also, researchers have shown that in seasonally-flooded ecosystems, species assemblages change as a function of duration and depth of seasonal flooding (Breen et al. 1988; Schessl, 1999; van der Valk 2005; Armentano et al. 2006).

In marl prairies within sub-population A, observed changes in vegetation composition in response to hydrologic shift can be explained based on the Flood Pulse Concept (Junk et al. 1989) and Community/Continuum Concept (Whittaker 1967). The flood pulse concept, first developed to describe relationships between seasonal changes in water levels and ecosystem function and the maintenance of species diversity, especially with reference to fisheries, on Amazonian Floodplains (Junk et al., 1989), has recently been extended to explain the vegetation change pattern in response to hydrologic changes in several other wetland systems characterized by seasonal fluctuation in water levels (Niering 1994; Zedler and Callaway 1999). Short-hydroperiod marl prairies in the Everglades are also flooded annually for varying period, and they remain dry for part of the year. Generally, in seasonally-flooded ecosystems similar to the Everglades marl prairies, species present in the vegetation mosaic are adapted to tolerate the alternating wet/dry conditions that are basically a part of any flood-pulsed environment (Junk and Piedade 1997; Middleton 1999). However, the inherent ability of plants to survive and grow under various hydrologic regimes varies among species, and the differences in species' optimum flooding tolerances usually form the basis for variation in vegetation composition in these ecosystems. In marl prairies also, the species differ in their hydroperiod optima and tolerances (Ross et al. 2006). Hence, any change in duration of periodic inundation would affect abundance of various species. This could be the reason that observed change in hydrologic regimes within the landscape in sub-population A probably also caused a change in relative cover of constituent species, resulting in a shift in vegetation composition towards either wetter or drier type. For instance, in the north-eastern part of the sub-population, where duration of annual dry-down increased, possibly owing to reduced deliveries through the S12 water structures, a two-fold increase in cover of blue-stem *Schizachyrium rhizomatum* was observed. *S. rhizomatum* has relatively short hydroperiod optimum and is a dominant species of short-hydroperiod marl prairies (Ross et al. 2006). In contrast, the relative cover of species like *Bacopa caroliniana*, *Cladium mariscus* ssp. *jamaicense*, and *Rhynchospora tracyii*, which are characteristic of relatively long hydroperiod in marl prairies, significantly decreased. These species remained dominant or their cover increased in the areas where hydroperiod has increased in recent years.

In the Everglades marl prairies, where there is a very small gradient in elevation, the breaks between discrete vegetation types, particularly in the herbaceous community, are usually very subtle, suggesting that plant community composition along the topographic gradient varies steadily, with gradual change in relative abundance of constituent species along the gradient - the fundamental basis of continuum concept (Whittaker 1956; Curtis 1959). The resulting changes in hydrologic condition along the gradient are of relatively small magnitude, and would not bring a drastic change in vegetation structure, but simply alter species composition toward more or less hydrophytic assemblages. Our results suggest that the change in hydrologic regimes over eight years have caused a gradual shift in species composition. Depending on the relative contribution of individual species that differ in their optimum hydrologic tolerances, such a shift in species composition in response to temporal and spatial variation in hydrologic changes was clearly expressed in similar changes in vegetation-inferred hydroperiod. Moreover, the spatial pattern in vegetation change was consistent with a change in the hydrologic regime estimated from Everglades Depth Estimation Network (EDEN) water depth data (Sah et al. 2010). The close resemblance between EDEN data-based and vegetation-inferred hydroperiods suggests that the latter can be used to track vegetation response to hydrologic changes in this part of marl prairie landscape.

Finally, the vegetation change towards wetter type in the western part of sub-population A in response to more hydric conditions in recent years in comparison to the early part of the decade is indicative of a deteriorated sparrow habitat within that region. Interestingly, it continued despite achievement of the mandated regulation water levels at NP-205, which resulted in an improved habitat in its vicinity. Because of such spatially differentiated trends in habitat characteristics, the limited numbers of sparrows that remain in sub-population A continue to be restricted to the eastern portion of the habitat (**Figure 2.8**). If increasing sparrow populations west of Shark Slough is the objective, then strategies that achieve desirable sparrow habitat conditions throughout the region while satisfying the broader ecosystem restoration goals of the Comprehensive Everglades Restoration Plan (CERP) should be considered.

3. Hydrating the rocky glades: incipient effects on marl prairie vegetation within the habitat of sub-populations C, E and F

3.1 Introduction

Marl prairie vegetation along the eastern boundary of Everglades National Park (ENP) has been heavily influenced by alternating changes in water management activities. For instance, in the 1960s and 70s, the marl prairies remained dry for extended periods each year (Rose et al. 1981; Van Lent et al 1993), affecting the plant communities. Later, in the 1980s and 90s, a pump station installed on the L31W delivered water directly into Taylor Slough, resulting in extended hydroperiod immediately downstream (Armentano et al. 2006), while marl prairies in the northern and western portions of the basin still remained dry (USACOE 1999). In recent years, however, management efforts have been directed toward shifting point source water deliveries to surface water flow in the Taylor Slough basin, and also re-hydrating the Rocky Glades, an area of thin marl soils and sparse vegetation that encompasses portions of sparrow habitat in sub-populations C, E and F (USACOE 2006). These three sub-populations, located between Shark River Slough and the eastern boundary of ENP, are among five designated critical habitat units that receive protection under section 7 of the Endangered Species Act against destruction and adverse modification of the habitat (USFWS 2007). Among the three, sub-population E holds the second highest number of sparrows (>500), after sub-population B, south of Long Pine Key. Sparrow numbers in sub-populations C and F are very small, usually <50, and in some years even null during the annual sparrow census, such as from 2007 to 2009 in sub-population F. To ensure the conservation of the species, the proper management of these eastern sub-populations and their habitat has become more important than before, especially after the exclusion of sub-population-A habitat in the recent designation of sparrow's critical habitat (USACOE 2006; USFWS 2007).

Hydrology influences sparrow habitat directly or through its impact on the incidence of fire events, thus affecting the sparrow sub-populations (Nott et al. 1998; Lockwood et al. 2003; La Puma et al. 2007; Baiser et al. 2008; Sah et al. 2010). In three of the five eastern sub-populations, sparrow numbers were lower in 1992 than in 1981, the year when the first comprehensive survey of sparrows conducted. While increase in hydroperiod and mean water depth is believed to be the cause of decline in sparrow number in sub-population D, the reduction in sparrow numbers in sub-population F was attributed to an increase in fire frequency associated with the excessive drying of the region (Pimm et al. 2002). Conversely, sub-population C was subjected to both wetting and drying conditions, though spatially differentiated. Beginning in 1980, pump station S332 delivered water from the L31W canal directly into the Taylor Slough, resulting in extended hydroperiod in its vicinity in the southern portion of the sub-population C. Vegetation in this area changed from short hydropeirod muhly grass (*Muhlenbergia capillaris* var. *filipes*) prairie to less suitable sparrow habitat dominated by dense sawgrass (*Cladium mariscus* ssp. *jamaicense*) (Armentano et al. 2006), perhaps also affecting the sparrow occurrences in this area. Concurrently, the northern portion of sub-population C continued to be over-drained due to reduced water level in the canal and in agricultural land immediately adjacent to it, resulting in an increase in fire-frequency and a reduced sparrow population (Pimm et al. 2002).

To mitigate the adverse effects caused by past water management practices and to restore more natural flows, several interim measures have been implemented in recent years, including the construction of a series of water pumps on the adjoining canals and construction of detention ponds along these canals, (USACOE 2006). Major changes in water management activities in the region include operation of the new pump stations and water delivery through the detention basins along the C111 and L-31 canals. In the lower Taylor Slough basin, water pump station S-332D replaced the operation of S-332 in 2000. Since then water has not been delivered directly into the slough, instead moving through a series of 3 interconnected detention ponds in the Frog Pond area and a flow way east of a levee along the border of Everglades Park. In this course, water seeps into the Park due to the hydrologic head thereby created.

In the upper Taylor Slough basin north of D, the B and C pump structures, constructed under Interim Operation Plan (IOP) to provide protection for the adjacent CSSS habitat, deliver water from the L31N canal into a series of inter-connected detention ponds. Detention areas include 64.7 ha (160-acre) and 97.1 ha (240 acre) basins west and north of B, respectively, and a 121 ha (300 acre) basin west of C. A reservoir of 83.4 ha (206-acre) connects the S332B and S332C ponds. The pond north of S332B has a large fixed-crest weir on the western levee that allows water from the pond to enter ENP marl prairies. In addition, subsurface water also may be entering ENP. The purpose of operating pump stations (B and S332-C) along the L-31N canal includes lowering canal and groundwater levels and creating a continuous hydraulic ridge to control seepage back to the canal while protecting the sparrow habitat from further deterioration (USACOE 2006). Pumping through B and C is intended to be adjusted seasonally to maintain the desired water conditions in eastern sparrow habitat within ENP, and thus serve to re-hydrate the marl prairies of the Rocky Glades. The question is whether the marl prairie vegetation composition within the habitat of sparrow sub-populations along the eastern boundary of ENP has changed in response to the recent management efforts.

3.2 Methods

3.2.1 Field Sampling

A network of 901 vegetation sampling sites, established and sampled for the first time between 2003 and 2005, included 124 transect sites and 132 census sites within the habitat of sparrow sub-populations C, E and F (Ross et al. 2006). Among the census sites, 121 unburned sites were re-sampled for vegetation composition during 2006-2009, 3 or 4 years after initial sampling. Sites not burned for four years prior to sampling were considered to be unburned sites. Census sites burned in 2007 or 2008 were sampled annually for 3 and 2 post-fire years, respectively. Additionally, sites on the transect F that included both unburned and burned locations were first sampled in 2004, and again in 2009 and 2010.

Vegetation sampling at each sampling location included records of species cover in ten 0.25 m^2 sub-plots in a 60x1 m^2 plot, and community structure in 30 sub-plots (Ross et al. 2006). At the transect sites, we also determined elevations in the ten compositional sub-plots, and estimated hydroperiod for the previous six years using water level data from nearby stage recorders.

3.2.2 Analytical Methods

We used Analysis of Similarity (ANOSIM) to quantitatively examine the differences in vegetation composition among years. In the analysis, we used the Bray-Curtis distance metric as a measure of dissimilarity. Likewise, vegetation data were also analyzed using trajectory analysis (Minchin et al. 2005; *also see sub-section* 2), which allowed us to examine change in community composition along a pre-defined hydrologic gradient in ordination space. The hydrology vector was derived from census sites at which water depth were measured in the field and plot level hydroperiod was calculated using water level data obtained from Everglades Depth Estimation Network (EDEN) database, and from topographically surveyed transect sites for which hydroperiod was calculated using the nearest stage recorders and the mean plot elevation.

Additionally, we analyzed the differences in mean vegetation-inferred hydroperiod, i.e. the hydroperiod for a site predicted from vegetation composition using a Weighted Averaging Partial Least Square (WAPLS) regression model developed in 2005 (Ross et al. 2006). We considered that significant change in vegetation-inferred hydroperiod between sampling years would support the hypothesis that vegetation at the sampled sites has changed in response to hydrologic variation.

3.3 Results and Discussion

Vegetation composition in the marl prairies in sub-populations C and F along the eastern edge of Everglades National Park, and in sub-population E near the eastern flank of Shark Slough, differed significantly (ANOSIM: $p < 0.05$) between two sampling periods, 2003-2005 and 2006-2009 (**Table 3.1**). In general, sites in these sub-populations showed a change in vegetation composition indicative of increasing wetness in recent years (**Figure 3.1**). However, the vegetation change pattern showed distinct spatial patterning, suggesting that vegetation within the sub-populations was responding to spatially variable hydrologic changes, resulted from the modifications in water management activities, and to the Mustang Corner fire of May-June 2008, which burned a large area in sub-population F, and few census sites in sub-population E.

Table 3.1: Global R and *p*-values from analysis of similarity (ANOSIM) testing temporal differences in vegetation composition at unburned sites that were first sampled between 2003 and 2005, and re-sampled between 2006 and 2009 in CSSS sub-populations C, E and F.

Sub-population	# of sites	R-statistic	p-value
C	35	0.044	0.032
E	56	0.070	<0.001
F	30	0.077	0.014

Sub-population C:

In sub-population C, trajectory analysis results showed that two thirds of unburned sites exhibited a shift towards increasing wetness, while the rest showed a shift in the opposite direction along the hydrologic gradient. The shift in vegetation composition was statistically significant at 25% of sites in each group (**Appendix A-3**). In this sub-population, vegetation change, as evidenced also by a change in vegetation-inferred hydroperiod, was spatially differentiated primarily in response to hydrologic variation. Along the eastern edge of the sub-population, immediately adjacent to L31W and south of water pumping station S332, vegetation-inferred hydroperiod at several sites decreased over the four year period (**Figure 3.2**), suggesting that the vegetation at these sites in 2006-2009 was indicative of drier conditions than in early 2000s. In contrast, an increase in vegetation-inferred hydroperiod at the sites south of the park road, and in the western part of the sub-population indicated an increase in wetness between the two surveys.

The observed pattern in vegetation change in sub-population C during 2003-09 seems to track water management activities in the region. During the first survey (2003-2005), vegetation represented the legacy of water management activities in 1990s. By the mid 90s, water management operations that were initiated in 1981 and involved water deliveries by S332 from L31W canal had raised water levels in Taylor Slough by about 30-40 cm above the levels of previous decades, resulting in a shift in vegetation composition (Armentano et al. 2006). The vegetation at the sites on the Taylor Slough transects had changed from muhly grass (*Muhlenbergia cappilaris* var. *filipes*) to sawgrass (*Cladium mariscus* ssp. jamaicense) or from sawgrass to spikerush (*Eleocharis cellulosa*) dominated communities. After the cessation of S332 operation in 2000, a reverse trend in vegetation change had already begun prior to 2003, and an obvious reduction in spikerush cover at slough sites was reported. But at the muhly-dominated sites on both flanks of the slough, sawgrass cover continued to increase through 2003 (Armentano et al. 2006). In subsequent years, however, the trend in vegetation change at all sites resembled the pattern of 1999 - 2003 at the slough sites, i.e. towards an assemblage indicative of increasing dryness (Sah et al. *manuscript in preparation*). At the CSSS vegetation monitoring sites, we did not have vegetation composition data prior to 2003. However, between our two surveys, a decrease in vegetation-inferred hydroperiod at vegetation census sites in the vicinity of Taylor Slough indicated that a drying trend continued through 2009. This probably was the result of reduced annual mean water level in Taylor Slough, resulting from cessation of direct delivery of water from L31W through the S332, replaced by surface flow through the operation S332D, located north of S332.

Vegetation in the western portion of the sub-population and north of S332 showed an increase in vegetation-inferred hydroperiod, suggesting a shift in vegetation composition indicative of increasing wetness in that area. This wetting trend is attributed to seepage from L31W. Since 1999, water has been delivered from the L31N into the L31W canal thorough a series of detention ponds in ‘Frog Pond’ region and a flow way cell into ENP near S332. However, it is likely that water from the detention ponds also seeps into the L31W canal from where it enters into ENP as sub-surface flows. In a recent study, Gaiser et al. (2008) suggested that in the vicinity of the L31N and L31W canals and adjacent basins, the groundwater contours are parallel to the L31N canal, indicating a predominant direction of groundwater flow toward the east and southeast, that is, away from ENP. However, seasonal variation in ground water contours, and the presence of groundwater mounds or “the “bulls-eye” associated with the water control structures on the L31N and L31W, suggest that water also flows towards the west during certain periods, particularly in the dry season (Gaiser et al. 2008). Water from L31W that flows into ENP, passing through the expanse of marl prairies, finally is drawn into the Taylor Slough, possibly near the bridge on the Everglades National Park road (SR 9336) and south of it. In

Taylor Slough, seasonal mean water level, which was strongly correlated with rainfall during pre- period (1961-1980) and with the delivery during 1980-2003 (Armentano et al. 2006), has more recently been well correlated with the volume of water delivered by D pumping station (**Figure 3.3),** perhaps due an hydrologic connection established through surface and sub-surface flow that also impacts marl prairie vegetation within the area.

Sub-population E

In Sub-population E, median vegetation-inferred hydroperiod between the first and second surveys were significantly different (n=56; Z = 2.86; p = 0.004) over the entire subpopulation E (**Figure 3.1**). However, within the sub-population vegetation change showed a distinct spatial pattern (**Figure 3.4**). The inferred hydroperiod at the sites in a small area in the southwest and in most of the north and eastern part of the sub-population increased over the period, suggesting a shift in vegetation composition in those areas towards relatively wet vegetation. The magnitude of increase in hydroperiod was higher (> 30 days) near the eastern boundary than the middle portion. Trajectory analysis results also showed that the shift in vegetation towards wetness along hydrology vector was statistically significant at several sites in this region, including both unburned and burned sites (**Appendix A-3**). In the northwest and central portion of the habitat, vegetation-inferred hydroperiod decreased between two sampling periods, suggesting a drying trend in the area. The observed spatially differentiated vegetation change patterns within the sub-population suggested an influence of localized hydrologic differences.

A shift in vegetation composition toward wetter type in the eastern portion of the sub-population is possibly in response to the changes in hydrologic regimes due to operation of a series of water pumps and detention areas along the eastern boundary. While water level at the stage recorder CR2, located between detention ponds and sub-population E, closely tracks the water delivery from the water pump S332B (**Figure 3.5**), high head water created in the detention ponds may have also caused water to flow westward, as the difference in water level between CR2 and CR3, and CR3 and A13, has increased in recent years (**Figure 3.6**), indicating a steepening of the water table gradient from the canals toward the west.

Sub-population F

In general, sparrow habitat in sub-population F has become wetter in the last 5 years than it was in the first half of the decade (**Figure 3.1**). At the census sites, which were surveyed once during 2003-2005 and re-surveyed during 2006-2009), an increasing trend in mean vegetation-inferred hydroperiod was observed (**Figure 3.7**), suggesting a change in vegetation composition indicative of increasing wetness. Median vegetation hydroperiod ranged between 126 days in 2003 and 176 days in 2007. The wetting trend, however, was not consistent throughout the sub-population. While vegetation change in the large area of the sub-population followed the general trend, change at a few sites in the northern portion of the sub-population suggested a drying trend **(Figure 3.8**).

Vegetation change in sub-population F tracked the changes in hydrologic regime that resulted from the operations of recently constructed water pumps and retention ponds. Two pumping stations, S332B and S332C deliver water from L31N to the detention areas constructed along the eastern edge of the subpopulation. To prevent the drainage of water from the Rocky Glades into the canal, water level in the detention areas is maintained at a level higher than in the adjoining prairies. However, as the result of high water level, water seeps toward the west and hydrates the glades. The strong influence of these detention ponds on hydrology, and therefore on vegetation was apparent in the spatial differences in the vegetation change pattern between the two groups of sites. Sites immediately west of the detention ponds showed a wetting trend, while locations northwest of the ponds showed a drying trend (**Figure 3.8**).

The effects of management-induced hydrologic changes on vegetation also depended on the scale of topographic variation. In the Rocky Glades, where surface elevation varies sharply within short distances, vegetation change pattern in response to hydrologic change observed at the broader scale was perhaps influenced by small scale topographic variation. For instance, vegetation change along Transect F, where sampling was at 100 m intervals, showed mixed results (**Figure 3.9**). At 70% of the plots on the transect, vegetation inferred hydroperiod increased, while at a mixture of unburned and burned sites located in the western portion of the transect, at greater distance from the detention ponds, vegetation-inferred hydroperiod was lower in 2010 than in 2004.

Vegetation change in burned sites

In 2008, 60% of the vegetation monitoring sites in Sub-population F, and four sites in sub-population E burned in the human-caused Mustang Corner fire. Additionally, three sites in the central portion of sub-population E also burned in 2008 in a lightening-ignited fire. Trajectory analysis revealed a subsequent shift in vegetation composition at many of these burned sites toward wetter or drier types, depending on their locations in the landscape (**Figure 3.10**). In general, post-fire vegetation composition follow the normal trajectory of recovery and return to vegetation similar to pre-fire condition within four to five years. However, if the sites are flooded immediately after the fire, vegetation may take a different trajectory (Sah et al. 2010; *also see section # 4*). Although sites burned in 2008 were not immediately flooded, many of them were affected by management-induced hydrologic changes. For instance, in sub-population F, the burned census sites that showed significant change in vegetation towards drier type were all located northwest of the detention ponds, and thus were perhaps not affected by overflow or seepage from the ponds. In contrast, in the southern portion of sub-population F and northeastern corner of sub-population E, vegetation at both unburned and burned sites showed a shift in species composition towards a more hydrophytic vegetation assemblage.

3.4 Conclusions and Management Implications

With the beginning 21st century, a shift in water management efforts in the Everglades from point delivery to surface flow aimed at hydrating the Rocky Glades landscape has had visible and sometimes positive effects on marl prairie vegetation and on populations of the Cape Sable seaside sparrow. For instance, in sub-populations C and F, the reduction in the number of sparrows between 1981 and early 1990s was attributed to either change in habitat due to increased hydroperiod or fire frequency. The current water management activities, however, have reversed the trend of vegetation change in the eastern portion of sub-population C or initiated the vegetation shift in other areas towards a wetter vegetation type, which is likely to reduce fire frequency, especially in northwestern and southern parts of sub-populations C and F, respectively. In general, because of its central location, the sub-population E is less impacted by hydrologic management activities that occur along the boundaries of ENP. However, even habitat within this sub-population seems to have been affected by ongoing hydrologic modifications.

If the recent trend in sub-population E, i.e. increase in sparrow population habitat in 2010 is considered as evidence of habitat improvement, a similar response is likely to occur as well in both sub-populations C and F. However, not all the effects of pumping of water from the canal directly into the marl prairies or indirectly through detention ponds are positive. For instance, in other parts of the Everglades, researchers have demonstrated that water input from the canals has altered soil phosphorus in the adjacent marsh, resulting in vegetation change in the impacted areas (Doren et al. 1997; Childers et al. 2003). In the eastern Everglades, phosphorus content in periphyton was higher at marl prairies sites near the L31W canal and in detention basins near the L31N than in adjacent marl prairie sites to the west, reflecting the long-term exposure of the canal-side sites to seepage (Gaiser 2006; Gaiser et al. 2008). Periphyton is known to show a quick response to increased phosphorus concentration in surface water, and is a precursor of P-enrichment in the soil (Gaiser et al. 2004). Therefore, a time lag in P-enrichment in the soil is inevitable. Therefore, it may be important to monitor the phosphorus loading in the water directly entering the Park from the canal, and its impact on prairie vegetation.

4. Fire and flooding interactions: vegetation dynamics trajectories

4.1 Background

Fire and flooding are integral parts of ecosystem processes in many wetlands, including floodplains, coastal prairies, and seasonally-flooded grasslands (McKee and Baldwin 1999; Lockwood et al. 2003). In seasonally flooded grasslands, particularly in those regions where the probability of wildfire is highest at the onset of the rainy season, there is a likelihood that a wildfire will be closely followed by flooding. The chances of such events are high in South Florida, where wildfires caused by lightning are frequent in the rainy season (Wade et al. 1980; Snyder 1991). Moreover, in the rainy season, there is always a possibility of torrential rains associated with tropical storms or hurricanes.

Vegetation recovery after a single burn event in many wetlands ends in a return to pre-burn community composition and structure within 3-4 years (Pahl et al. 2003; La Puma et al. 2007). However, sequential disturbances, such as fire followed by flooding, may result in changes in wetland community character by removing dominant species and facilitating the growth of opportunistic species (Zedler and Krecher 2004). We examined community level responses to interactions of fire and flooding in Southern Everglades marl prairies, where three major fires burned 1,348 ha of marl prairie within the CSSS habitat in 2005. In May, the “Aerojet” fire burned 76.7 ha areas in sub-population D, outside Everglades National Park, and in August the “Keyhole” and “Sisal” fires burned 611 and 660 ha areas, respectively, in sub-population B (**Figure 4.1**). The area burned in May (May_burn) remained unflooded for >1 month after fire, and for two months thereafter experienced a gradual increase in water level, while the area burned in August (Aug_burn) were flooded by more than a foot (30 cm) of water by Hurricane Katrina within 7-15 days of fire. Prior to the 2005 fires, these sites were all sampled once during the previous three years. After the sites burned in 2005, they were sampled annually, between March and May, for five consecutive years. At each sampling site, vegetation was sampled following the methods described in Ross et al. (2003).

4.2 Analytical methods

Trajectory Analysis:

We used analysis of similarity (ANOSIM) to examine the differences in vegetation composition between pre-burn and post-burn years. This analysis of vegetation change was supplemented at individual sites by trajectory analysis (Minchin et al. 2005; *also see sub-section* 2.2.2), which allowed the examination of change in community composition along a pre-defined target vector representing time since last fire (TSLF) and hydrologic influence in ordination space. To define the TSLF and hydrology (mean annual days per year flooded) vectors, the census and transect sites sampled between 2003 and 2005 in five eastern sub-populations (B-F) were included in the NMS ordination. The sites sampled in 2003 to 2005 represented a range of 0 to 25 years along TSLF gradient (Ross et al. 2006). The hydrology vector was derived by calculating plot level hydroperiod, using water level data obtained for the nearest stage recorders and mean plot elevation, obtained by surveying from the nearest vertical control benchmark to each subplot.

Because the purpose of the trajectory analysis was to examine recovery in vegetation composition since fire, Year 1 after fire (2006) was considered as the base year, and the shift in position of samples along TSLF vector in the ordination space was assessed.

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) from pre-fire (2005), and five post-fire (2006 - 2010) years. Images from all years except 2006 and 2008 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. The spatial resolution of TM images was 30 m x 30 m. The images had been geo-referenced and atmospherically corrected for interferences from atmospheric reflectance by the Remote Sensing and GIS Center of Florida International University. We used the corrected images and the computer program ArcGIS 9.3 to compute NDVI as: NDVI = (NIR-RED)/(NIR+RED). 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. Likewise, the NDVI difference between the first post-fire year and subsequent years was used to quantify vegetation recovery.

4.3 Results

Vegetation composition in both May_burn and Aug_burn 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_burn - $R = 0.511$, $p = 0.002$; Aug_burn - $R = 0.732$, $p = 0.001$), and mean total plant cover were only 43.0% and 33.8% of the initial cover in the May_burn and Aug_burn groups, respectively (**Figure 4.2**). Slow recovery of vegetation cover at these sites probably resulted from post-fire hydrologic conditions, as the majority of sites burned in 2005 experienced various levels of flooding after fire.

Fire followed by flooding often results in changes in community characteristics by removing dominant species and facilitating the growth of opportunistic species. 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 4.3**). At Aug_burn sites, relative cover of four dominant species, i.e., sawgrass (*Cladium mariscus* ssp. *jamaicense*), bluestem (*Schizachyrium rhizomatum*), muhly grass (*Muhlenbergia capillaris* var. *filipes*) and black-top sedge (*Schoenus nigricans*) significantly decreased immediately after fire followed by flooding, and remained much lower than before the fire even five years later. In contrast, relative cover of several minor species, such as spadeleaf (*Centella asiatica*), southern beakrush (*Rhynchospora microcarpa*), gulfdune paspalum (*Paspalum monostachyum*) and bluejoint panicgrass (*Panicum tenerum*) was higher in the fifth year after fire than in pre-burn samples. Interestingly, at May_burn 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. mariscus* ssp. *jamaicense*) was observed. After five years of 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 cellolosa*), and southern beakrush (*R. microcarpa*) were significantly higher five years after fire than pre-fire.

In both May_burn and Aug_burn groups, the change in relative cover of dominant species and the growth of opportunistic species in post fire years resulted in variation in mean species richness in post fire years. Mean species richness was significantly lower in May_burn than in Aug_burn sites prior to the 2005 fires (Repeated Measures ANOVA, Bonferrroni test: $p < 0.01$), possibly due to differences in their hydrologic conditions. May_burn sites were relatively wet sites, and in pre-fire year they had 12 species/plot, less than half of the number of species (27 species/plot) present at Aug_burn sites. While the richness in both the groups declined in the first year after fire, the mean number of species in the May-burned plots recovered to pre-fire levels in the following year and remained at the same level through the fifth year after fire (**Figure 4.4**). In contrast, the number of species present in the Aug-burned plots was low in the first two post-fire years, and recovered to pre-fire level in the 3rd year.

The post-fire vegetation change pattern was also analyzed using trajectory analysis. The results revealed that only a few sites in Aug_burn groups showed a significant (p-value <0.1) shift along the time-since-last-fire (TSLF) vector, expressed as the amount (Δ) and rate (slope) of change in vegetation composition (**Table 4.1**). Contrary to our expectation, none of the May_burn sites showed a significant rate of change in vegetation along the TSLF vector. However, five years after the fires, both mean degree (delta) and rate (slope) of change in vegetation composition along the TSLF vector were higher in the May_burn sites (Δ = 0.14, slope = 0.037) than in Aug_burn sites (Δ = 0.06, slope = 0.0.029) sites. In both groups, there was high within group variation (Coefficient of variance: CV>1.0) in both degree (delta) and rate of vegetation change (slope) along the TSLF vector. In the pre-burn analysis, samples were positioned in ordination space near the high end of the TSLF vector (**Figure 4.5**), so post-burn sites that approached the pre-burn condition were likely to show a significant shift along the TSLF vector. In contrast to most of sites burned in 2005, the composition of two reference sites burned in 2003 and sampled annually for four years resembled their pre-burn state by the 4th year after fire.

The analysis of NDVI change revealed that vegetation recovery pattern varied over the 5-year post-fire period in both the May_burn and Aug_burn fires (**Figure 4.6)**. While the mean NDVI in Year-1 after fire was relatively low in both fires (0.219 and 0.200 in May_burn and Aug_burn, respectively), the decrease in NDVI from pre-fire to 1st post-fire year was significantly greater in Aug_burn (47.7%) than May_burn sites (38.7%), suggesting that damage to vegetation from fire was more severe in the area that was burned and immediately flooded (**Figure 4.7**). In subsequent years, vegetation recovery was faster in May_burn sites, reaching the pre-burn level of NDVI within 2 post-fire years. In contrast, mean NDVI in Aug_burn sites increased linearly and reached the pre-fire level in four years (**Figure 4.8**). However, some parts of the May_burn area had very low NDVI in 2010, five years after fire, suggesting slow recovery in those areas.

Table 4.1: Delta (amount of change in target direction) and slope (rate of change in the target direction in half-changes per year) calculated for 2 sites burned in 2003 and 21 sites burned in 2005. The 2003 and 2005 burned sites were monitored for 4 and 5 years after fire, respectively. Time since last fire (TSLF) vector in the non-metric multidimensional scaling (NMS) ordination was the target direction. The base year for change in vegetation was the first year after fire. Statistical significance ($p \leq 0.1$) of delta and slope was tested using Monte Carlo's simulations with 10,000 permutations.

Burn Group	Site	Delta	P-value	Slope	P-value
Ref-2003	B-01-01	0.686	0.000	0.207	**0.000**
	B-01-04	0.316	0.147	0.116	0.109
Aug_burn	B-05-06	0.141	0.196	0.020	0.281
	B-05-07	0.051	0.384	0.040	0.159
	B-05-08	0.113	0.172	0.044	**0.048**
	B-06-05	0.061	0.335	-0.008	0.580
	B-06-07	0.087	0.297	0.031	0.155
	B-06-08	0.226	0.041	0.043	**0.063**
	B-10-03	0.117	0.274	0.065	**0.083**
	B-10-05	0.130	0.154	0.031	0.141
	B-10-09	-0.301	0.852	-0.016	0.703
	B-11-03	-0.060	0.623	0.016	0.362
	B-11-04	-0.157	0.832	-0.001	0.511
	B-11-05	0.212	0.103	0.046	0.107
	B-13-10	0.188	0.118	0.067	**0.037**
May_burn	D-01-10	-0.001	0.501	-0.018	0.621
	TD-1900	0.060	0.416	0.019	0.373
	TD-2000	0.204	0.274	0.062	0.200
	TD-2100	0.226	0.301	0.095	0.174
	TD-2200	0.118	0.368	0.044	0.265
	TD-2300	0.009	0.476	-0.007	0.519
	TD-2400	0.370	0.187	0.068	0.235
	TD-2500	0.040	0.408	0.011	0.416

4.4 Discussion and Conclusions

The observed pattern in marl prairies burned in 2005 differs from results reported for other fires. Several authors reported that vegetation after a single burn in seasonally-flooded wetlands returns to pre-burn conditions within 3-5 years of fire (Werner 1975; Pahl et al. 2003; La Puma et al. 2007). A similar pattern of vegetation recovery was also reported at two sites burned in spring 2003, and sampled annually for four years thereafter (Sah et al. 2008, 2009). Inconsistency between the present study and earlier research probably results from differences in post-fire hydrologic conditions, as the majority of sites burned in 2005 were flooded after fire.

Most graminoids normally re-sprout 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. mariscus* ssp. *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). Persistence of the relatively low cover of dominant species in post-fire years at the Aug_burn sites has also facilitated the growth of other species. At May_burn 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 per-burn samples may also be a function of differences in dead materials. Prior to the 2005 fires, the sites had not burned for 14 years, and retained a large component of dead sawgrass.

Assessment of vegetation recovery dynamics in post-fire years was well illustrated by both trajectory and NDVI change analyses. In the trajectory analysis, none of the May_burn sites showed a significant shift in species composition toward pre-burn conditions, and only a few sites in Aug_burn groups showed a significant shift in vegetation composition with time-since-last-fire (TSLF). Our expectation was that the mean rate of change, i.e. vegetation recovery, would be faster in May_Burn than in the Aug_Burn group. Visual analysis of trajectories revealed that the trajectory of several May_burn 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_burn 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_burn sites, vegetation recovery towards pre-burn composition was slow, and vegetation composition even five years after fire was significantly different from pre-burn samples at most of these sites.

Post-fire NDVI change pattern that represented vegetation recovery trajectories differed between the two burned areas, suggesting in differences in inherent resilience of pre-fire vegetation types, and in post-fire environmental conditions, particularly hydrology. Pre-fire vegetation in the May_burn sites was dominated by sawgrass, which was indicative of relatively long hydroperiod, whereas vegetation in the Aug_burn sites were typical of short hydroperiod wet prairies. In general, fire impacts on vegetation tend to be less severe at marsh sites than the prairie sites, due to high plant and soil moisture content. Post-fire flooding had killed most of vegetation at the Aug_burn sites, while the vegetation at the May_burn 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_burn and Aug_burn contrasted somewhat with field based estimates of plant cover. NDVI is affected by several factors, including total plant cover, biomass, plant and soil moisture, and leaf area index. Since NDVI is sensitive to cholorophyll 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_burn and Aug_burn 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 vegetation mosaics within the landscape, particularly when its effects on vegetation structure and composition are mediated through other disturbances, such as changes in hydrologic regime. While the interval between fire and post-fire hydrologic events is important in shaping the response of vegetation to the synergetic effects of these two disturbances, 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. 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.

References:

Abdel Malak, D. and Pausas, J. G. 2006. Fire regime and post-fire Normalized Difference Vegetation Index changes in the eastern Iberian Peninsula (Mediterranean basin). *International Journal of Wildland Fire* **15**: 407-413.

Armentano, T. V., Sah, J. P., Ross, M. S., Jones, D. T., Cooley, H. C. and Smith, C, S. 2006. Rapid responses of vegetation to hydrological changes in Taylor Slough, Everglades National Park, Florida, USA. *Hydrobiologia* **569**: 293-309.

Basier, R. L., Boulton, R. L. and Lockwood, J. L. 2008. Influence of water depth on nest success of the endangered Cape Sable seaside sparrow in the Florida Everglades. *Animal Conservation* **11**: 190-197.

Ball, J. P. 1990. Influence of subsequent flooding depth on cattail control by burning and mowing. *Journal of Aquatic Plant Management* **28**: 32-36.

Breen, C. M., Rogers, K. H. and Ashton, P. J. 1988. Vegetation processes in swamps and flooded plains. In: J. J. Symoens (Ed.). *Vegetation of Inland Waters*. pp. 223-248. Kluwer Academic Publishers, Dordrecht.

Chapin III, F. S., Robards, M. D., Huntington, H. P., Johnstone, J. F., Trainor, S. F., Kofinas, G. P., Ruess, R. W., Fresco, N., Natcher, D. C. and Naylor. R. L. 2006. Directional Changes in Ecological Communities and Social Ecological Systems: A Framework for Prediction Based on Alaskan Examples. *American Naturalist*, **168** (Supplement): S36–S49.

Childers, D. L., Doren, R. F., Jones, R., Noe, G. B. and Scinto, L. J. 2003. Decadal change in vegetation and soil phosphorus pattern across the Everglades landscape. *Journal of Environmental Quality* **32**: 344-362.

Clarke, K. R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**: 117-143.

Collins, S. L. 2000. Disturbance frequency and community stability in native tallgrass prairie. *The American Naturalist* **155** (3): 311-325.

Curtis, J. T. 1959. *The vegetation of Wisconsin: an ordination of plant communities*. University of Wisconsin Press, Madison, WI.

Doren, R. B., Armentano, T. V., Whiteaker, L. D. and Jones, R. D. 1997. Marsh vegetation patterns and soil phosphorous gradients in the Everglades ecosystem. *Aquatic Botany* **56**: 145–163.

Forthman, C. A. 1973. The effects of prescribed burning on sawgrass, *Cladium jamaicense* Crantz, in south Florida. M.S. Thesis, University of Miami, FL, 83 pp.

Gaiser, E., Scinto, L., Richards, J., Jayachandran, K., Childers, D., Trexler, J. and Jones, R. 2004. Phosphorus in periphyton mats provides the best metric for detecting low-level P enrichment in an oligotrophic wetland. *Water Research* **38** (3): 507-516.

Gaiser, E. 2006. Characterization of periphyton response to hydroperiod in marl prairie wetlands of the Everglades. Final Comprehensive Report 2006. Submitted to Everglades National Park, Homestead, FL.

Gaiser, E., Price, R., Scinto, L. and Trexler, J. 2008. Phosphorus retention and sub-surface movement through the detention basins on the eastern boundary of Everglades National Park. Year 3 Final Report to Everglades National Park November 7, 2008.

Herndon, A., Gunderson, L. and Stenberg, J. 1991. Sawgrass (*Cladium jamaicense*) survival in a regime of fire and flooding. *Wetlands* **11**:17-27.

Hope, A., Tague, C. and Clark, R. 2007. Characterizing post-fire vegetation recovery of California chaparral using TM/ETM+ time-series data. *International Journal of Remote Sensing* **28**: 1339-1354.

Junk, W. J. and Piedade, M. T. F. 1997. Plant life in the floodplain with special reference to herbaceous plants. *In The Central Amazon Floodplain: Ecology of a Pulsing System*, ed. by W. J. Junk, pp. 147–185. Springer, Berlin.

Junk, W. J., Bayley P. B. and Sparks, R. E. 1989. The flood pulse concept in river floodplain systems. In *Proceedings of the International Large River Symposium(LARS)*, ed. by D. P. Dodge, pp. 110–127. Canadian Special Publication of Fisheries and Aquatic Sciences, Ottawa, Canada.

Kirkman, L. K. and Sharitz, R. R. 1994. Vegetation disturbance and maintenance of diversity in intermittently flooded Carolina Bays in South Carolina. *Ecological Applications* **4**: 177-188.

Kotun, K. Sonenshein, R., and DiFrenna, V. 2009. Analysis of flow across Tamiami Trail: An historical perspective. South Florida Natural Resources Center, Everglades National Park, Homestead, Florida. Technical Report (Unpublished Report).

La Puma, D. A., Lockwood, J. L. and Davis, M. J. 2007. Endangered species management requires a new look at the benefit of fire: The Cape Sable seaside sparrow in the Everglades ecosystem. *Biological Conservation* **136**: 398-407.

Light, S. S. and Dineen, W. 1994. History of water management in the Everglades. In S. Davis & J. Ogden (Eds.), *The Everglades: The ecosystem and its restoration*. pp. 47-84. Delray Beach, FL: St. Lucie Press.

Lockwood, J. L., Ross, M. S. and Sah, J. P. 2003. Smoke on the water: the interplay of fire and water flow on Everglades restoration. *Frontiers in Ecology and Environment* **1** (9): 462-468.

McCullagh, P. and Nelder, J. A. 1989. Generalized Linear Models. Chapman and Hall: London.

McKee, K. L. and Baldwin, A. H. 1999. Disturbance regimes in North American wetlands, p. 331–363.*In* L. R. Walker (ed.), Ecosystems of Disturbed Ground. Ecosystems of the World 16. Elsevier, Amsterdam, The Netherlands.

Middleton, B. A. 1999. Wetland Restoration, Flood Pulsing and Disturbance Dynamics. John Wiley & Sons, New York.

Minchin P. 1998. DECODA: Database for Ecological Community Data. Anutech Pty. Ltd, Canberra, Australia.

Minchin, P. R., Folk, M. and Gordon, D. 2005. Trajectory Analysis: a New Tool for the Assessment of Success in Community Restoration. Meeting Abstract, Ecological Society of America 90th annual meeting, Montreal, Quebec, August 7-12, 2005.

Niering, W. 1994. Wetland vegetation change: A dynamic process. *Wetland Journal* **6:** 6-15.

Nott, M. P., Bass, Jr., O. L., Fleming, D. M., Killeffer, S. E., Fraley, N., Manne, L., Curnutt, J. L., Brooks, T. M., Powell, R. and Pimm, S. L. 1998. Water levels, rapid vegetational changes, and the endangered Cape Sable seaside sparrow. *Animal Conservation* **1**:23-32.

Pahl, J. W., Mendelssohn, I. A., Henry, C. B. and Hess, T. J. 2003. Recovery trajectories after in situ burning of an oiled wetland in coastal Louisiana, USA. *Environmental Management* **31** (2): 236-251.

Pimm, S. L., Lockwood, J. L., Jenkins, C. N., Curnutt, J. L., Nott, P., Powell, R. D. and Bass, O. L. Jr. 2002. Sparrow in the Grass: A report on the first ten years of research on the Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*). Report to Everglades National Park, Homestead, FL.

Ponzio, K. J., Miller, S. J. and Lee, M. A. 2004. Long-term effects of prescribed fire on *Cladium jamaicense* crantz and *Typha domingensis* pers. densities. *Wetlands Ecology and Management* **12**: 123-133.

Rose, P. W., Flora, M. D. and Rosendahl, P. C. 1981. Hydrologic Impacts of L-31W on Taylor Slough, Everglades National Park. Report T-612. South Florida Research Center, Everglades National Park, Homestead, Florida, USA.

Ross. M. S., Sah, J. P., Ruiz, P. L., Jones, D. T., Cooley, H. C., Travieso, R., Snyder, J. R. and Schaeffer, C. 2003. Effect of Hydrology Restoration on the Habitat of the Cape Sable Seaside Sparrow. Report to Everglades National Park. June 30, 2003.

Ross, M. S., Sah, J. P., Snyder, J. R., Ruiz, P. L., Jones, D. T., Cooley, H., Travieso, R., Tobias, F. and Hagyari, D. 2006. Effect of hydrological restoration on the habitat of the Cape Sable seaside sparrow. 2004-2005. Year-3. Final Report submitted to Everglades National Park, Homestead, FL and U. S. Army Corps of Engineers, Jacksonville, FL. March 2006. 50 pp.

Sah, J. P., Ross, M. S., Snyder, J. R., Ruiz, P. L, Stoffella, S., Kline, M., Shamblin, B., Hanan, E., Ogurcak, D., Gomez, D. and Barrios. B. 2008. Effect of hydrological restoration on the habitat of the Cape Sable seaside sparrow. Annual Report of 2006-2007. A Report submitted to Everglades National Park, Homestead, FL. 47 pp.

Sah, J. P., Ross, M. S., Snyder, J. R., Ruiz, P. L, Stoffella, S., Kline, M., Shamblin, B., Hanan, E., Lopez, L. and Hilton, T. J. 2009. Effect of hydrological restoration on the habitat of the Cape Sable seaside sparrow. Annual Report - FY 2008. A report submitted to USACOE, Jacksonville, FL. Cooperative Agreement # W912EP-08-C-0018. 52 pp.

Sah, J. P., Ross, M. S., Snyder, J. R., Ruiz, P. L., Stoffella, S., Colbert, N., Hanan, E., Lopez, L., and Camp, M. 2010. Cape Sable seaside sparrow habitat – Vegetation Monitoring. FY 2009 Final Report submitted to U. S. Army Corps of Engineers, Jacksonville, FL. Cooperative Agreement t # W912EP-09-C-0024. Jan. 2010. 50 pp.

Schessl, M. 1999. Floristic composition and structure of floodplain vegetation in the northern Pantanal of Mato Grosso, Brazil. 1999. *Phyton* **39** (2): 303-336.

Snyder, J. R., 1991. Fire regimes in subtropical South Florida. In: Proceedings of the Tall Timbers Fire Ecology Conference no. 17, High Intensity Fire in Wildlands: Management Challenges and Options. May 18–21, 1989. Tall Timbers Research Station, Tallahassee, FL, USA, pp. 303–319.

Snyder, J. R. and Schaeffer, C. 2004. Seasonal fire effects on muhly grass (*Muhlenbergia capillaris* var. *filipes*). Final Report # IAA Number F5120010007 submitted to Big Cypress National Preserve, Ochopee 34141, FL.

U.S. Army Corps of Engineers (USACOE). 1999. Central and Southern Florida Project Comprehensive Review Study: Final Integrated Feasibility Report and Programmatic Environmental Impact Statement. Jacksonville District, Jacksonville, Florida.

U.S. Army Corps of Engineers (USACOE). 2006. Interim Operational Plan for Protection of the Cape Sable seaside sparrow, Central and Southern Florida Project for Flood Control and Other Purposes, Final Supplemental Environmental Impact Statement, 2006. Jacksonville District, Jacksonville, Florida.

U.S. Fish and Wildlife Service (USFWS). 2007. Endangered and threatened wildlife and plants: Critical habitat revised designation for the Cape Sable seaside sparrow – Final Rule. *Federal Register* **72** (214): 32736-62766.

van der Valk, A. G. 2005. Water-level fluctuation in North American prairie wetlands. *Hydrobiologia* 539: 171-188.

van Leeuwen, W. J. D., and Huete, A. R. 1996. Effects of standing litter on the biophysical interpretation of plant canopies with spectral indices. *Remote Sensing of Environment* **55**: 123–138.

van Lent, T. A., Johnson, R. A. and Fennema, R. J. 1993. Water management in Taylor Slough and effects on Florida Bay. South Florida Natural Resources Center, Everglades National Park, Homestead, FL, USA. Technical Report 93-3.

Viedma, O., Melia, J., Segarra, D. and Garcia-Haro, J. 1997. Modeling Rates of Ecosystem Recovery after fires by using Landsat TM Data. *Remote Sensing of Environment* **61**: 383-398.

Wade, D., Ewel, J. and Hofstetter, R. 1980. Fire in south Florida ecosystems. US Forest Service, General Technical Report # SE-17, 125 pp.

Werner, H. 1975. The Biology of the Cape Sable seaside sparrow, In Report to US Fish and Wildlife Service. Everglades National Park, Homestead.

Whittaker, R. H. 1956. Vegetation of the Great Smoky Mountains. *Ecological Monographs* **26**: 1-80.

Whittaker, R. H. 1967. Gradient analysis of vegetation. *Biological Review* **42**: 207-264.

Zedler, J. B. and Callaway, J. C. 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories? *Restoration Ecology* **7**: 69-73.

Zedler, J. B. and Kercher, S. 2004. Causes and consequences of invasive plants in wetlands: opportunities, opportunists, outcomes. *Critical Reviews in Plant Sciences* **23**:431–452.

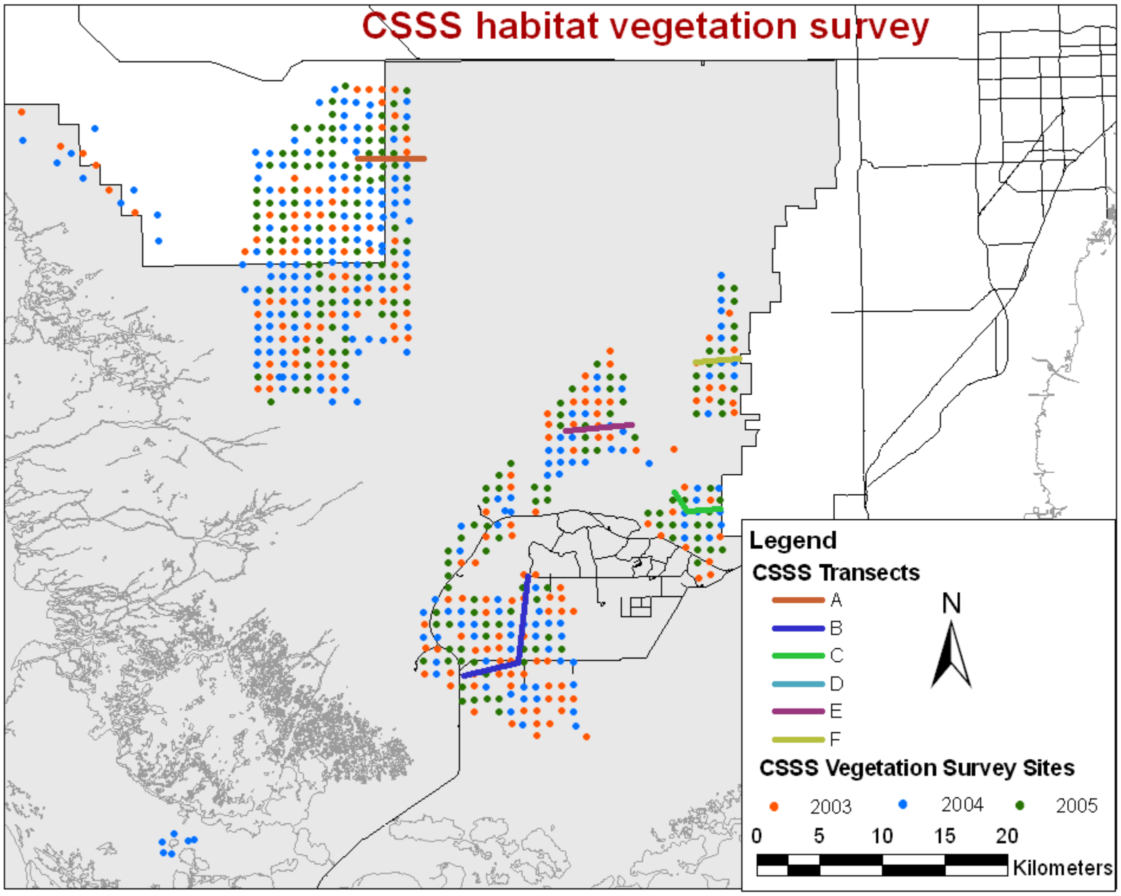

Figure 2.1: Location of vegetation sampling sites within the Cape Sable seaside sparrow (CSSS) habitat. The sites were initially sampled between 2003 and 2005, thereafter sub-sets of sites were re-sampled between 2006 and 2009.

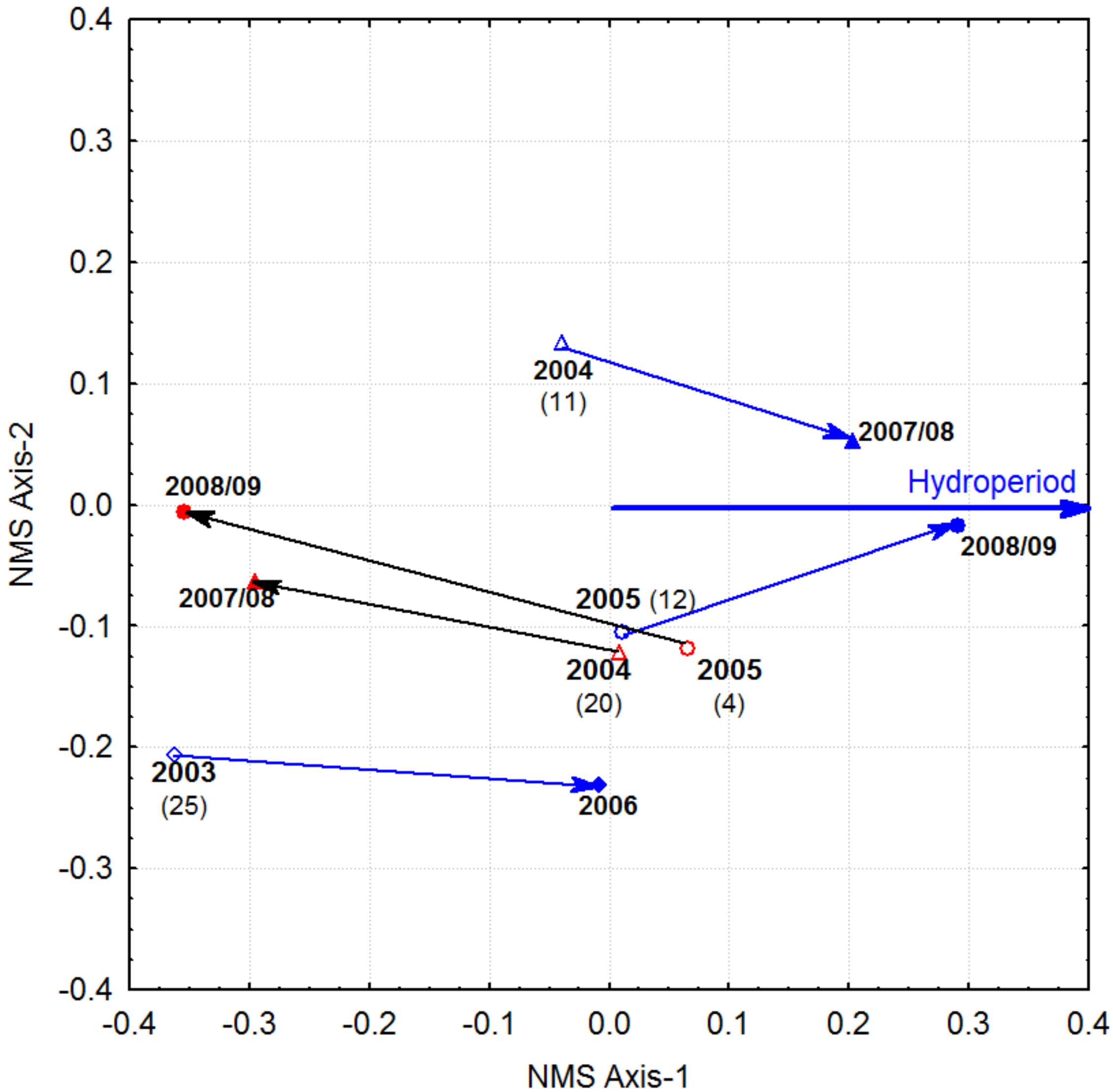

Figure 2.2: The non-metric multidimensional scaling (NMDS) ordination showing the trajectory of centroids of sub-population A census sites that were sampled twice over seven years (2003-2009). Sites are grouped by the year when they were first sampled and direction of their trajectory shift along hydroperiod vector. Beginning and end of trajectory represents the year of first (2003-2005) and second (2006-2009) sampling event, respectively.

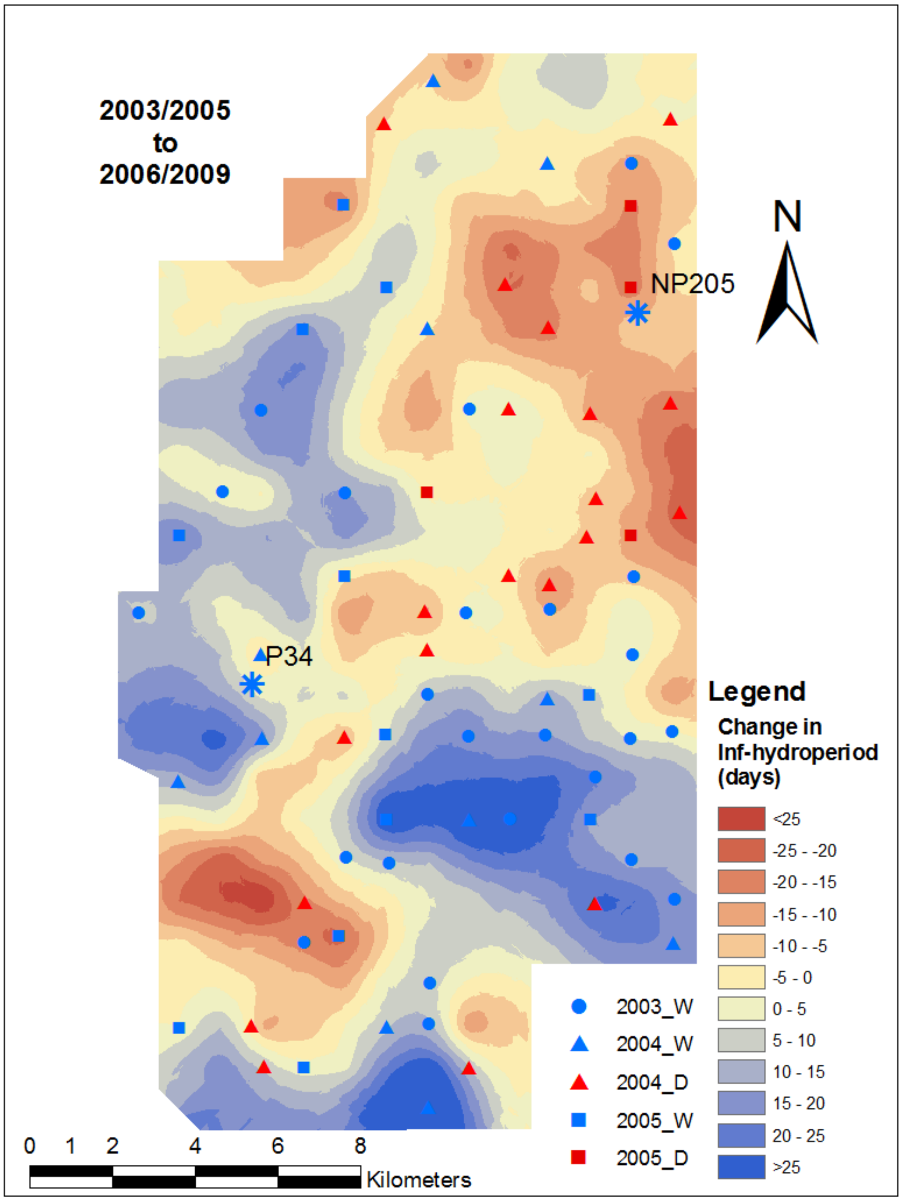

Figure 2.3: Change in vegetation-inferred hydroperiod in sub-population A between two sampling periods, 2003-2005 and 2006-2009. The sites that were sampled twice over seven years (2003-2009) and showed a significant trajectory shift along the hydrology gradient are overlaid. Sites are categorized by year when they were first sampled, and direction of shift in vegetation composition. W = towards indicator of longer hydroperiod, D = towards indicator of shorter hydroperiod.

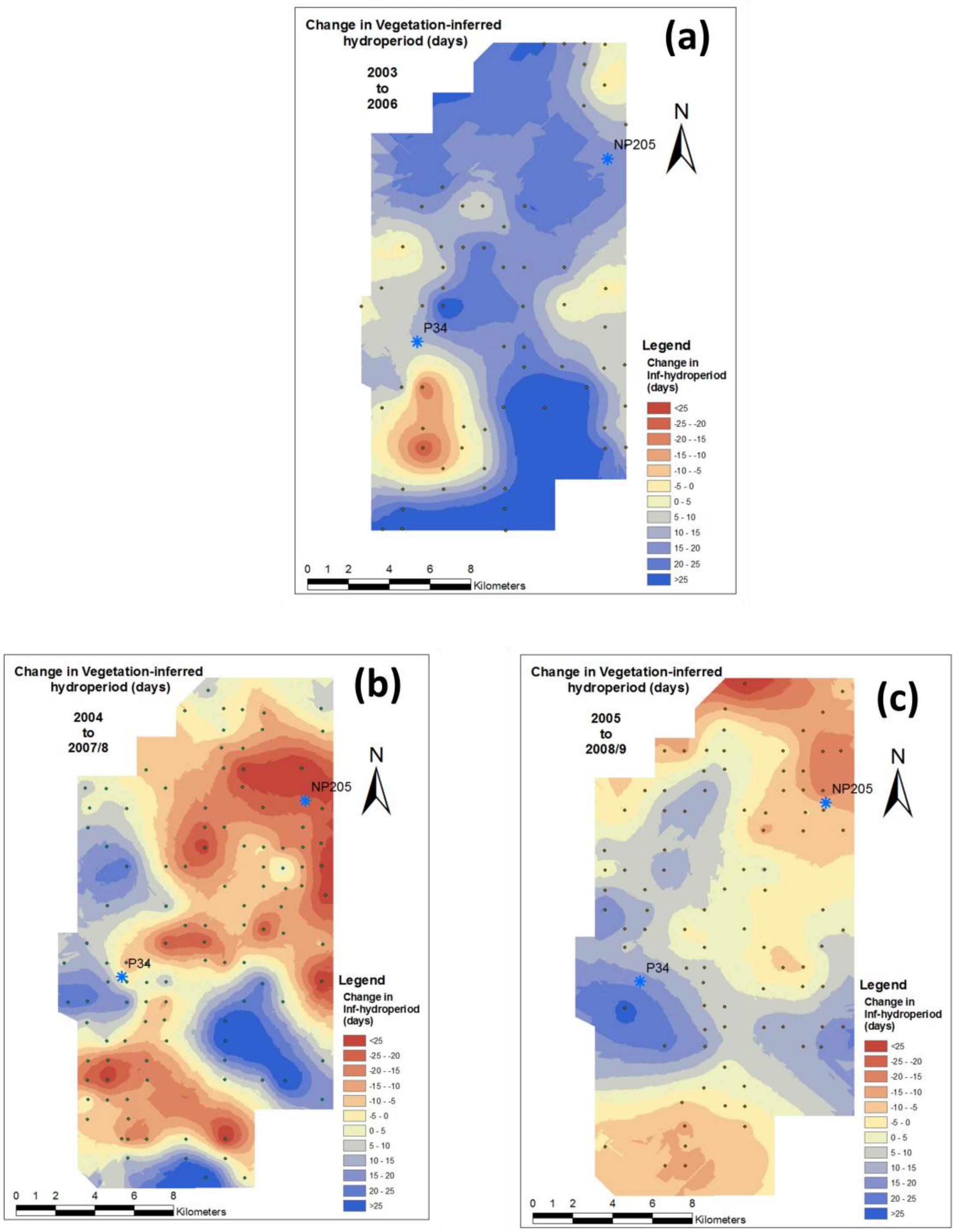

Figure 2.4: Change in vegetation-inferred hydroperiod in sub-population A between two sampling events, (a) 2003 and 2006, (b) 2004 and 2007 or 2008, and (c) 2005 and 2008 or 2009.

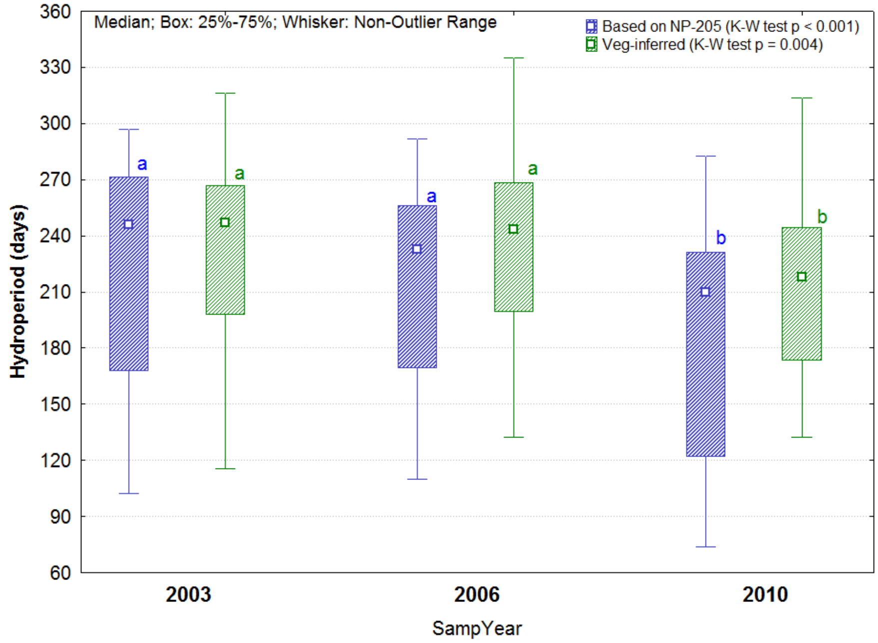

Figure 2.5: Box-plots showing the NP-205 based and vegetation-inferred hydroperiod at the sites on Transect A.

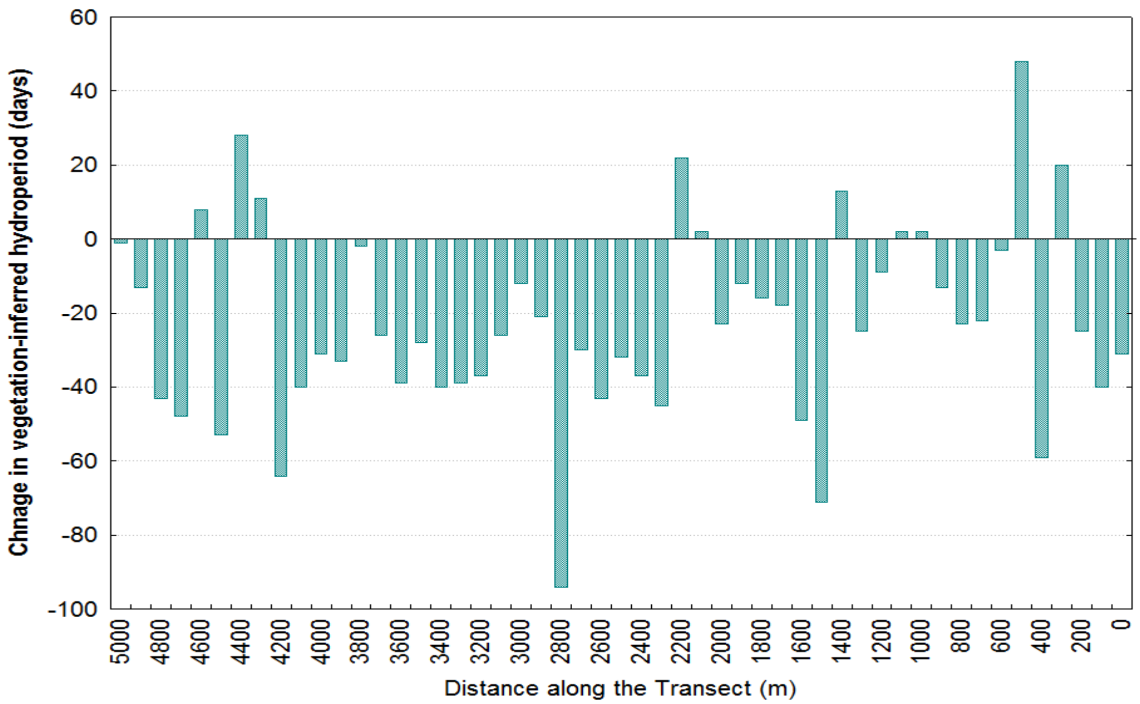

Figure 2.6: Change in vegetation inferred hydroperiod between 2003 and 2010 at the sites on Transect A.

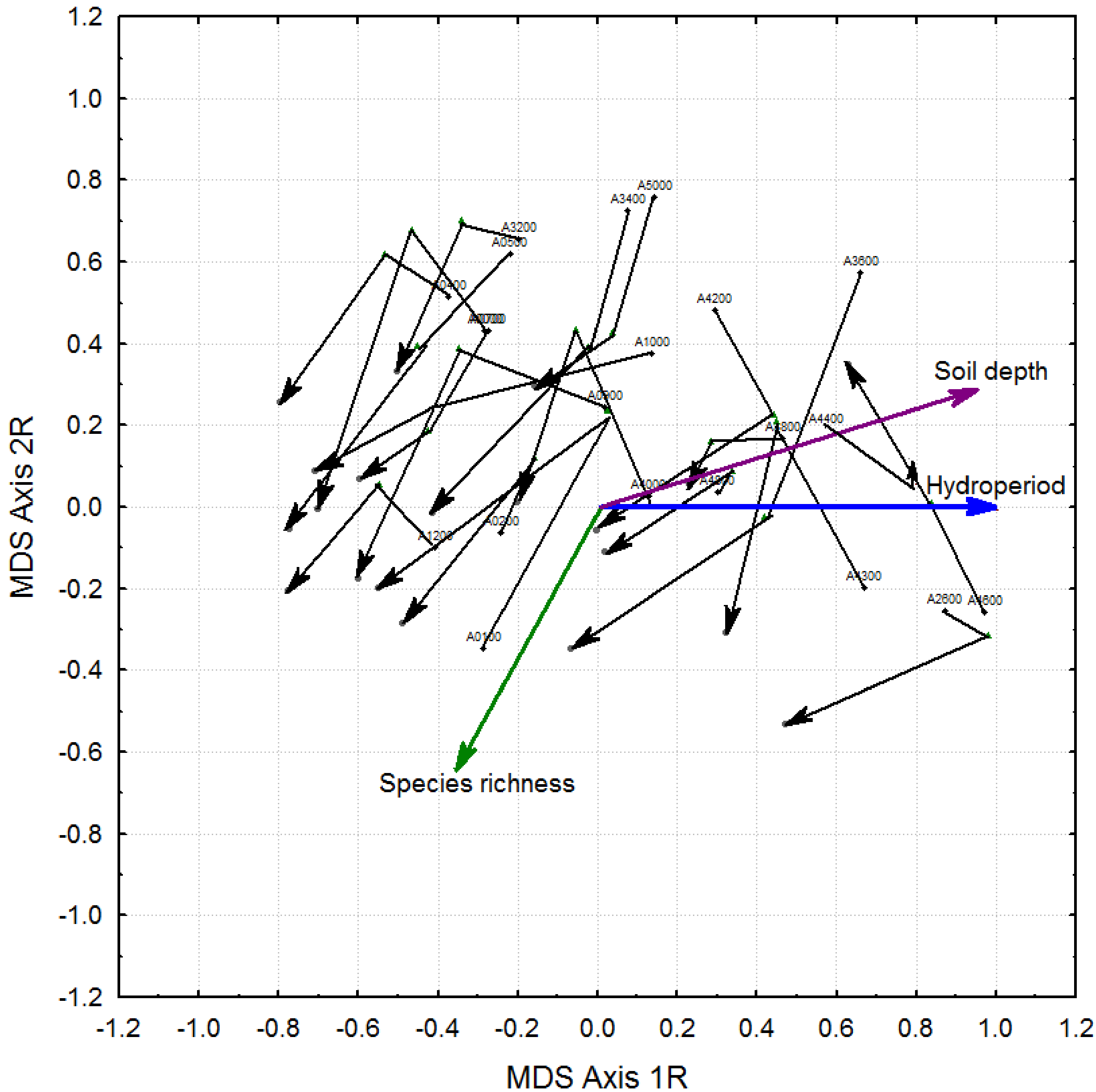

Figure 2.7: The non-metric multidimensional scaling (NMDS) ordination showing the trajectory of sites from CSSS vegetation Transect A sampled in 2003, 2006 and 2010. Only the sites which showed significant trajectory shift representing a in species composition along the hydrology gradient are shown. Ordination axes were rotated to perfectly align the hydroperiod vector with the first axis. Initial point on each site trajectory represents the 2003 sampling, and the end of arrows represents the 2010 sampling.

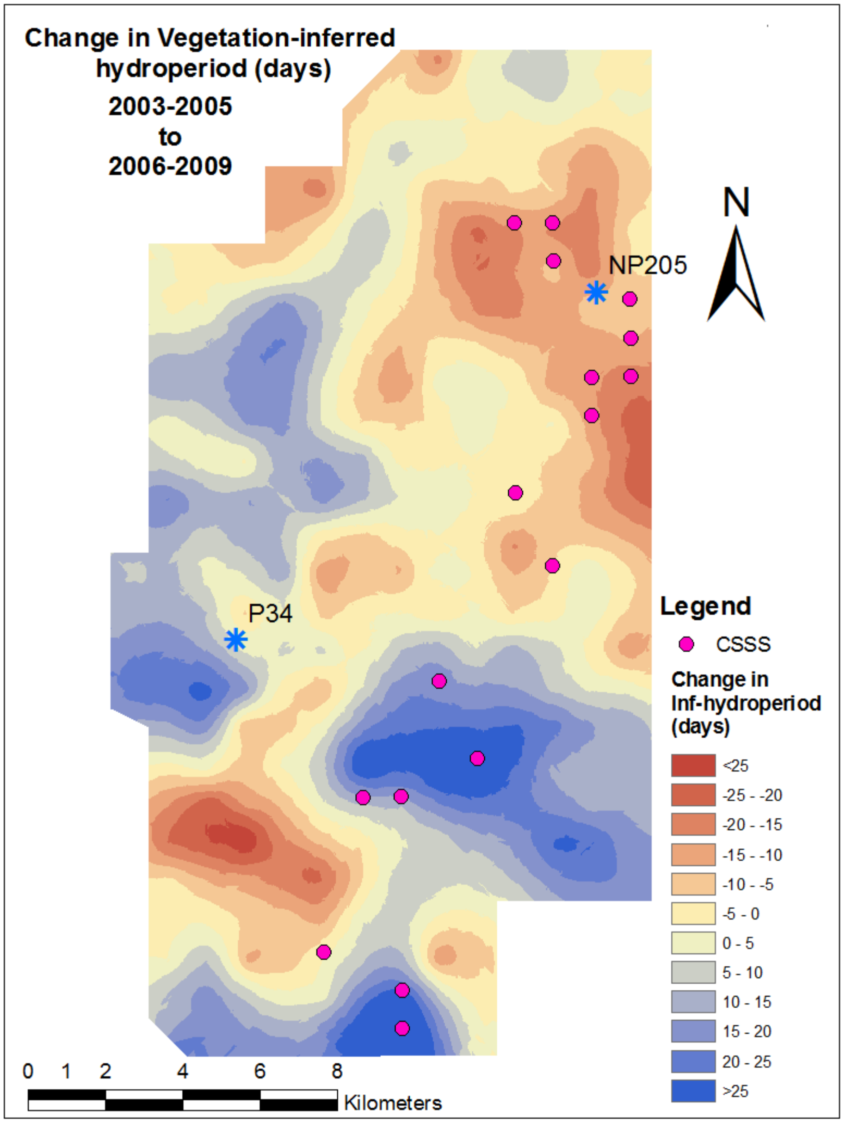

Figure 2.8: Change in vegetation-inferred hydroperiod in sub-population A between the two sampling events, 2003-2005 and 2006-2009, and the sites at which occurrence of Cape Sable seaside sparrow was recorded in one or more years between 2006 and 2010.

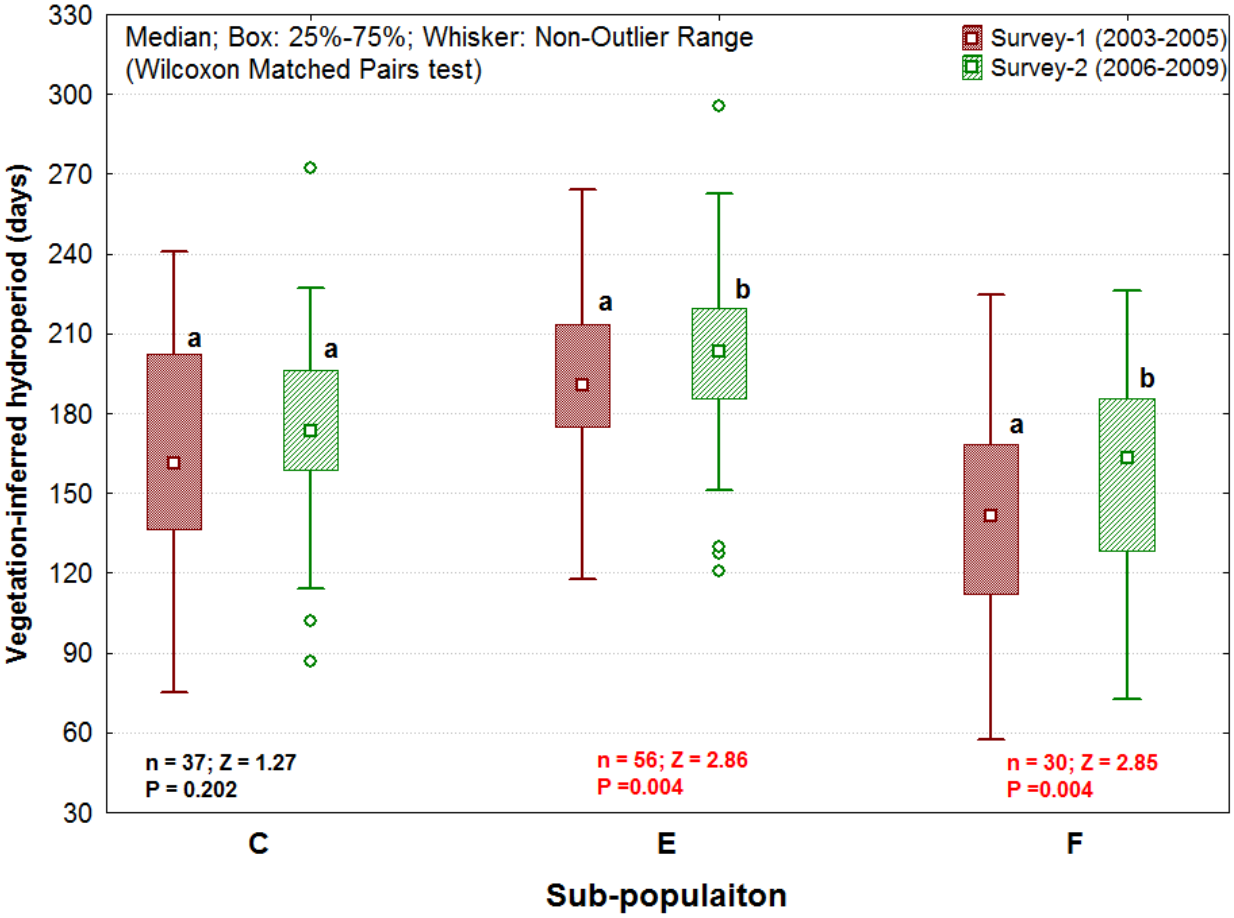

Figure 3.1: Box plots showing vegetation-inferred hydroperiod in the sub-populations C, E and F between the two sampling periods, 2003-2005 and 2006-2009.

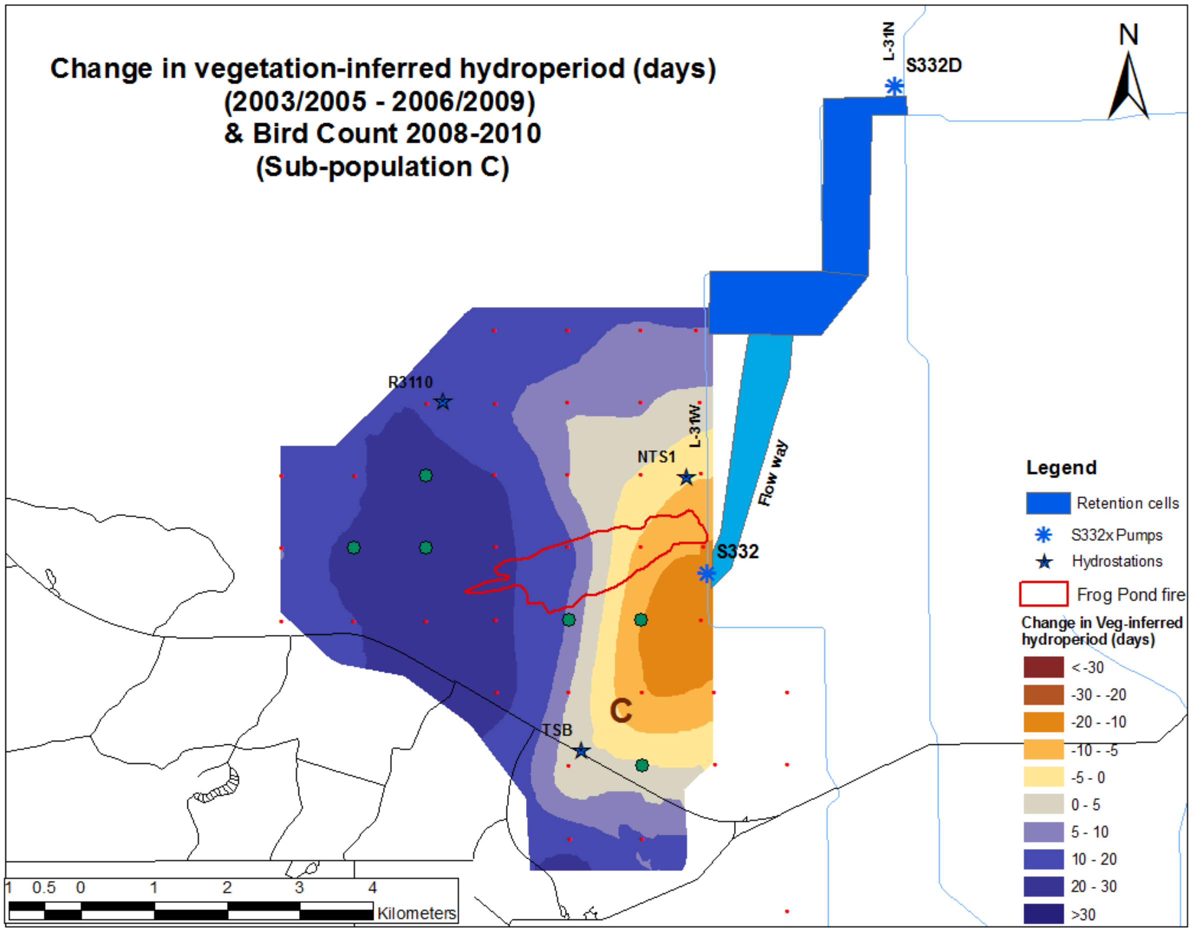

Figure 3.2: Change in vegetation-inferred hydroperiod in sub-population C between the two sampling periods, 2003-2005 and 2006-2009, and the sites with number of sparrows recorded over three years, 2008 to 2010.

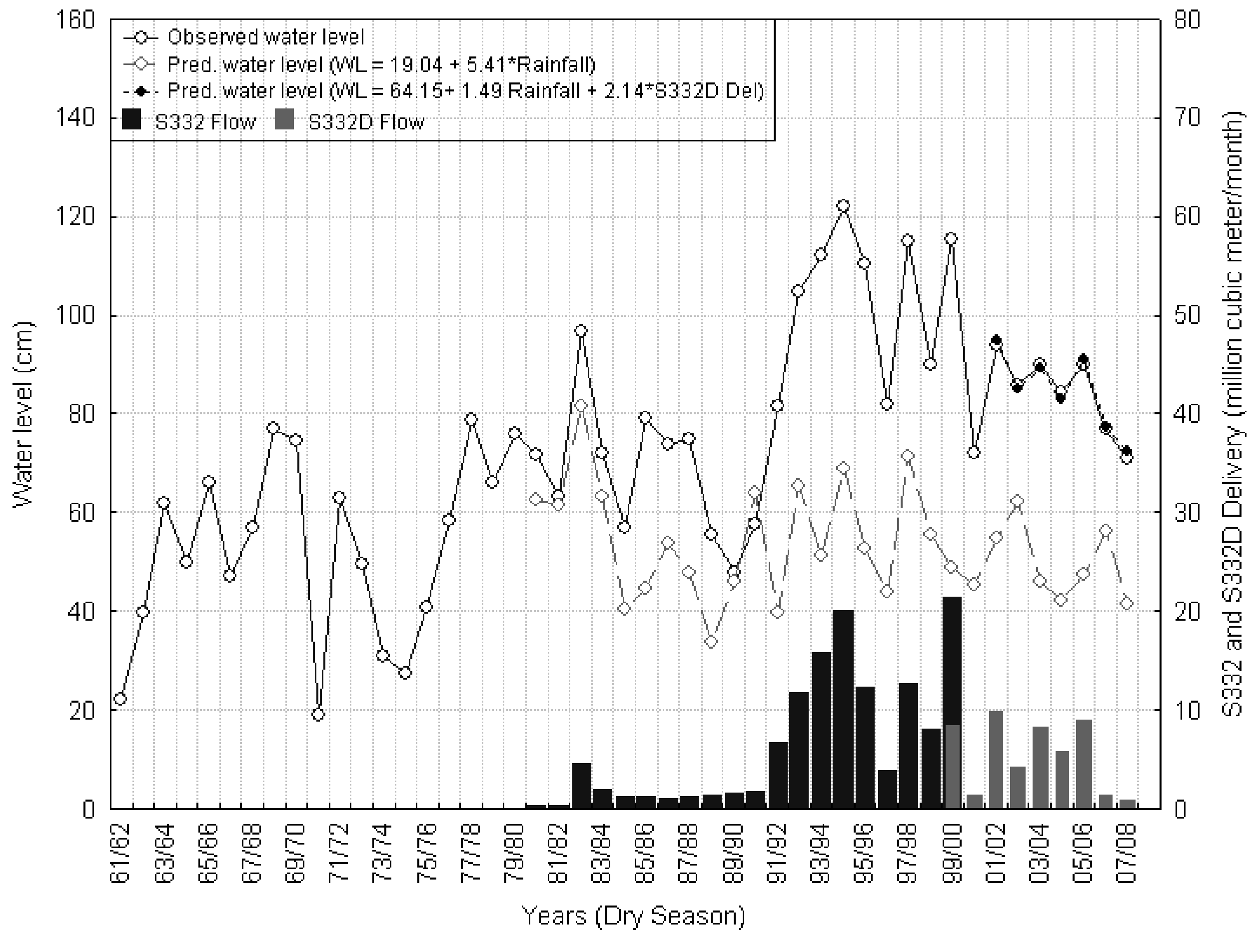

Figure 3.3: Observed and predicted water level at the TSB stage recorder and water flow from L31W canal through S332 into Taylor Slough and from L31N through S332D into retention pond during the 1961-2009 dry seasons.

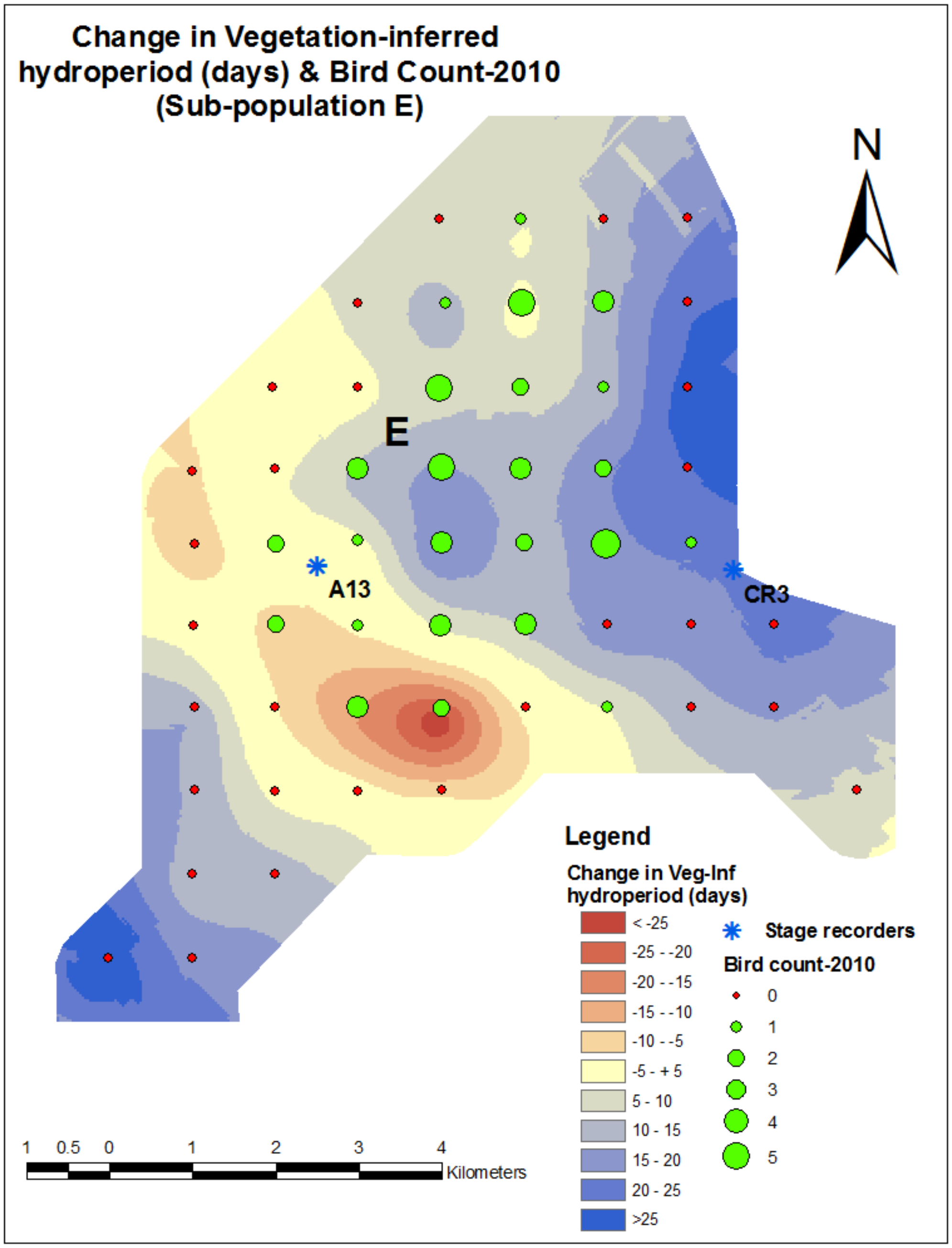

Figure 3.4: Change in vegetation-inferred hydroperiod in sub-population E between the two sampling periods, 2003-2005 and 2006-2010, and sites with number of sparrows recorded in 2010.

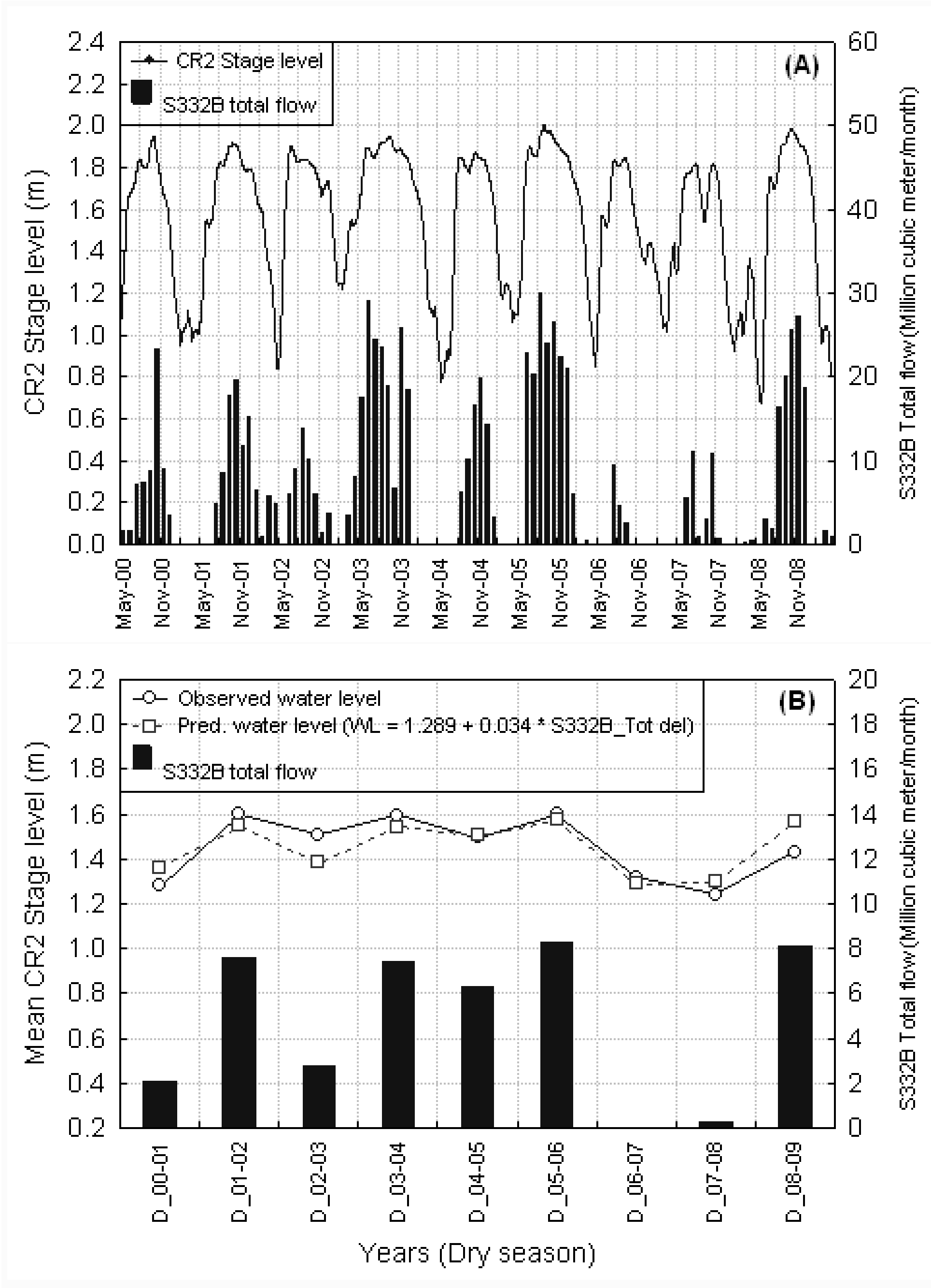

Figure 3.5: CR2 stage level and S332B total flow. (A) 30-day moving average water level at CR2 stage recorder and monthly mean S332B total flow. (B) Observed and predicted water level and seasonal total water flow through S332B (N & W).

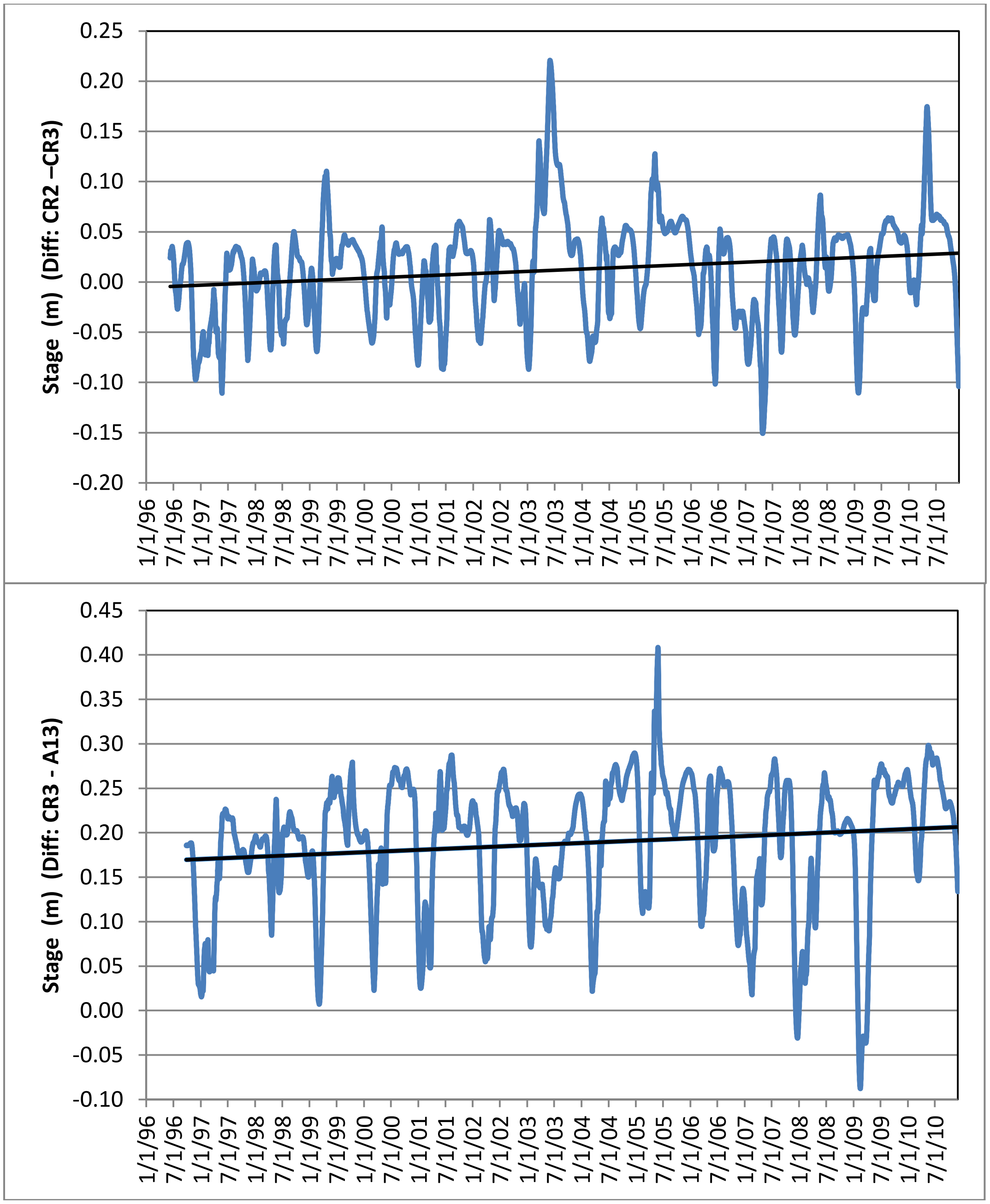

Figure 3.6: Trends in difference in water level between pairs of three stage recorders located between the eastern boundary of ENP and western edge of Shark River Slough. (a) Difference between CR2 and CR3, and (b) Difference between CR3 and A13.

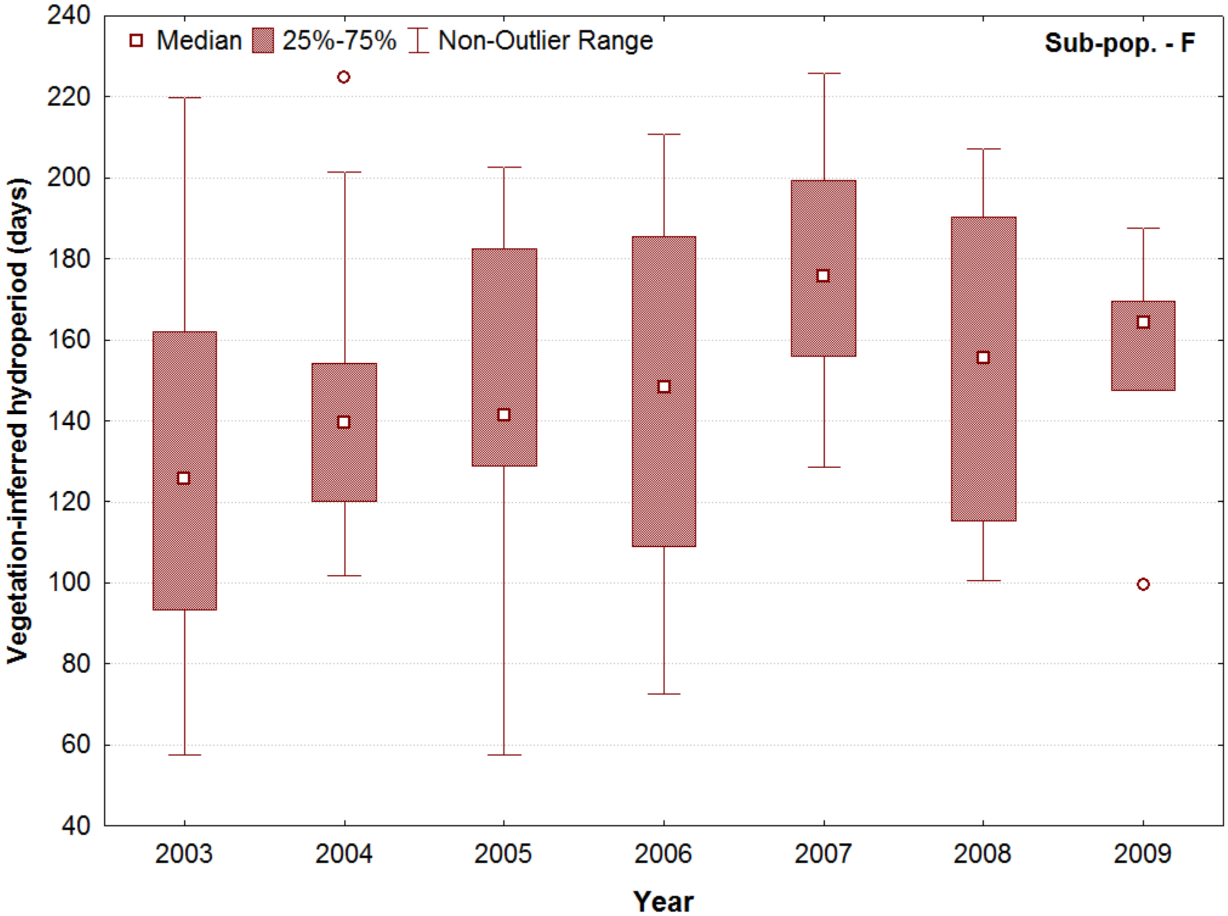

Figure 3.7: Box-plots showing median vegetation-inferred hydroperiod in at the sites sampled for vegetation composition in sub-population F between 2003 and 2009.

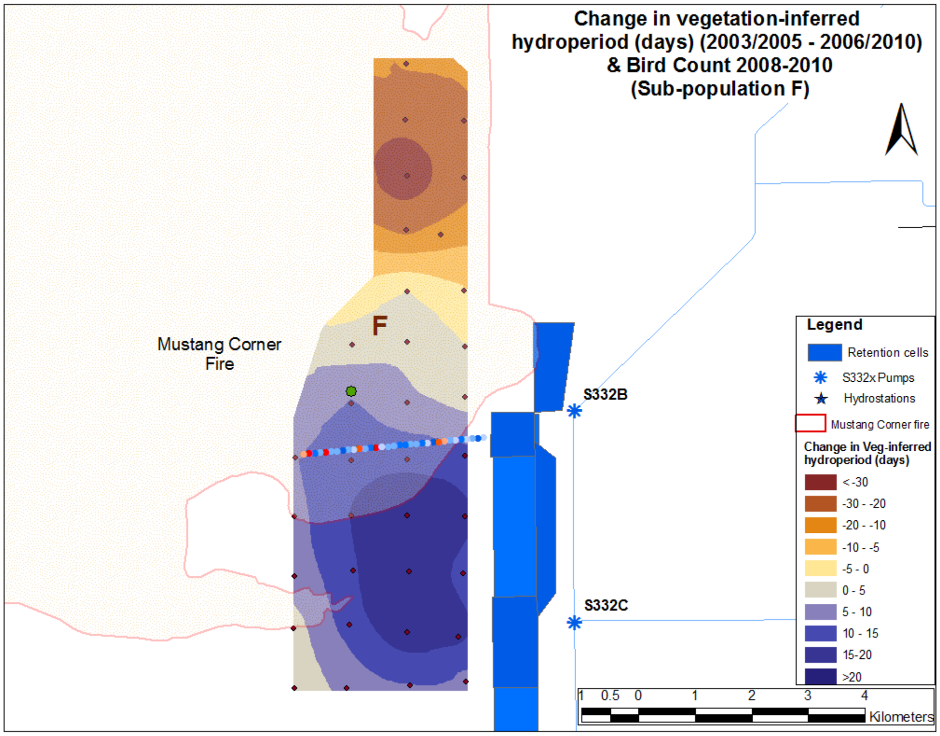

Figure 3.8: Change in vegetation-inferred hydroperiod in sub-population F between the two sampling periods, 2003-2005 and 2006-2010, and sites with number of sparrows recorded over three years, 2008 to 2010.

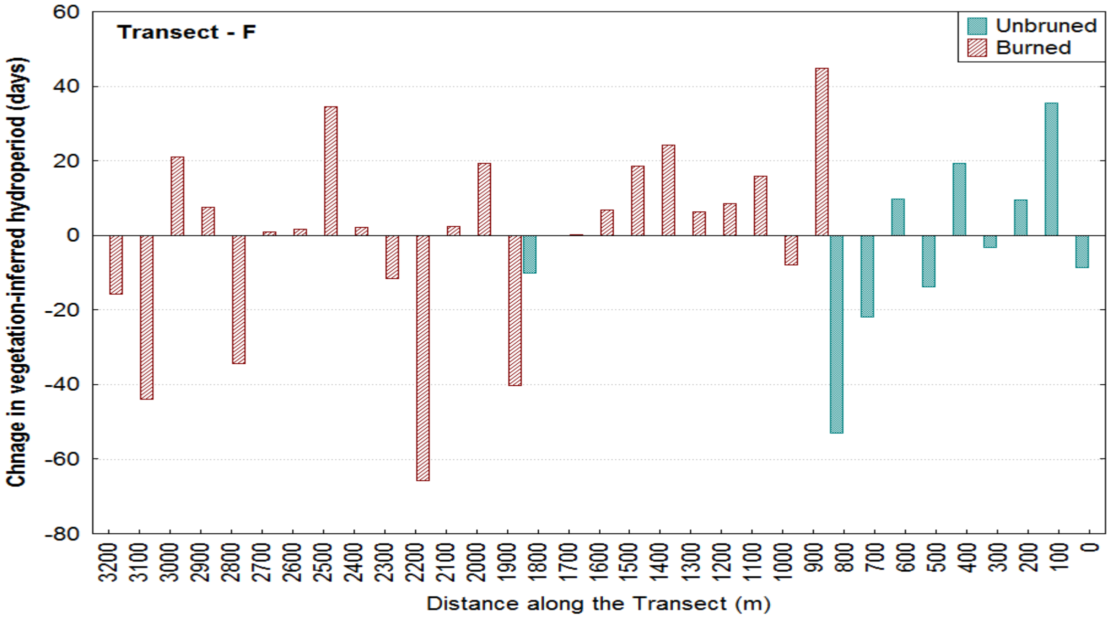

Figure 3.9: Change in vegetation inferred hydroperiod between 2004 and 2010 at the sites on Transect F.

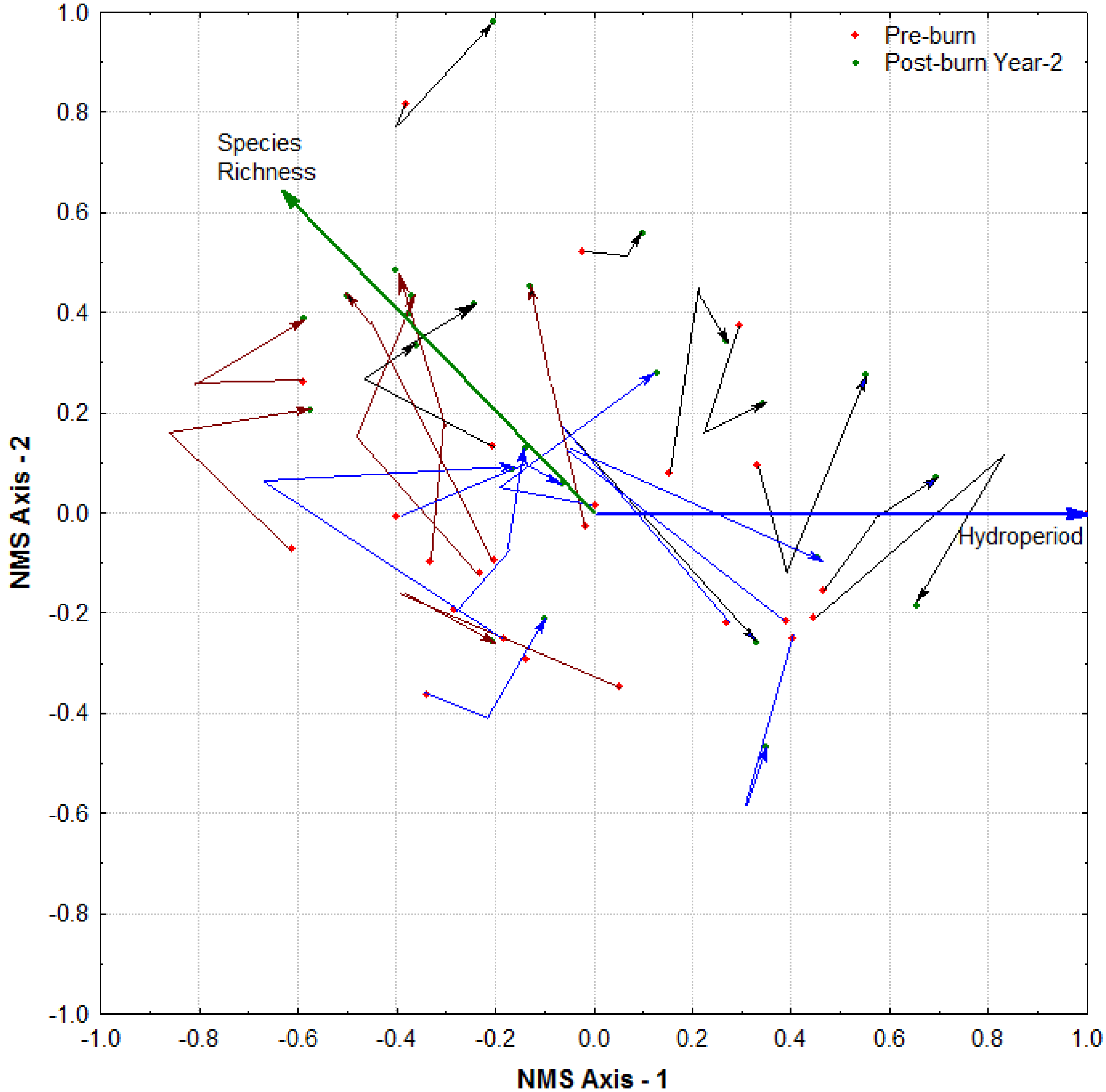

Figure 3.10: Non-metric multidimensional scaling (NMS) ordination showing the trajectory of sites burned in Mustang Corner and Radius Rod fire in CSSS subpopulation E and F. Ordination axes were rotated to perfectly align the hydroperiod vector with the first axis. Initial point on each site trajectory represents the pre-fire, and the end of arrows represents the 2010 (two years after fire) sampling.

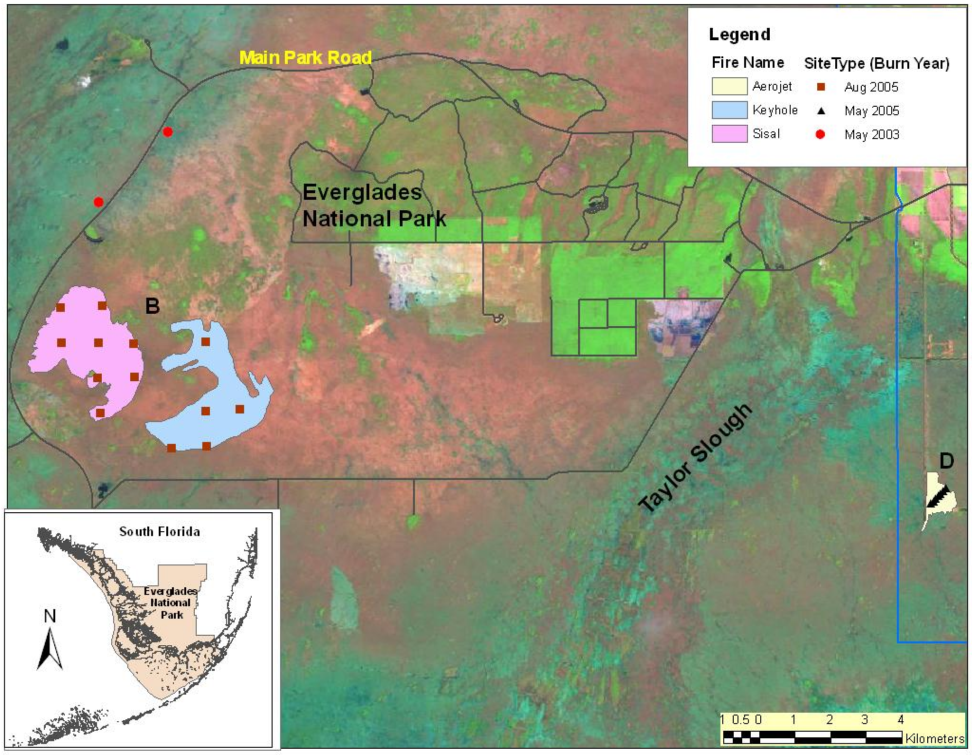

Figure 4.1: CSSS vegetation monitoring sites burned in 2005 fires. Three fires, two in sub-population B and one in sub-population D, burned 21 sites. Two sites burned in 2003 and monitored for 4 years after fire are shown as reference sites.

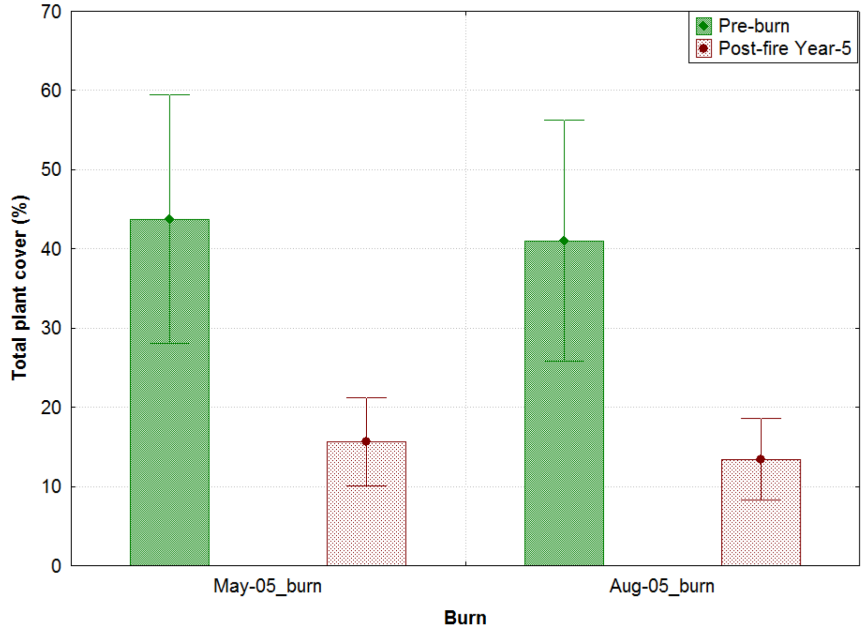

Figure 4.2: Mean (± 1 SD) total plant cover (%) in pre-burn and five year after fire for the two groups of sites, one burned in May 2005 and the other in August 2005.

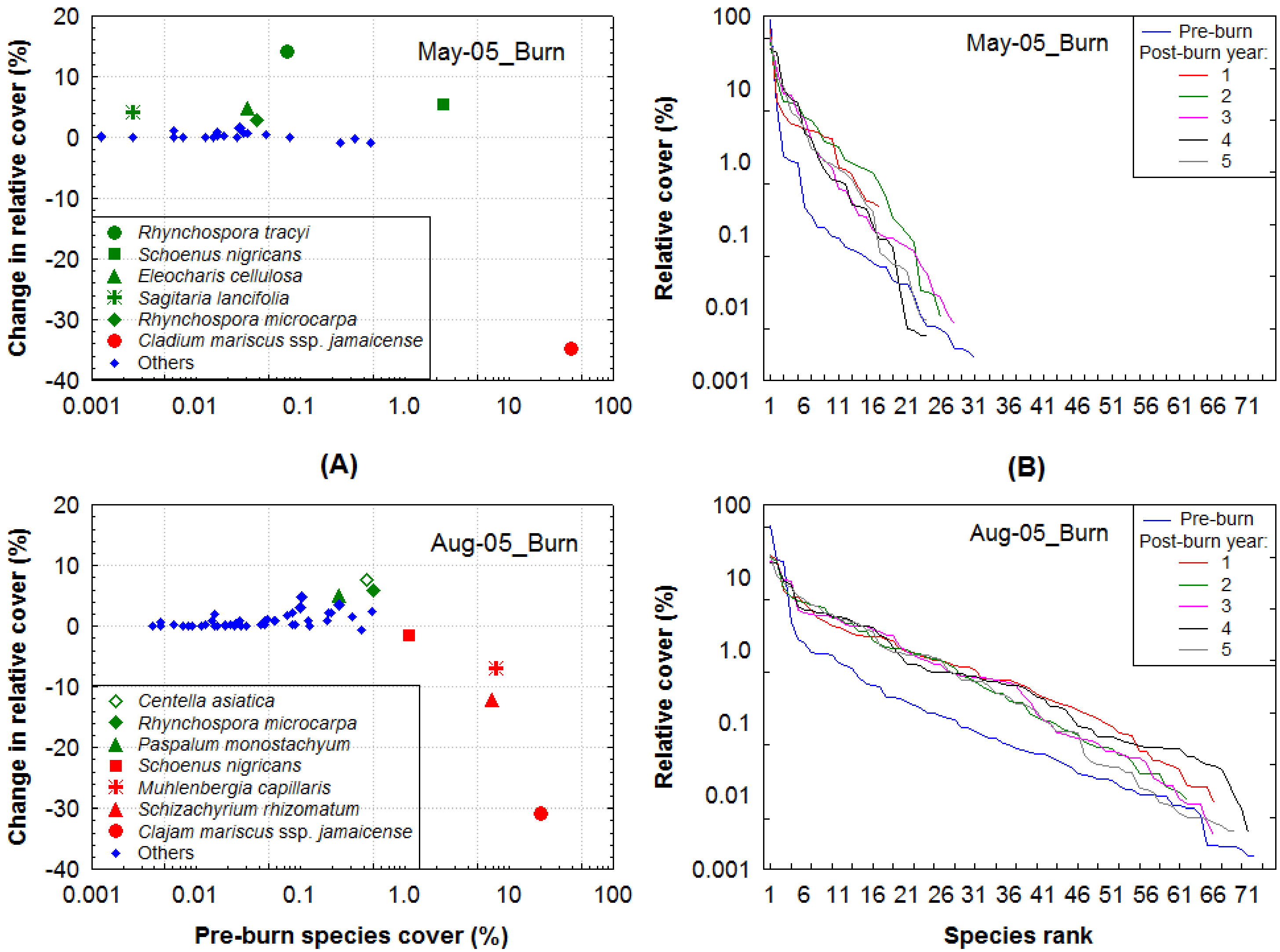

Figure 4.3: Change in the relative cover of species (A) in five years after fire, and species rank abundance (B) at the sites burned in May 2005 (May_burn) and August 2005 (Aug_burn), and re-sampled annually for five post-fire years.

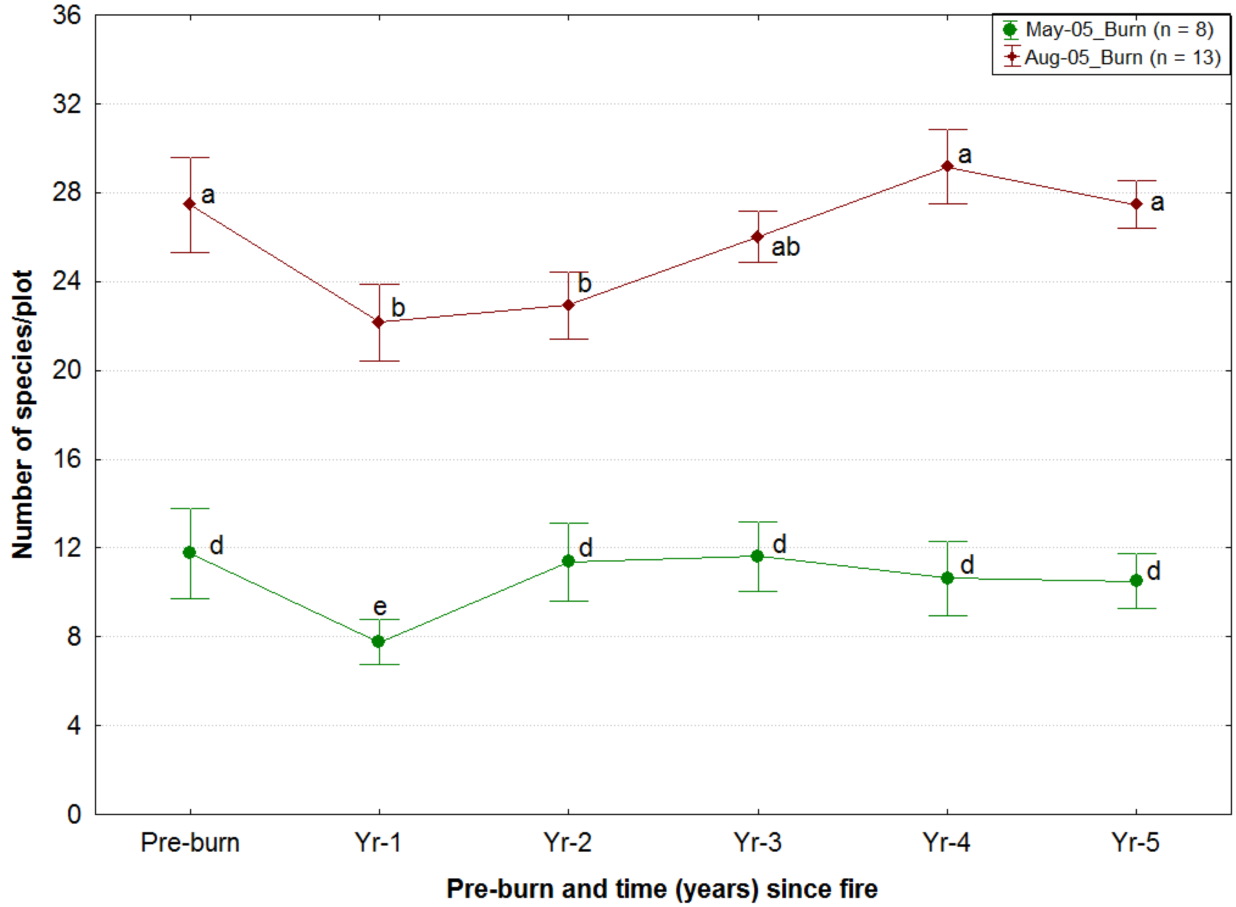

Figure 4.4: Change in number of species at the two sets of census sites, burned in May 2005 (May_burn) and August 2005 (Aug_burn), and sampled annually for five post-fire years. Different letters show differences between burns over the years, and among years within each burn are significantly (Repeater Measures ANOVA, Bonferroni test: $p<0.05$) different.

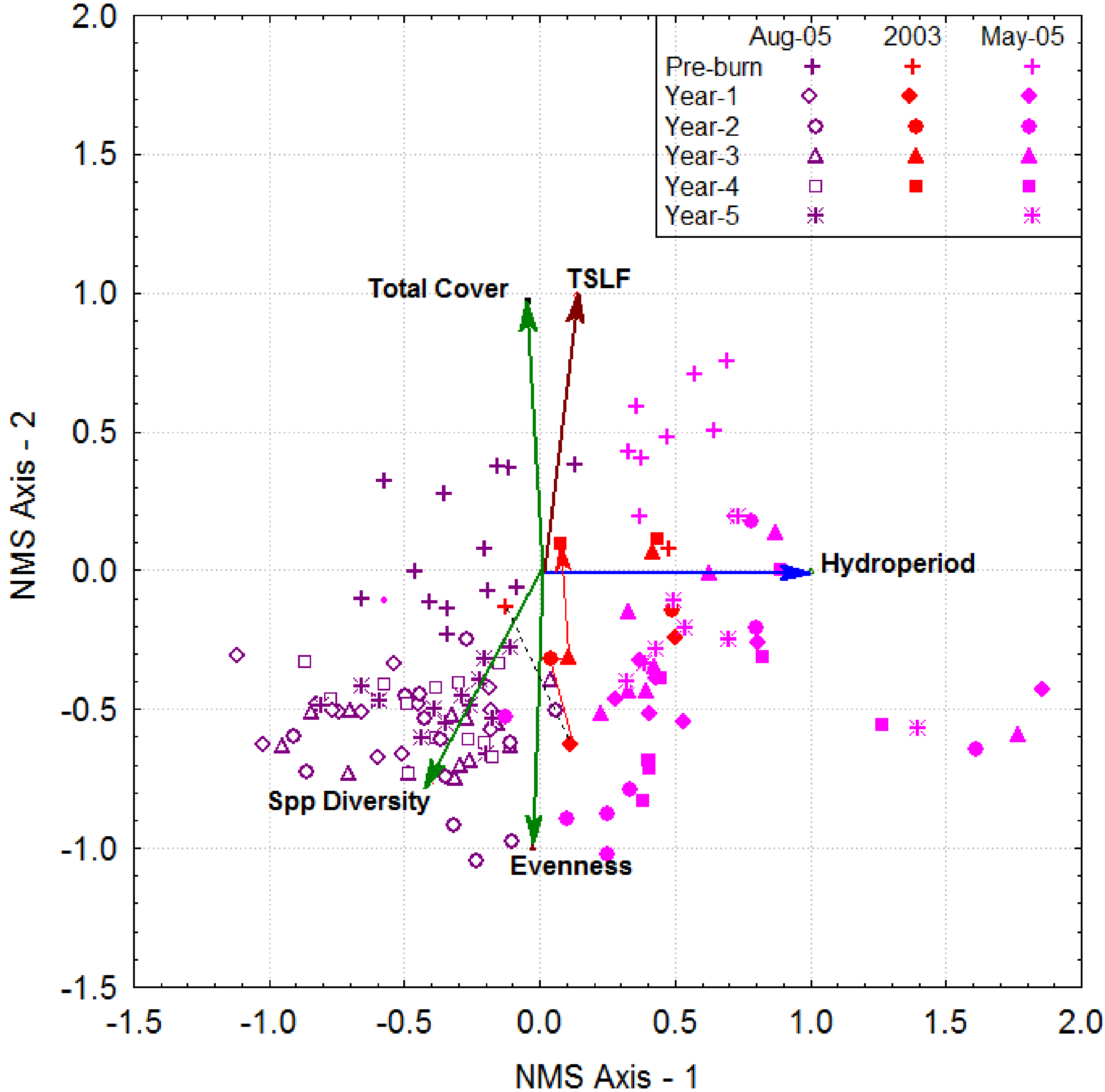

Figure 4.5: Site scores for Axis-1 and 2 from 3-Axis non-metric multidimensional scaling (NMS) ordination based on total cover at 2 sites burned in 2003 and 21 sites burned in 2005. Two sites burned in 2003 were sampled for four years after fire, and are used as reference sites. Sites joining the repeated samples of the same site show trajectory of the site.

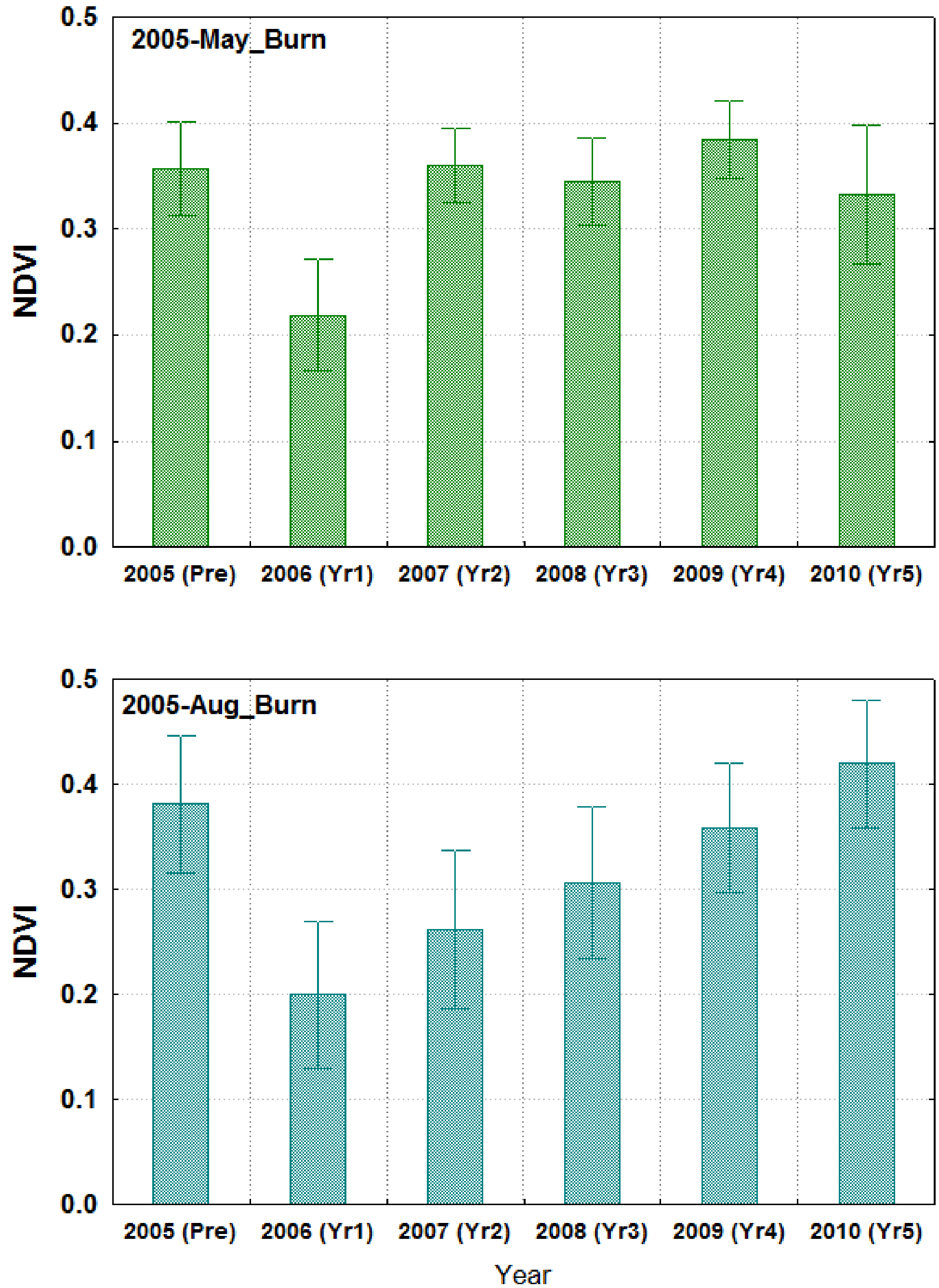

Figure 4.6: Mean (± S.D.) Normalized Difference Vegetation Index (NDVI) in pre-burn and five year after fire in two areas, one burned in May 2005 (Sub-population D) and the other in August 2005 (Sub-population B).

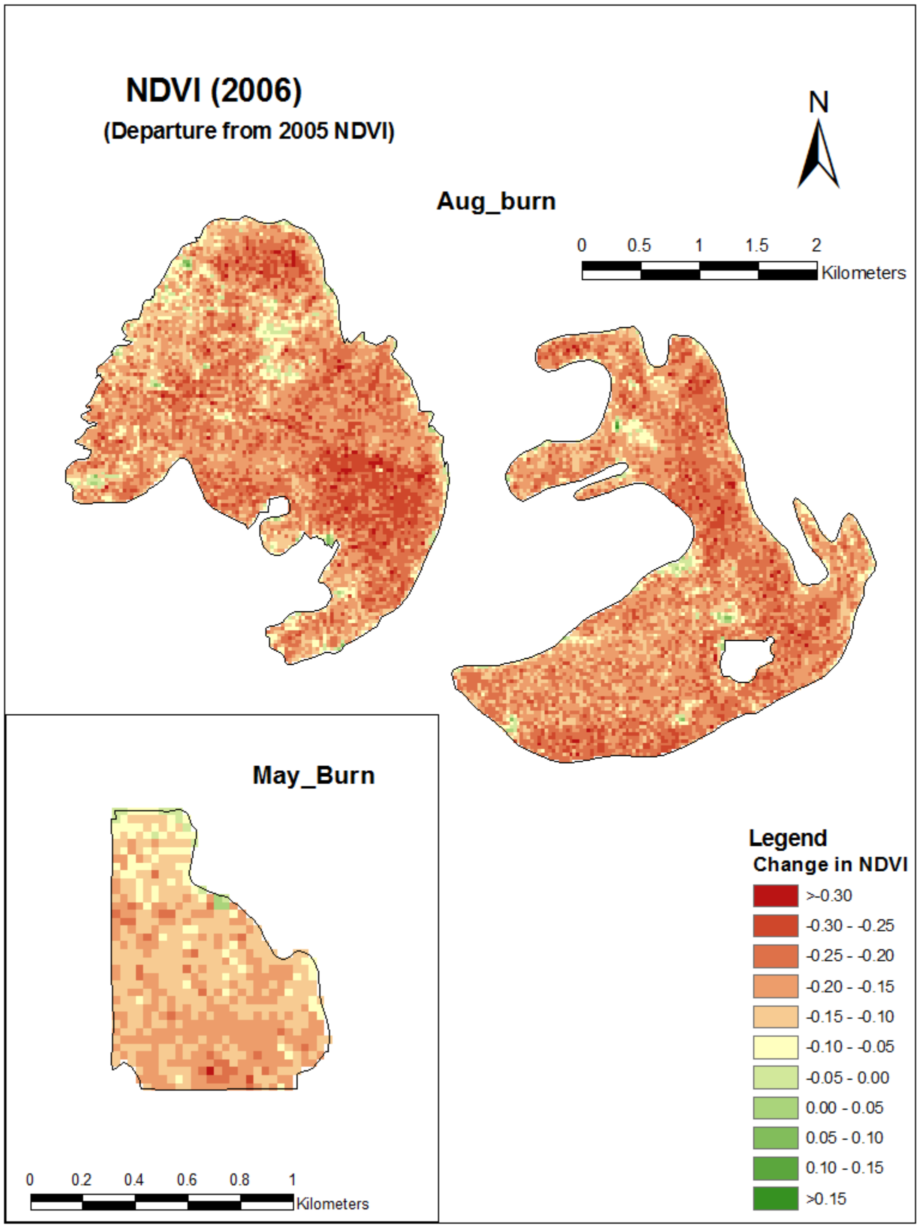

Figure 4.7: Change in NDVI (Normalized Difference in Vegetation Index) between 2005 (pre-fire) and 2006 (Post-fire Year-1) in two areas, one burned in May 2005 (Sub-population D) and the other in August 2005 (Sub-population B).

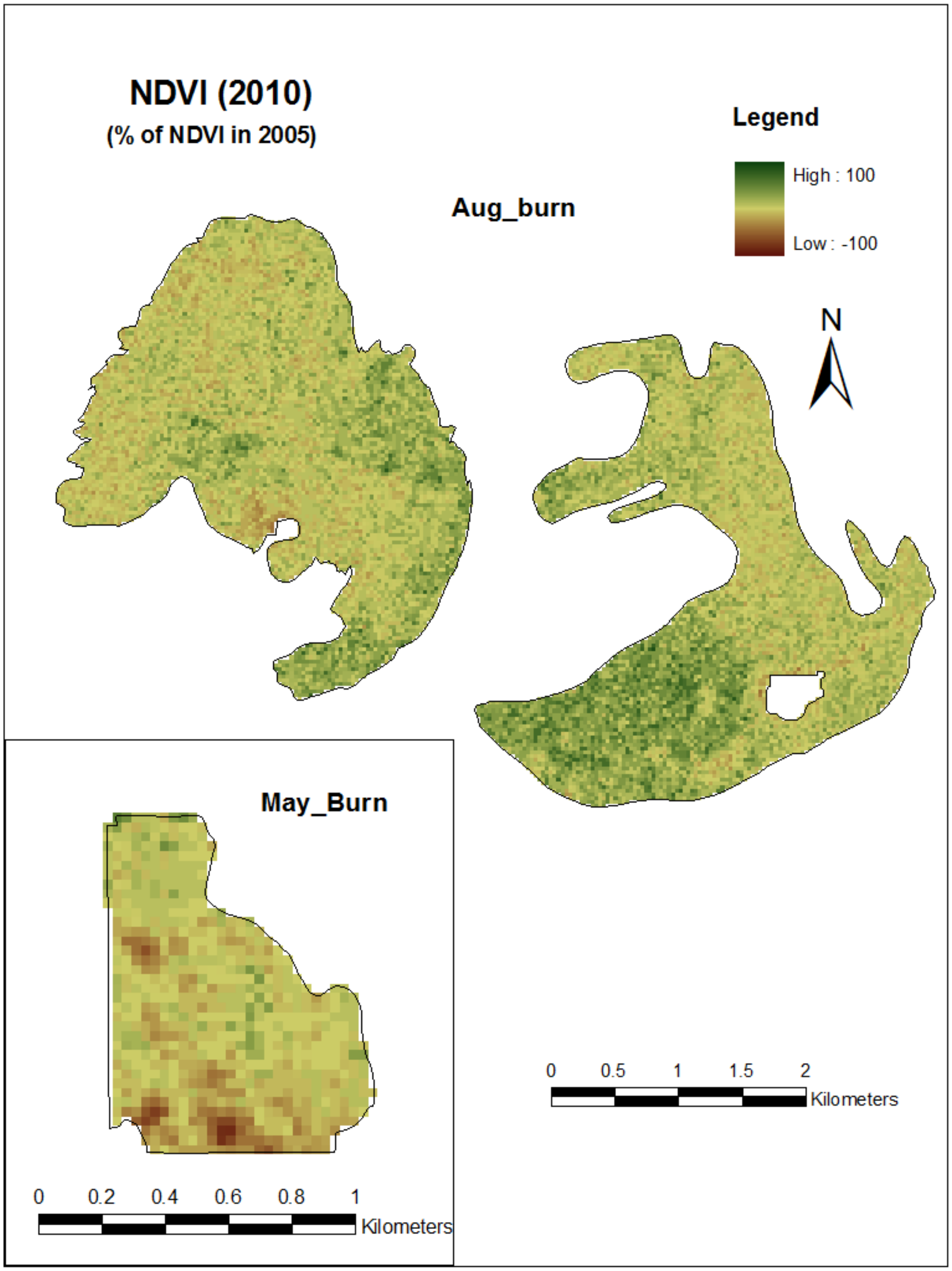

Figure 4.8: NDVI (Normalized Difference in Vegetation Index) values in 2010 (Post-fire Year-5) as a percentage of NDVI in 2005 (pre-fire) in two areas, one burned in May 2005 (Sub-population D) and the other in August 2005 (Sub-population B).

Appendices

A-1: Mean amount (delta) and rate (slope) of trajectory shift representing vegetation change between 2003 and 2009 in the target direction at the census sits within **sub-population A**. Out of 271 census sites, only those sites that showed significant change in vegetation composition are listed. Hydroperiod (days/year) vector fitted in the Bray-Curtis distance-based non-metric multidimensional scaling (NMS) 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.

Site	**X_UTM83**	**Y_UTM83**	**Samp_ year-1**	**Samp_ year-2**	**Veg_ Type (Year-1)**	**Delta**	***p-value***	**Slope**	***p-value***
A-01-06	515125	2844858	2003	2006	CRM	0.415	0.078	0.139	0.078
A-01-08	516146	2842899	2003	2006	RCM	0.966	0.000	0.322	0.000
A-03-01	511118	2833996	2003	2006	SCWP	0.437	0.014	0.146	0.014
A-03-02	513155	2834079	2003	2006	SCWP	0.377	0.021	0.126	0.021
A-03-03	515162	2834850	2003	2006	CRM	0.244	0.000	0.082	0.000
A-03-04	515132	2832965	2003	2006	CM	0.407	0.052	0.136	0.052
A-03-05	516090	2831118	2003	2006	CRM	0.374	0.019	0.125	0.019
A-03-06	515089	2830946	2003	2006	CM	0.151	0.071	0.050	0.071
A-03-07	513029	2831037	2003	2006	SCWP	0.418	0.023	0.139	0.023
A-03-08	511174	2831001	2003	2006	SCWP	0.505	0.005	0.168	0.005
A-03-10	510182	2832018	2003	2006	SCWP	0.548	0.044	0.183	0.044
A-04-02	512186	2829011	2003	2006	CM	0.453	0.026	0.151	0.026
A-04-03	514251	2830027	2003	2006	CRM	0.487	0.071	0.162	0.071
A-04-05	515117	2828015	2003	2006	CM	0.318	0.000	0.106	0.000
A-04-07	516163	2827057	2003	2006	CM	0.108	0.059	0.036	0.059
A-05-07	510251	2825027	2003	2006	CWP	0.301	0.003	0.100	0.003
A-05-08	510217	2824036	2003	2006	CM	0.281	0.008	0.094	0.008
A-06-06	507215	2826006	2003	2006	CRM	0.081	0.035	0.027	0.035
A-06-07	508219	2828071	2003	2006	CM	0.596	0.001	0.199	0.001
A-06-09	509274	2827943	2003	2006	CM	0.343	0.015	0.114	0.015
A-07-01	504175	2829916	2004	2008	ERM	0.147	0.057	0.037	0.057
A-07-06	506175	2832964	2004	2008	RCM	0.177	0.002	0.044	0.002
A-07-08	507193	2831970	2004	2007	RCM	-0.342	0.054	-0.114	0.054
A-08-01	503198	2833998	2003	2006	CRM	0.179	0.068	0.060	0.068
A-08-07	508180	2836880	2003	2006	RCM	0.128	0.056	0.043	0.056
A-08-09	505223	2836901	2003	2006	CM	0.155	0.053	0.052	0.053
A-09-01	506169	2838881	2003	2006	CRM	0.430	0.010	0.143	0.010
A-09-09	511196	2838896	2003	2006	CM	0.152	0.003	0.051	0.003
A-10-03	513091	2831909	2004	2007	SCWP	0.424	0.005	0.141	0.005
A-10-10	513144	2834674	2004	2008	CRM	-0.631	0.029	-0.158	0.029
A-11-01	514044	2835822	2004	2008	CWP	-0.729	0.013	-0.182	0.013
A-11-02	514273	2836753	2004	2008	CM	-0.537	0.009	-0.134	0.009

Site	X_UTM83	Y_UTM83	Samp_ year-1	Samp_ year-2	Veg_ Type (Year-1)	Delta	*p-value*	Slope	*p-value*
A-11-04	516286	2836395	2004	2008	CM	-0.262	0.068	-0.065	0.068
A-11-08	514123	2838811	2004	2008	CWP	-0.303	0.093	-0.076	0.093
A-12-01	511195	2822992	2004	2007	CM	-0.128	0.048	-0.043	0.048
A-12-06	511195	2828986	2004	2008	SCWP	0.330	0.064	0.082	0.064
A-12-08	514248	2826938	2004	2008	CM	-0.232	0.033	-0.058	0.033
A-12-09	516129	2825994	2004	2007	CM	0.075	0.078	0.025	0.078
A-13-03	505932	2824005	2004	2008	CM	-0.142	0.060	-0.036	0.060
A-13-06	509200	2823965	2004	2007	CRM	0.375	0.016	0.125	0.016
A-13-07	506240	2823010	2004	2008	CM	-0.252	0.025	-0.063	0.025
A-13-09	510208	2822032	2004	2007	CRM	0.439	0.052	0.146	0.052
A-14-08	507222	2826980	2004	2008	CRM	-0.277	0.024	-0.069	0.024
A-15-05	506185	2830955	2004	2007	RCM	0.248	0.077	0.083	0.077
A-16-05	512161	2834880	2004	2008	CWP	-0.436	0.065	-0.109	0.065
A-16-07	510141	2834020	2004	2008	ERM	-0.250	0.076	-0.062	0.076
A-16-08	510189	2833097	2004	2008	RCM	-0.277	0.093	-0.069	0.093
A-17-01	510176	2840851	2004	2008	CRM	0.164	0.061	0.041	0.061
A-17-05	512163	2838916	2004	2007	CRM	-0.316	0.039	-0.105	0.039
A-19-05	512065	2841929	2004	2007	CRM	-0.151	0.020	-0.050	0.020
A-19-06	513112	2840887	2004	2007	CWP	-0.270	0.098	-0.090	0.098
A-19-10	516073	2839044	2004	2007	SCWP	-0.515	0.006	-0.172	0.006
A-20-01	510343	2846852	2004	2008	CRM	0.193	0.065	0.048	0.065
A-20-02	509154	2845817	2004	2008	CRM	-0.068	0.019	-0.017	0.019
A-20-06	516073	2845920	2004	2008	CM	-0.218	0.081	-0.055	0.081
A-20-08	513075	2844842	2004	2007	CM	0.106	0.040	0.035	0.040
A-21-06	508166	2843826	2005	2009	CRM	0.300	0.075	0.075	0.075
A-21-10	509194	2841839	2005	2008	RCM	0.175	0.011	0.058	0.011
A-22-07	515122	2841829	2005	2009	RCM	-0.602	0.028	-0.151	0.028
A-22-09	515116	2843812	2005	2009	CM	-0.602	0.022	-0.151	0.022
A-24-04	507180	2840827	2005	2009	CRM	0.142	0.023	0.036	0.023
A-25-04	504188	2835849	2005	2009	RCM	0.235	0.022	0.059	0.022
A-26-03	508179	2834854	2005	2009	RCM	0.146	0.067	0.036	0.067
A-26-07	510194	2836875	2005	2009	CRM	-0.300	0.021	-0.075	0.021
A-27-04	514096	2831997	2005	2009	SCWP	0.235	0.031	0.059	0.031
A-27-10	515128	2835840	2005	2009	CM	-0.178	0.041	-0.044	0.041
A-28-06	509211	2828988	2005	2009	SCWP	0.573	0.029	0.143	0.029
A-28-07	509180	2831039	2005	2009	CRM	0.247	0.077	0.062	0.077
A-29-01	504191	2823944	2005	2009	CRM	0.349	0.016	0.087	0.016
A-29-03	507218	2822973	2005	2009	CM	0.145	0.024	0.036	0.024
A-29-07	508062	2826150	2005	2008	CM	0.612	0.002	0.204	0.002
A-30-09	514119	2828965	2005	2008	CM	0.186	0.076	0.062	0.076

A-2: Mean amount (delta) and rate (slope) of trajectory shift representing vegetation change between 2003 and 2010 in the target direction calculated for 51 sites on **Transect A**. Hydroperiod (days/year) vector fitted in the Bray-Curtis distance-based non-metric multidimensional scaling (NMS) 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.

Site	X_UTM83	Y_UTM83	Veg_type (2003)	Delta	*p-value*	Slope	*p-value*
TA-0000	516665	2841401	SCWP	-0.421	0.017	-0.060	0.018
TA-0100	516565	2841401	PCM	-0.265	0.096	-0.044	0.063
TA-0200	516446	2841401	CWP	-0.247	0.057	-0.038	0.041
TA-0300	516365	2841401	SCWP	-0.257	0.101	-0.035	0.114
TA-0400	516265	2841401	SCWP	-0.424	0.036	-0.061	0.034
TA-0500	516165	2841401	SCWP	-0.554	0.001	-0.079	0.001
TA-0600	516065	2841401	SCWP	0.139	0.735	0.016	0.696
TA-0700	515965	2841401	SCWP	-0.324	0.097	-0.046	0.105
TA-0800	515865	2841401	SCWP	-0.355	0.187	-0.044	0.224
TA-0900	515765	2841401	SCWP	-0.620	0.035	-0.087	0.043
TA-1000	515665	2841401	SCWP	-0.846	0.000	-0.118	0.000
TA-1100	515565	2841401	PCM	-0.141	0.211	-0.022	0.182
TA-1200	515465	2841401	SCWP	-0.368	0.048	-0.053	0.048
TA-1300	515365	2841401	SCWP	-0.008	0.491	0.001	0.503
TA-1400	515265	2841401	SCWP	0.067	0.592	0.012	0.605
TA-1500	515164	2841401	CRM	-0.199	0.190	-0.033	0.150
TA-1600	515065	2841401	CWP	-0.198	0.212	-0.031	0.188
TA-1700	514965	2841401	CWP	0.043	0.554	0.007	0.568
TA-1800	514865	2841401	CM	-0.055	0.391	-0.012	0.341
TA-1900	514765	2841401	CM	-0.265	0.199	-0.038	0.194
TA-2000	514665	2841401	CRM	0.034	0.537	0.005	0.530
TA-2100	514565	2841401	RCM	-0.071	0.385	-0.012	0.358
TA-2200	514465	2841401	CM	-0.330	0.172	-0.050	0.153
TA-2300	514365	2841401	RCM	-0.400	0.109	-0.059	0.102
TA-2400	514264	2841401	CRM	-0.441	0.112	-0.068	0.100
TA-2500	514165	2841401	CRM	-0.206	0.183	-0.033	0.152
TA-2600	514065	2841401	CM	-0.403	0.111	-0.061	0.090
TA-2700	513965	2841401	PCM	-0.061	0.368	-0.009	0.356
TA-2800	513865	2841401	SCWP	-0.060	0.437	-0.009	0.439
TA-2900	513765	2841401	CM	-0.063	0.384	-0.010	0.378
TA-3000	513665	2841401	SOWP	-0.223	0.135	-0.032	0.138
TA-3100	513565	2841401	SOWP	-0.217	0.172	-0.032	0.165
TA-3200	513465	2841401	SOWP	-0.305	0.075	-0.043	0.074
TA-3300	513365	2841401	PCM	-0.289	0.143	-0.045	0.119
TA-3400	513265	2841401	SOWP	-0.491	0.062	-0.072	0.058

Site	X_UTM83	Y_UTM83	Veg_type (2003)	Delta	*p-value*	Slope	*p-value*
TA-3500	513165	2841401	CRM	-0.190	0.206	-0.029	0.192
TA-3600	513064	2841401	PCM	-0.729	0.002	-0.105	0.002
TA-3700	512965	2841401	CRM	-0.248	0.161	-0.036	0.156
TA-3800	512865	2841401	CWP	-0.234	0.079	-0.032	0.081
TA-3900	512765	2841401	CWP	-0.111	0.312	-0.018	0.296
TA-4000	512665	2841401	SCWP	-0.330	0.016	-0.047	0.018
TA-4100	512565	2841401	SOWP	-0.288	0.163	-0.042	0.161
TA-4200	512465	2841401	SOWP	-0.296	0.097	-0.046	0.082
TA-4300	512365	2841401	CM	-0.348	0.059	-0.049	0.061
TA-4400	512265	2841401	CM	0.227	0.972	0.031	0.963
TA-4500	516665	2841401	CRM	-0.211	0.215	-0.035	0.176
TA-4600	516565	2841401	CM	-0.343	0.062	-0.049	0.060
TA-4700	516446	2841401	CM	-0.139	0.273	-0.019	0.281
TA-4800	516365	2841401	CRM	-0.283	0.081	-0.043	0.067
TA-4900	516265	2841401	SOWP	-0.293	0.105	-0.042	0.111
TA-5000	516165	2841401	SOWP	-0.303	0.086	-0.044	0.085

A-3: Mean amount (delta) and rate (slope) of trajectory shift representing vegetation change between 2003 and 2009 in the target direction at the census sites within sub-population C, E & F, sites on Transect F. Sites that showed significant change in vegetation composition are listed. Hydroperiod (days/year) vector fitted in the Bray-Curtis distance-based non-metric multidimensional scaling (NMS) 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.

FieldID	U_B	X_UTM83	Y_UTM83	Samp_ year-1	Samp_ year-2	Veg_ Type (Yr-1)	Delta	*p-value*	Slope	*p-value*
C-01-05	UB	540298	2814227	2003	2006	MWP	0.302	0.011	0.101	0.011
C-01-06	UB	538380	2810405	2003	2006	MWP	0.609	0.003	0.203	0.003
C-01-10	UB	541130	2811251	2003	2006	PCM	-0.300	0.051	-0.100	0.051
C-02-02	UB	538298	2812210	2004	2008	SCWP	0.244	0.041	0.061	0.041
C-02-06	UB	539296	2815161	2004	2007	CWP	-0.306	0.021	-0.102	0.021
C-03-02	UB	541061	2814191	2005	2008	MWP	-0.279	0.096	-0.093	0.096
C-04-02	UB	537346	2814186	2005	2008	MWP	0.255	0.082	0.085	0.082
C-04-03	UB	536331	2813196	2005	2008	MWP	0.280	0.075	0.093	0.075
C-04-04	UB	535344	2813189	2005	2008	MWP	0.282	0.025	0.094	0.025
E-01-01	UB	529376	2822048	2003	2006	CWP	0.213	0.024	0.071	0.024
E-01-02	UB	530372	2824055	2003	2006	CWP	0.203	0.007	0.068	0.007
E-01-04	UB	530350	2822044	2003	2006	SCWP	0.203	0.071	0.068	0.071
E-01-05	UB	531351	2822037	2003	2006	CWP	0.215	0.021	0.072	0.021

FieldID	U_B	X_UTM83	Y_UTM83	Samp_ year-1	Samp_ year-2	Veg_ Type (Yr-1)	Delta	*p-value*	Slope	*p-value*
E-01-08	Burn	532285	2825069	2003	2010	CWP	0.404	0.063	0.073	0.014
E-01-09	UB	532348	2822051	2003	2006	CWP	0.252	0.066	0.084	0.066
E-02-04	UB	529367	2820210	2003	2006	CWP	0.142	0.093	0.047	0.093
E-02-06	UB	531403	2820153	2003	2006	SCWP	0.195	0.034	0.065	0.034
E-02-08	UB	532358	2819185	2003	2006	CWP	0.386	0.005	0.129	0.005
E-02-09	UB	534364	2818180	2003	2006	CWP	0.227	0.022	0.076	0.022
E-03-02	Burn	527397	2817139	2004	2010	SCWP	0.300	0.038	0.043	0.054
E-03-07	Burn	528348	2817156	2004	2010	CWP	0.300	0.083	0.050	0.080
E-04-04	UB	529346	2821021	2004	2008	CRM	-0.414	0.009	-0.104	0.009
E-04-05	UB	530351	2821046	2004	2007	CWP	-0.309	0.096	-0.103	0.096
E-05-03	Burn	531318	2825013	2005	2010	CWP	0.222	0.088	0.036	0.143
E-06-03	UB	530353	2820150	2005	2009	MWP	0.285	0.070	0.071	0.070
E-06-07	UB	527373	2815160	2005	2009	SCWP	0.294	0.014	0.074	0.014
E-06-08	UB	527361	2814156	2005	2009	CWP	-0.321	0.038	-0.080	0.038
E-06-10	UB	526327	2815182	2005	2009	CWP	-0.174	0.080	-0.044	0.080
F-01-02	Burn	542251	2826192	2003	2010	MWP	0.598	0.002	0.047	0.054
F-02-09	UB	540244	2824095	2004	2006	MWP	0.435	0.005	0.145	0.005
F-03-01	Burn	541200	2831069	2005	2010	CM	-0.361	0.117	-0.088	0.056
F-03-03	Burn	541228	2826091	2005	2010	MWP	0.242	0.092	0.044	0.114
TF-0100	UB	542482	2825465	2004	2010	MWP	0.232	0.088	0.023	0.204
TF-0300	UB	542283	2825448	2004	2010	CM	-0.386	0.037	-0.063	0.041
TF-0400	UB	542183	2825438	2004	2010	SCWP	0.298	0.020	0.037	0.052
TF-3000	Burn	539594	2825202	2004	2010	MWP	-0.470	0.013	-0.076	0.014

*Vegetation type: CM = *Cladium* marsh; CRM = *Cladium-Rhynchospora* marsh; CWP = *Cladium* wet prairie; ERM = *Eleocharis-Rhynchospora* marsh; MWP = *Muhlenbergia* prairie; PCM= *Paspalum-Cladium* marsh; RCM = *Rhynchospora-Cladium* marsh; SCWP = *Schizachyrium* wet prairie; SOWP = *Schoenus* wet prairie;