FIU Institute of Environment
FLORIDA INTERNATIONAL UNIVERSITY

Status of Vegetation Structure and Composition within the Habitat of Cape Sable seaside sparrow Subpopulation D

(PO # 4500147501)

Project period: March 06, 2024 - Dec 31, 2024

Final Report – 2024

Submitted to:

Eric Cline
South Florida Water Management District
3301 Gun Club Road
West Palm Beach, Fl. 33460
Tel. (561) 734-3897; Email: ecline@sfwmd.gov

Jay P. Sah[1], Michael S. Ross[1, 2], Susana Stoffella[1], Bianca Constant[1], Sophie Ramos[1], Santiago Castaneda[1],

1. Institute of Environment
2. Department of Earth and Environment

Florida International University, Miami FL 33199

December 31, 2024

Authors' Affiliation

Jay P. Sah, Ph.D. – *Research Professor*
Florida International University
Southeast Environmental Research Center
11200 SW 8th Street, Miami, FL 33199
Tel. (305) 348-1658; sahj@fiu.edu

Michael S. Ross, Ph.D. – *Professor*
Florida International University
Southeast Environmental Research Center
and Department of Earth & Environment
11200 SW 8th Street, Miami, FL 33199
Tel. (305) 348-1420; rossm@fiu.edu

Susana Stoffella, M.S. – *Research Analyst*
Florida International University
Southeast Environmental Research Center
11200 SW 8th Street, Miami, FL 33199
Tel. (305) 348-0493; stoffell@fiu.edu

Bianca Constant – *Research Technician*
Florida International University
Southeast Environmental Research Center
11200 SW 8th Street, Miami, FL 33199
Tel. (305) 348-6066; biconsta@fiu.edu

Santiago Castaneda – *Research Technician*
Florida International University
Southeast Environmental Research Center
11200 SW 8th Street, Miami, FL 33199
Tel. (305) 348-6066; scastane@fiu.edu

Sophie Ramos – Student *Research Technician*
Florida International University
Southeast Environmental Research Center
11200 SW 8th Street, Miami, FL 33199
Tel. (305) 348-6066; soramos@fiu.edu

…………..

Cover photo: *Cladium* Marsh photographed on 04/05/2024 at the site (TD-02-03) which had *Cladium* Wet Prairie vegetation in 2011 (*Photo: Santiago Castaneda*)

Executive Summary

Cape Sable seaside sparrow (CSSS), a federally endangered species, and vegetation within its habitat are highly sensitive to changes in hydrologic regimes. Thus, to help ensure that the impacts of Everglades restoration projects do not impede the continued existence of sparrows in their habitat, the C-111 Spreader Canal Western project, funded by the SFWMD, sponsored regular monitoring of the CSSS sub-population D and its habitat. As per requirements stated in Biological Opinion issued by the US Fish and Wildlife Service (USFWS), baseline conditions of the CSSS sub-population D and its habitat were studied in 2011. A follow up study was also conducted in 2014, two years after the project was implemented, and again in 2016, 2018, 2020 and 2022. With funding support from SFWMD (PO # 4500147501) for FY 2024, the present study examined vegetation composition within CSSS habitat and any vegetation shift that might have occurred between 2011 and the subsequent surveys.

The sampling design included two groups of sites: (1) sparse vegetation sampling sites (SS sites), and (2) concentrated vegetation sampling sites (CS sites). The 44 SS sites were 500 m to 1 km apart, whereas the 36 CS sites were at the corners of each 250 x 250 m grid cell in an area of 1.25 km x 1.25 km. At each site, vegetation was sampled using a nested design: a 5 m x 5 m shrub plot was nested within a 10 m x 10 m tree plot. Within shrub plots, cover of shrubs and vines were estimated. Herbaceous plants were surveyed within five 1-m^2 subplots located within each shrub plot. In addition to species cover, a suite of structural parameters was recorded in a 0.25 m^2 quadrat in the southeast corner of each subplot. Hydrology data from Everglades Depth Estimation Network (EDEN), a USGS website that supplies hydrology data for the Greater Everglades, was used to calculate annual mean daily water depth and hydroperiod for the plots. Vegetation change analysis included Analysis of Similarity (ANOSIM), change in vegetation-inferred hydroperiod, and trajectory analysis. Changes in vegetation-inferred hydroperiod between successive samplings are indicative of vegetation changes in response to hydrology of the period. The trajectory analysis method has made it possible to detect a shift in vegetation composition along a gradient representative of increasing or decreasing wetness. General linear mixed models (GLMM) were used to test for differences in vegetation structural variables (vegetation cover and height), biomass, and vegetation-inferred hydroperiod among five sampling events, whereas a generalized linear mixed model was used to test for differences in species richness. Non-parametric Friedman-Test together with Wilcoxon matched-pair test was used to test differences in 4-year average hydroperiod and mean annual water depth, and major species' abundance among sampling events.

Hydrologic conditions within the habitat of sub-population D have become much wetter in recent years than a decade ago. For instance, the 4-year average hydroperiod in 2024 was 99 days longer and mean annual water depth was 14.2 cm higher during the 2024 survey than the pre-project period, i.e., before the baseline survey in 2011. Marl prairie vegetation within the habitat of this sub-population included vegetation assemblages, mainly grouped into two broad-groups: i) wet prairie and ii) marsh, arranged along the full hydrologic gradient. Since 2011,

vegetation change was marked by an increase in wetness of several sites and a consequent shift in species composition toward a vegetation type characteristic of wetter conditions. Between 2011 and 2024, vegetation at sixty-nine percent of marl wet prairie sites had changed to marl marsh vegetation types. However, a shift in species composition toward a more hydric type occurred even between 2011 and 2014, i.e., in the first 3-years after the baseline survey. Thereafter, relatively dry conditions in 2014 and 2015 might have helped to improve habitat condition, as evidenced by an increase in ephemeral sparrow population in those years. However, in the 2016 dry season (Nov 1^{st} – April 30^{th}), the mean water level was unusually high, more than 11.2 cm above the 33-year average of dry season water depths. Similarly, in the next eight water years, hydroperiod as well as mean annual water level was higher than the long-term (33-year) average, which resulted in the vegetation shift to a wetter type in comparison to the baseline survey.

Since an increasing trend in wetness in marl prairies beyond 210 days hydroperiod is envisaged as gradual deterioration of sparrow breeding habitat conditions, the increase in 4-year average vegetation-inferred hydroperiods from 210 days in 2011 to 233 days in 2024 could be an indication of deteriorating CSSS habitat. However, more successful sparrow nesting in three years (2018-2020) than previous years and higher number of sparrows recorded in 2021 and 2022 than during previous surveys were contrary to our expectation. Regardless of the early signs of recovery of sparrow population in the area, sub-population D is vulnerable and in the future is likely to be adversely impacted by increasing wetness and shift in vegetation from short-hydroperiod wet prairie to marsh types. Thus, it is important to minimize the chances of high-water condition in coming years, especially during the dry season, so that the observed vegetation trend will not accelerate. This is essential especially within the sub-population D habitat, where the hydrologic conditions are likely to continue being impacted by the C-111 Spreader Canal Western Project activities.

Table of Contents

1. Introduction

In the Everglades, Cape Sable seaside sparrow (CSSS; *Ammodramus maritimus mirabilis*) and its habitat have been at the pivot of several water management activities for the last two decades, affecting marl prairie vegetation on both sides of the Shark River Slough (SRS). The reason for its critical role is that CSSS is a federally listed endangered species endemic to the short-hydroperiod marl prairies of the Everglades, and both the sparrow and vegetation that structures its habitat are highly sensitive to changes in hydrologic regime (Nott et al. 1998; Pimm et al. 2002). Unusually high-water conditions during the sparrow's breeding period can cause sharp decline of the sparrow population, either directly by inflicting mortality or impairing breeding success, or indirectly through destruction of its habitat (Pimm et al. 2002; Jenkins et al. 2003; Virzi et al. 2011). Flooding that extends hydroperiod can cause the short-hydroperiod marl prairie to change to long-hydroperiod sawgrass marsh as quickly as within 3-5 years (Armentano et al. 2006; Sah et al. 2014), resulting in the habitat becoming unsuitable for sparrows (Nott et al. 1998; Jenkins et al. 2003). Thus, to help ensure that impacts of Everglades restoration projects to sparrow habitat do not impede the survival and continued viability of sparrows, several water-management projects in the Southern Everglades include regular monitoring of the sparrow population and its habitat as integral components.

The C-111 Spreader Canal Western Project (C-111 SC Project or 'The Project") aims to restore the quantity, timing, and distribution of water delivered to Florida Bay via Taylor Slough and to improve hydroperiod and hydro-pattern in the area south of the C-111 canal, known as the Southern Glades and Model Lands. To ensure that the project impacts to CSSS Designated Critical Habitat Units 2 and 3 (also referred to as subpopulations C and D, respectively) do not exceed the impacts recognized in the US Fish and Wildlife Service (USFWS's) Incidental Take Statement (ITS), the SFWMD is mandated to conduct monitoring of CSSS subpopulation D and its habitat. As per the requirements stated in Term and Condition #6 of ITS, baseline conditions of the CSSS sub-population D and its habitat were studied with funding support from SFWMD in 2011, before implementation of project operations (Virzi et al. 2011). After the completion of the construction of C-111 Spreader Canal by SFWMD in Feb 2012, project operations began in the same year and a follow up study of CSSS sub-population D habitat was conducted in 2014 and every two years thereafter, i.e., 2, 4, 6, 8 and 10 years after the implementation of the project, respectively (Sah and Ross 2014; Sah et al. 2016, 2018, 2020, 2022). The baseline study concluded that the population had declined from a peak of 400 birds in 1981 to only a few pairs in the mid-2000s (Virzi et al. 2011), which corresponded with a change in vegetation from short-hydroperiod prairie to the long-hydroperiod sawgrass marsh during that period (Ross et al. 2004). The study also emphasized that the population began to show signs of improvement during 2007-2010 period that corresponded with an enhancement in habitat conditions resulting from a drying trend in the late 2000s (Virzi et al. 2011). However, it was then expected that the trend would be disrupted upon project implementation, as computer simulation modeling results indicated that operations would result in an increased hydroperiod, and thus adversely affect the habitat conditions within the CSSS subpopulation D critical habitat (USFWS 2009).

In 2014, an examination of daily stage data at EVER4, located in the center of the CSSS sub-population D habitat, revealed that the three year-period (May 1, 2011 – April 30, 2014) following the 2011 baseline survey (Project period) was slightly wetter than during the three years (May 1, 2008 – April 30, 2011) before the survey (Pre-project period). In agreement with wetter hydrologic conditions in the post-project than pre-project period, a shift in species composition toward a vegetation composition characteristic of wetter conditions was also observed (Sah and Ross 2014). However, at the time it was not clear whether the shift in habitat conditions was due to project activities or natural annual variability in hydrologic conditions, or both.

After 2014, the shift in vegetation composition towards wetter type continued for the next four years, though at a slower pace. In fact, a non-significant difference in vegetation-inferred hydroperiod between 2014 and 2016 suggested that the habitat condition had not declined any further (Sah et al. 2016) during the period. In contrast, dry conditions in 2014 and 2015 might have helped to improve habitat condition, as evidenced by an ephemeral increase in sparrow population in those years. A mix of both positive and negative trends in the sparrow population in subpopulation D was observed during the following two years, 2014 and 2015 (Virzi and Davis 2014; Virzi et al. 2015). In the 2016 dry season, however, the water level was unusually high, more than 11.2 cm above the 33-year average of dry season water depths, even limiting the scope of that year’s sparrow survey (Virzi and Davis 2016). The long-term effect of the unusual high-water condition on vegetation was also uncertain at that time and was expected to depend on the hydrologic regime in subsequent years (Sah et al. 2016). In the next six years, while the vegetation condition was trending towards a wetter type (Sah et al. 2018, 2020, 2022), the sparrow surveys revealed mixed results. For instance, sparrow population in 2017 was moderately lower, but in 2018, 2019 and 2020, the sparrow number was higher than in 2014 or 2015 (Virzi and Davis 2017; Virzi and Murphy 2018; Virzi and Tafoya 2019, 2020). While a fine scale survey of the sparrow population had not been conducted after 2020, the Park helicopter survey showed an increasing sparrow population for the next two years, 2021 and 2022, but then a decline in 2023, suggesting that a certain level of uncertainty persists regarding the sparrow population and their habitat in the area. Thus, it was obvious that only a regular monitoring of the vegetation could provide a conclusive assessment of the course of the sparrow habitat and its population within the sub-population D habitat where the hydrologic conditions are impacted by the project activities.

With funding support from SFWMD (PO # 4500147501) for FY 2023/2024 (hereafter FY 2024), we studied the status of sparrow subpopulation D habitat in 2024. The specific objective of this study was to document the status of vegetation structure and composition within the habitat of CSSS sub-population D, and to analyze the magnitude and direction of any vegetation change that might have occurred since the baseline survey was performed in 2011. In this study we ask if there is a consistent increase in wetness within the sparrow habitat, and how the vegetation structure and composition track the changes in hydrologic conditions during the twelve years post-project. We hypothesized that vegetation would continue trending towards wetter types in the post-project period.

2. Methods

2.1 Study design

The study area was within the critical habitat of CSSS sub-population D (Figure 1). The survey design was the same used in the 2011 baseline and the previous five (2014, 2016, 2018, 2020 and 2022) post-project surveys, and included two groups of sites, (1) sparse vegetation survey sites (SS sites), and (2) concentrated vegetation survey sites (CS sites). Together there were 80 sites - 44 SS and 36 CS sites. The SS sites included 17 previously surveyed vegetation census sites located at the corners of 1 km x 1 km grid cells (Ross et al. 2006), and 27 sites that were established in 2011 either at the corners of additional grid cells included in the critical habitat boundary of sub-population D, or at the centers of the aforementioned grid cells. The CS sites were at the corners of each 250 x 250 m grid cell within a 1.25 km x 1.25 km area that included a set of occupied CSSS territories that had been delineated by Dr. Thomas Virzi and group (Virzi et al. 2011; Virzi and Davis 2013) at the time of project initiation.

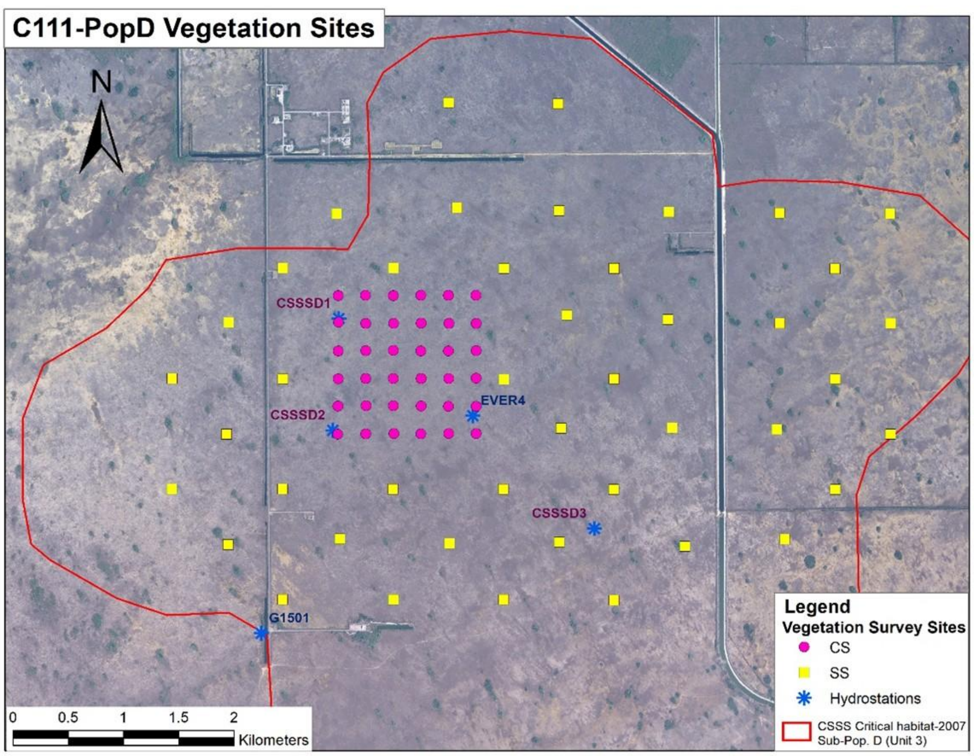

Figure 1: Vegetation survey sites within C-111 Spreader Canal Western Project – CSSS Sub-population D area. The sampling design included two groups of sites: (1) sparse vegetation sampling sites (SS sites), and (2) concentrated vegetation sampling sites (CS sites)

2.2 Field Sampling

We commenced vegetation sampling on March 9th and continued through April 10th, 2024. First, we sampled 4 sites by accessing them on foot. We accessed these four sites by driving to the nearest point on the Aerojet Road, and then walking to them. Later, as per our schedule to work using HMC helicopter services, we worked for six days (March 20th, 28th, and April 1st, 5th, 8th, and 10th), and sampled at 76 sites (Figure 2). In the field, we recorded structural and compositional vegetation parameters at both SS and CS sites following the methods used in the 2011 baseline survey (Sah et al. 2011; Virzi et al. 2011).

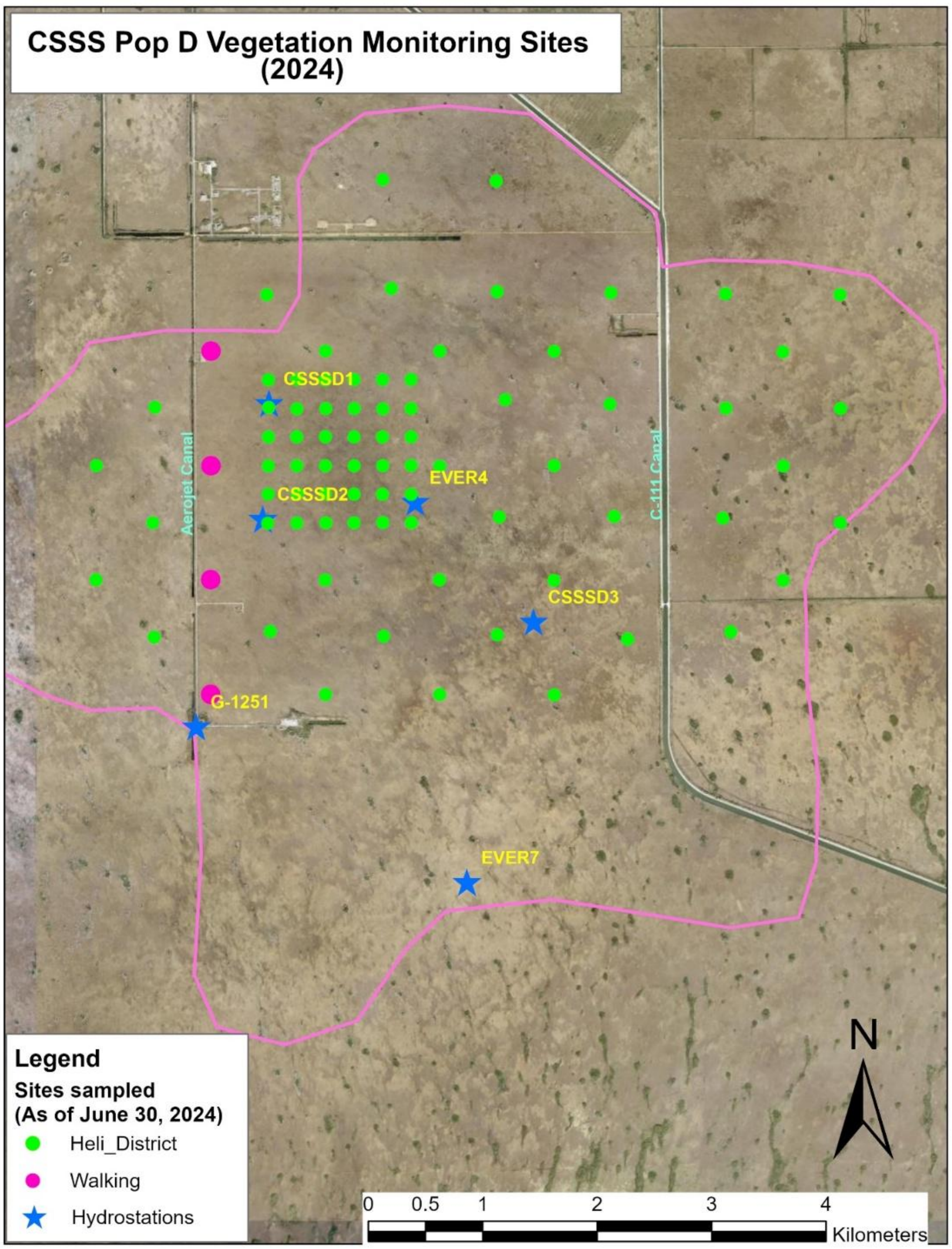

Figure 2: Location of sites within the Cape Sable seaside sparrow (CSSS) sub-population D habitat sampled for vegetation structure and composition in FY 2024.

At each sampling site, a 3-ft tall PVC pole marked the SE corner of a 10 m x 10 m tree plot. Nested within each tree plot, a 5 m x 5 m herb/shrub plot was laid out, leaving a 1-m buffer strip along the southern and eastern border of the tree plot (Figure 3).

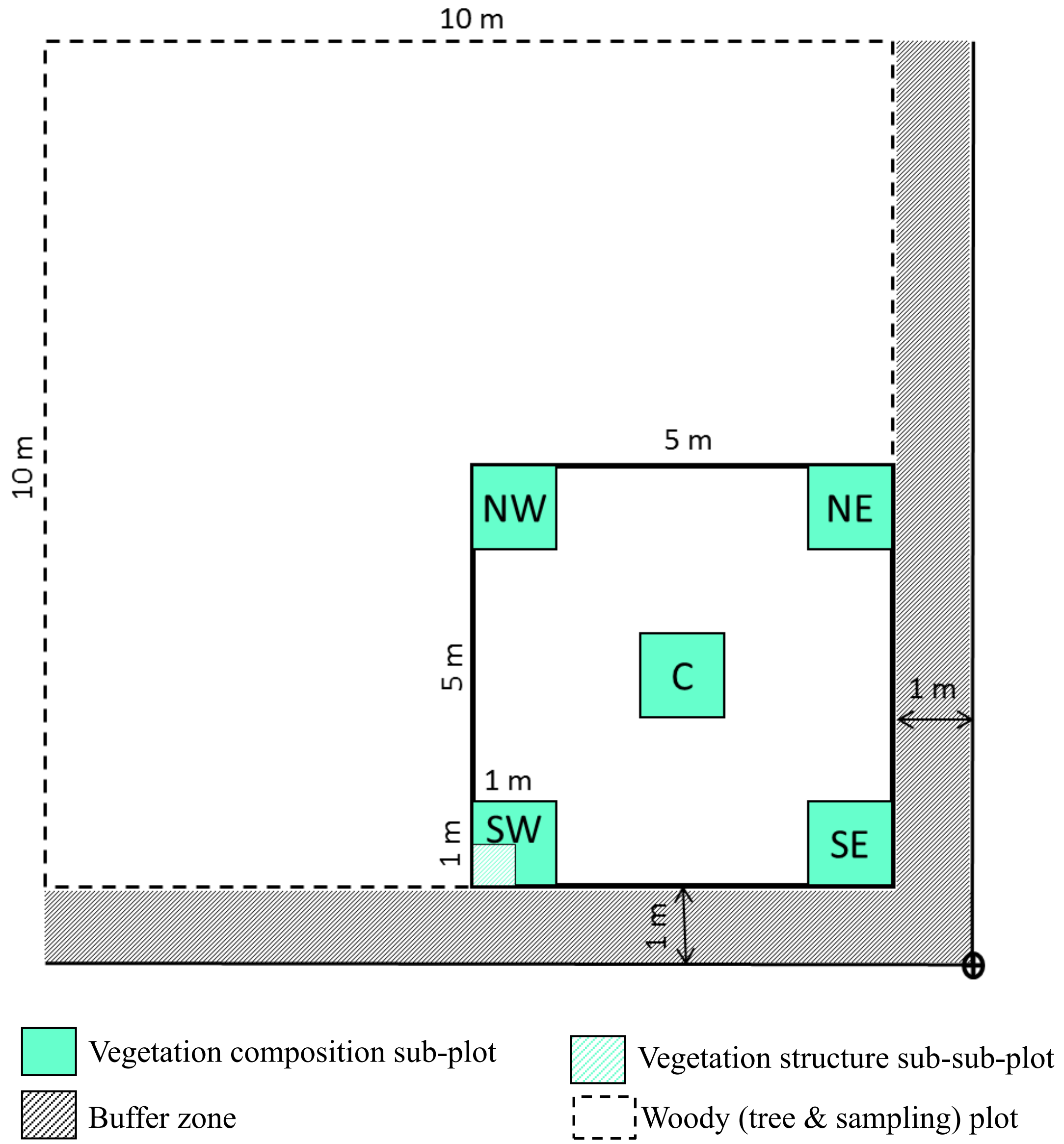

Figure 3: Vegetation sampling design at each of 80 sites sampled in 2024 to document status of vegetation structure and composition in the habitat of CSSS sub-population D within C-111 Spreader Canal Project Area.

In the tree plots, whenever there were trees present, we measured the DBH and crown length and width of any woody individuals of ≥ 5 cm DBH. Within each 5 m x 5 m herb/shrub plot, we estimated the cover class of each species of shrub (woody stems >1m height and < 5cm DBH) and woody vines, using the following categories: < 1%, 1-4%, 4-16%, 16-33%, 33-66%, and > 66%. Herbaceous plants were surveyed within five 1-m^2 subplots located at the four corners and center of each herb/shrub plot. In 1-m^2 subplots, we estimated the percent cover of each vascular plant species, using the same categories as we used for shrub cover. If an herbaceous species was present in the 5 m x 5 m herb/shrub plot but not found in any of the subplots, it was assigned a mean cover of 0.01%. In addition, a suite of structural parameters was recorded in a 0.25 m^2 quadrat in the southeast corner of each subplot. Structural measurements included the following attributes: 1) Crown height, i.e., the tallest vegetation crown present within a cylinder of ~5 cm width, measured at 4 points in each 0.25 m^2 quadrat; 2) The height and species of the tallest plant in the quadrat; 3) Total vegetative cover, in percentage; and 4) Live vegetation, expressed as a percent of total cover. The number of woody individuals (height ≤ 1 m) present in the subplots was also recorded. In addition, if there was standing water in the herb/shrub plots, we also measured water depth in each subplot. We took photographs of some of survey sites to document the field conditions in the digital format (Figure 4). During our fieldwork, we took extra precautions to minimize the impact when we surveyed the sparrow-occupied area mapped by Virzi and Murphy (2018) and Virzi and Tafoya (2019, 2020).

In the field, whenever there was standing water, we measured water depth in each sub-plot. However, during the vegetation survey, about 4% of sub-plots were dry and the rest 14.5% subplots had shallow (<10 cm, mean ± SD: 5.9±1.9 cm) standing water with high variability. Thus, in the wet season of 2024, we measured water depths in all those subplots in which water depths were <10 cm during the time of vegetation survey, so that all subplots, but one, had ≥10 cm water depth when measurements were taken. One sub-plot had water depth of 9 cm.

Figure 4: Vegetation survey in C111-CSSS subpopulation D (Site: TD-04-05) (*Photo: Santiago Castaneda*).

2.3 Data analysis

2.3.1 Hydrology

Until 2022, for consistency in data analysis across the sampling years, hydrological variables were calculated based on elevations determined from water depths measured in the wet season of 2011, when most of (98.7%) sites in the region were inundated with standing water. At the time, water depths were measured at three locations within each 5 m x 5 m plot. While it allowed us to calculate the mean ground elevation at the plot level, we were unable to calculate within plot microtopographic variation. Thus, to improve the accuracy, we measured water depths in all the sub-plots in 2024.

Using the field water depths measured in 2024 and water surface elevations provided by USGS's Everglades Depth Estimation Network (EDEN)) for the specific date when water depths were measured, we calculated ground elevation for each plot. The EDEN water surface elevation data were not available for 10 sites east of the C-11-canal, Thus, the analysis of hydrology data was based on only 70 sites. Hydroperiod was defined annually as the discontinuous number of days in a water year (WY: May 1 - April 30) when water level was above the ground surface.

2.3.2 Vegetation classification and change

A hierarchical agglomerative cluster analysis was performed using PCORD version 6.0 (McCune and Mefford 2011) to classify the vegetation survey-sites based on the vegetation data collected in 2022. However, to keep the vegetation identified at those sites in coherence with the classification adapted for the marl prairie vegetation encompassing all the subpopulations, the analysis also included vegetation data collected at 608 census sites surveyed in 2003-2005 within both historical (Cape Sable) and recent range (six subpopulations) of CSSS habitat. We followed the procedure described in Ross et al. (2006), i.e., we used species cover percent data, eliminated the species that were present in less than 12 sites, and relativized the species cover data by plot total. We then used the Bray-Curtis dissimilarity as our distance measure, and the flexible beta method to calculate relatedness among groups and/or individual sites (McCune and Grace 2002). Dendrograms were cut to arrive at the same ten vegetation groups that had been initially recognized based on data from 608 census sites (Ross et al. 2006).

To examine changes in vegetation composition over time, the vegetation composition data was summarized using a non-metric multidimensional scaling (NMDS) ordination. Prior to NMDS analysis, we pre-processed the species cover data. We first transformed the species' cover categorical data to percent species cover by taking mid-value of the range that each category represents. We then calculated relative frequency (%) and relative cover (%) of each species for each site. Thereafter, we calculated species' importance value (IV) as follows:

Importance Value (IV) = (Relative Frequency (%) + Relative Cover (%))/2

Species IV data was then standardized by species' maximum i.e., all IV values for a species were divided by the maximum IV attained by that species to reduce excessive influence of any dominant species in the calculation of dissimilarities (Faith et al., 1987). The site x species matrix used for the ordination had 558 sites (80 sites per survey for five surveys, 2011, 2014, 2018, 2022, and 2024, and 79 sites per survey for 2016 and 2020) and 101 species. In the analysis, only species that had a minimum of three occurrences across all surveys were retained. Thus, the final site x species matrix used for ordination had 558 sites and 66 species. In the ordination, a vector fitting technique was used to find the best fit of environmental and community variables to the species composition data (Kantvilas and Minchin, 1989). Analysis of Similarity (ANOSIM), a nonparametric multivariate analytical procedure, was used to examine the differences in vegetation composition among the survey years (Clarke et al. 2014).

Vegetation change analysis included calculation of vegetation-inferred hydroperiod, the hydroperiod for a site indicated from its vegetation composition using a Weighted Averaging regression model (see Armentano et al. 2006 for details). The analysis was performed using C_2 program, version 1.7.6 (Juggins 2014). A change in vegetation-inferred hydroperiod between successive surveys reflects the amount and direction of change in vegetation, expressed in units of days (0-365) along a gradient in hydroperiod. Additionally, vegetation response to hydrologic changes was also analyzed with trajectory analysis (Minchin et al. 2005; Sah et al. 2014), which uses a change in community composition along a vector representing hydrologic condition. In the species' IV-based NMDS ordination space, the reference vector for the hydrologic gradient was defined by the vector fitting technique in which a gradient is defined in the direction through the ordination that produces maximum correlation between the measured environmental attribute and the scores of the sampling units along the vector (Minchin 1998). The orientation of the ordination was then rotated so that 4-year average annual mean daily water depth had a perfect correlation ($r = 1.0$) with Axis-1, the ordination's principal axis. In trajectory analysis, two statistics (delta (Δ) and slope) were calculated to quantify the degree and rate of change in vegetation composition along the hydrology vector (Minchin et al. 2005; Sah et al. 2014). In this analysis, the slope was calculated as the linear regression coefficient of projected scores on the target vector in sampling years. The statistical significance of both delta (Δ) and slope was tested using Monte Carlo simulations with 1,000 permutations. The NMDS and trajectory analysis were performed using DECODA (Kantvilas and Minchin 1989; Minchin 1998).

2.3.3 Vegetation structure and biomass

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

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

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

To account for the variability caused by the repeated measures of vegetation structural variables (vegetation height, cover and biomass) and vegetation-inferred hydroperiod, Linear Mixed Models were used. General Linear Mixed Models (GLMM) were used to examine differences in structural variables between WP and M sites and among survey years, whereas Generalized Linear Mixed Models (GLMMs) were used to examine differences in species richness, a count variable. Vegetation cover and height data were square root transformed, whereas biomass and vegetation-inferred hydroperiod data were log-transformed to approximate normality. Models were run in R v.4.3.2 (R core team, 2023) using the *lmer* (for general linear mixed model) and *glmer* (for generalized linear mixed model) functions in the 'lme4' package (Bates, 2014). Sites (PlotID) were treated as a random variable. We treated sampling event (Sampyear) as a fixed effect to examine the differences in cover, height, biomass, and species richness among survey years that was done in the post hoc test using the '*glht'* function implemented in the 'multcomp' package. When General Linear Model (GLM) assumptions were not satisfied, non-parametric tests were used to analyze the data. For instance, Friedman-ANOVA (Non-parametric test for multiple dependent variables) together with Wilcoxon matched-pair test was used to test differences in 4-year average hydroperiod and mean annual water depth and percent cover as well as IV of major species among five sampling events. Spatio-temporal variation in hydrological and vegetation structural parameters was illustrated on the map using ArcGIS Pro v. 2.8.

3. Results

3.1 Hydrologic condition

In this study, analysis of hydrologic conditions at the vegetation survey sites revealed that both hydroperiod and mean annual water depth continued to increase in the post-project period i.e., after water year 2012 (Figure 5). During the last 33 water years (WY) for which hydrologic data on the EDEN website are available, both the annual mean hydroperiod and water depth were the lowest in WY2005, and the highest in WY2024. In fact, eleven out of twelve post-project water years had mean water depth higher than the long-term (33-Water year: 1991/92 – 2023/24) average (Figure 5b). In contrast, before the baseline survey in 2011, between WY1992 and WY2012, the annual mean water depths were below average in fourteen out of twenty-one years. During that period, the annual mean water depth was relatively high only in WY 1994, 1995, 1996, 1998, 2000, 2004 and 2010 (Appendix 1).

When averaged over a four year-period prior to vegetation survey, the mean (±SD) annual hydroperiod and water depth in 2011 were 221 ± 43 days and 0.21 ± 5.79 cm, respectively. However, both hydroperiod and mean annual water depth were consistently higher in post-project survey years than in 2011. The 4-year average hydroperiods were 22, 46, 42, 62, 77 and 99 days longer, and mean annual water depths were 2.94, 5.70, 5.85, 8.38, 10.15 and 14.24 cm higher during the 2014, 2016, 2018, 2020, 2022 and 2024 surveys, respectively than the pre-project period, i.e., before baseline survey in 2011 (Figure 6a, b). The differences in 4-year average hydroperiod and mean annual water depth among post-project periods were also significantly different (Wilcoxon pair-test, $p < 0.5$), and among all the surveys, both hydroperiod and water depth were the highest before the 2024 vegetation survey.

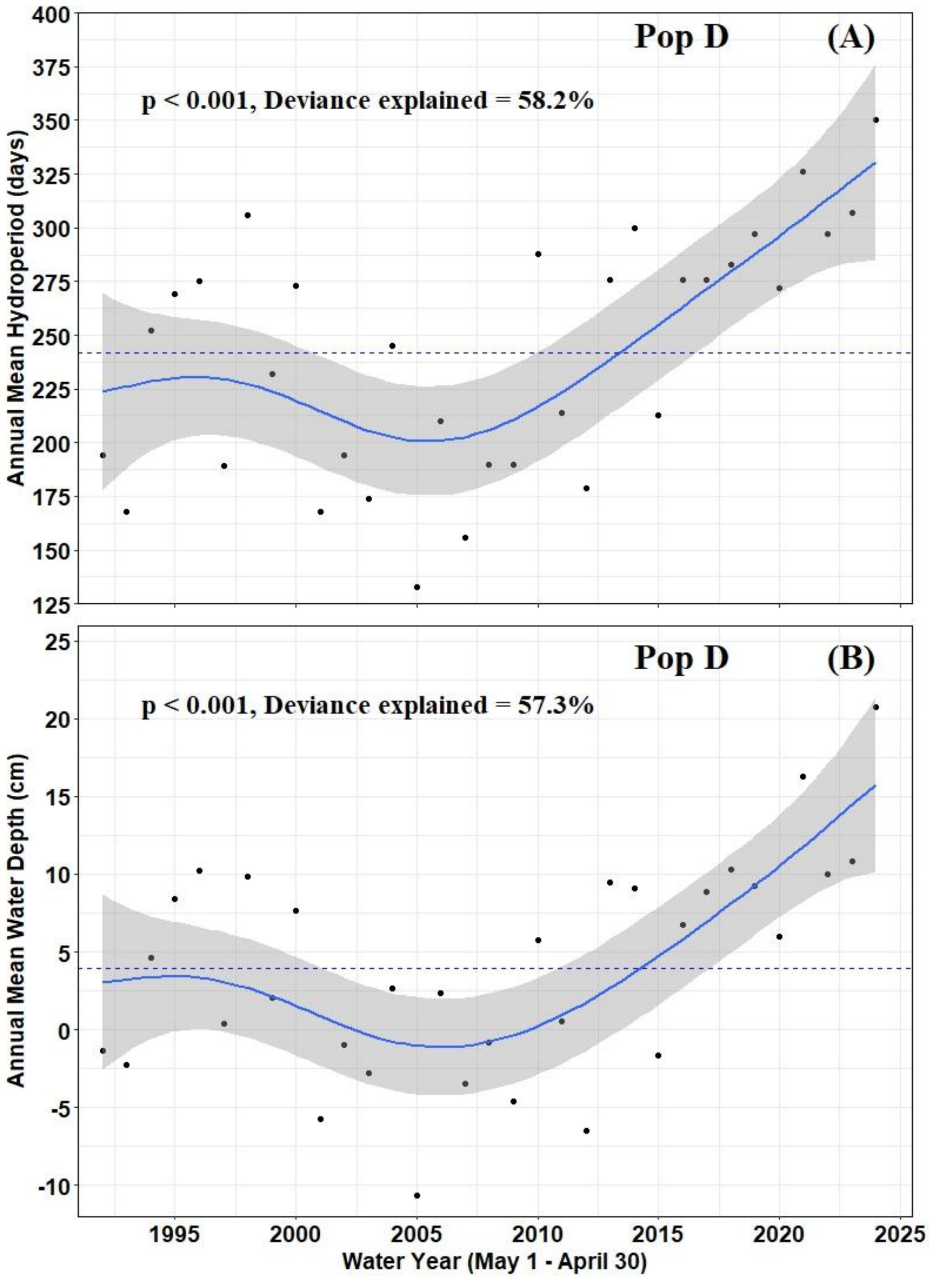

Figure 5: Trend in annual mean hydroperiod and annual mean water depth at the vegetation survey sites (n = 70) between 1992 and 2024 water years (WY): May 1 – April 30. Dashed line is the 33-year (WY1992-WY2024) average value. Hydroperiod for each site was calculated using field water depth-based ground elevation and EDEN water surface time-series data.

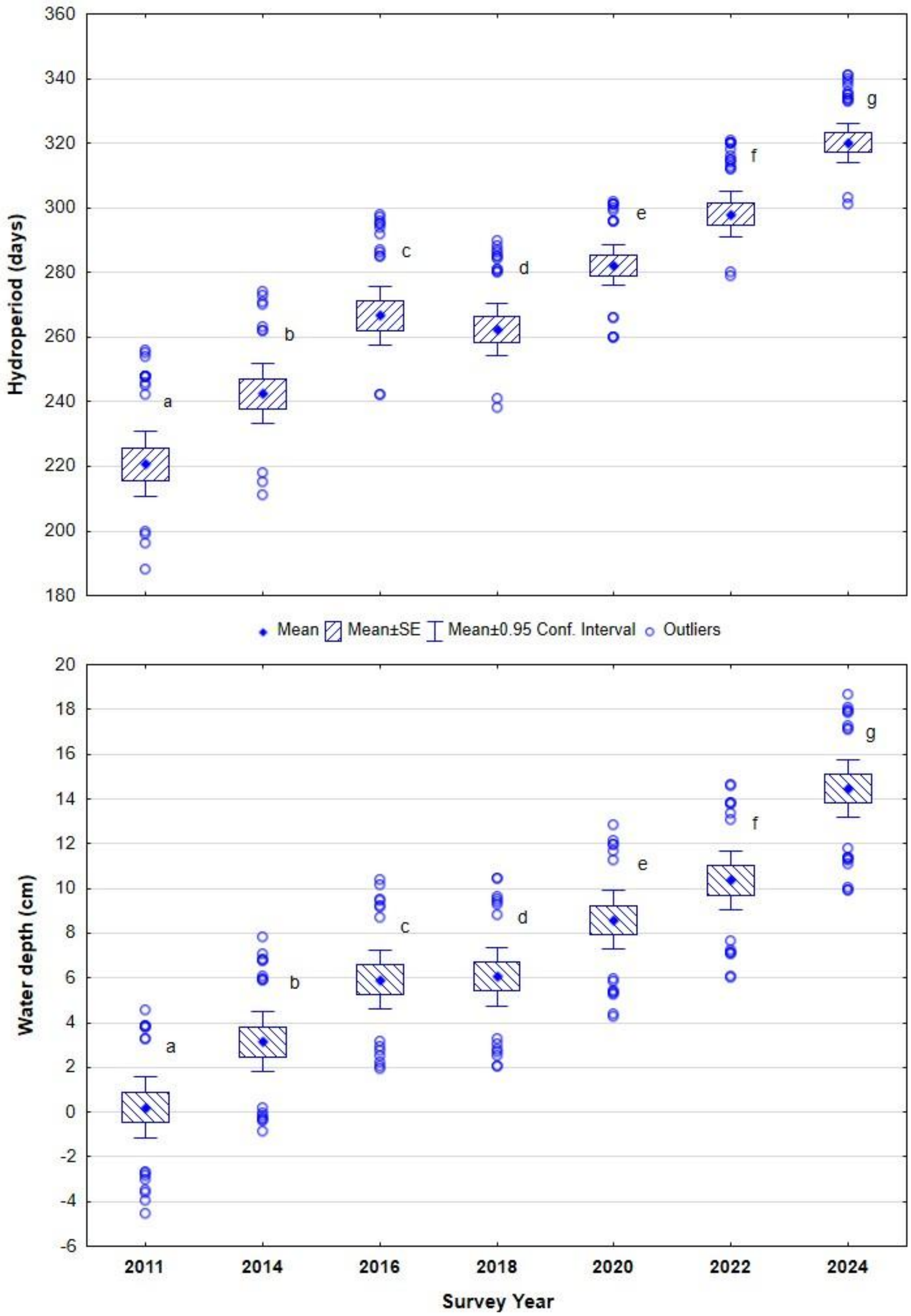

Figure 6: Four-year (four water-years prior to vegetation sampling year) average hydroperiod and mean annual water depth at the vegetation survey sites (n = 70) surveyed during the base year (2011) and in post-project survey years (2014, 2016, 2018, 2020, 2022, and 2024). Different letters in superscript represent the significant difference as determined in non-parametric, Wilcoxon-matched-pair test.

3.2 Vegetation composition

As in 2011, marl prairie vegetation within the habitat of sub-population D in 2024 was also broadly categorized into two groups, ‘wet prairie’ and ‘marsh’. Wet prairie (WP) vegetation mainly included mixed dominance of sawgrass (*Cladium jamaicense*) and/or blacktop sedge (*Schoenus nigricans*), and they were concentrated at the CS sites and western SS sites (Figure 7). However, there were much less WP sites in 2024 (20%) than that were in 2011 (65%).

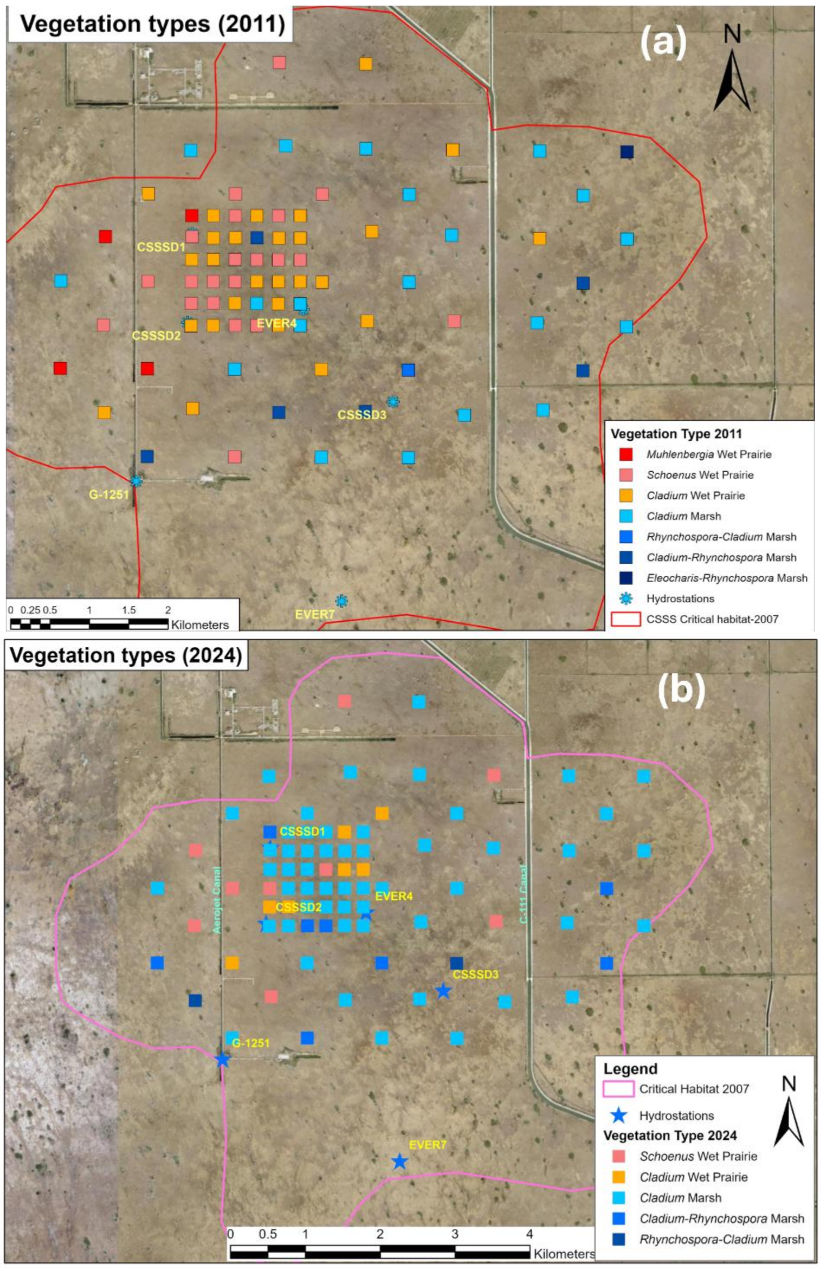

Figure 7: Vegetation types at 80 sites in the habitat of CSSS sub-population D within C-111 Spreader Canal Western Project Area. Vegetation type at each site was identified through cluster analysis of species cover values at 688 sites, including 608 census sites sampled in three years (2003-05). Vegetation types represent from dry (red) to wet (dark blue) community types and are based on (A) 2011, and (B) 2024 vegetation composition data.

In 2011, there were four sites classified as *Muhlenbergia* WP (Figure 7a), however, no sites in 2024 had high enough dominance of muhly grass (*Muhlenbergia capillaris*) to be classified as *Muhlenbergia* WP (Figure 7B). Across all sites, the mean cover of muhly grass decreased from 3.25% in 2011 to 0.18% in 2024. During the 2024 survey, marsh (M) vegetation was present at 80% of total sites, and those sites had 4-year average hydroperiods of ≥270 days. The vegetation assemblages at the majority (84%) of marsh sites were sawgrass (*C. jamaicense*) marsh type. The remaining 16% of sites had sawgrass-beakrush sedge (*Cladium-Rhynchospora*) or beakrush-sawgrass (*Rhynchospora-Cladium*) marsh (Figure 7b).

In NMDS ordination, the first axis, which was aligned to parallel the fitted vector of 4-year average mean annual water depth, separates the marsh sites from wet prairie sites (Figure 8), suggesting that species composition along the gradient is primarily influenced by hydrology (hydroperiod: $r = 0.76$, $p < 0.001$; mean annual water depth: $r = 0.80$, $p < 0.001$) (Table 1). Soil depth as well as three community characteristics – species richness, vegetation cover, and aboveground biomass - were also significantly correlated. Moreover, species composition in all post-project survey years was significantly different (ANOSIM: p-value < 0.001) from that in 2011 (Table 2). While vegetation composition in 2022 differed from that in 2014, 2016, 2018 and 2020, ANOSIM results suggest that the difference in vegetation composition between 2018 and 2020 and between 2022 and 2024 was not statistically significant.

Table 1: Maximum correlations (r) of significant environmental and community characteristic vectors fitted in the 3-dimensional NMDS ordination for plant species' importance value (IV) data. Probabilities (P) were calculated using 10,000 random permutations.

Environment and Community Variables	N	r	p-value
Soil Depth (cm)	80	0.5258	**<0.001**
4-Yr average Hydroperiod (Days)	481	0.7640	**<0.001**
4-Yr average water depth (cm)	481	0.7992	**<0.001**
Species richness	558	0.7462	**<0.001**
Vegetation Cover (%)	558	0.3288	**<0.001**
Biomass (g m^2)	558	0.4957	**<0.001**

Table 2: Global R and *p*-values from analysis of similarity (ANOSIM) testing for among-year differences in vegetation composition before (2011) and after (2014, 2016, 2018, 2020, 2022 and 2024) the operation of the C-111 spreader canal western project that began in 2012. *p-value: * <0.5, ** < 0.01, *** <0.001*.

Sampling event	2011 (base line survey)	2014	2016	2018	2020	2022
2014	0.083***					
2016	0.204***	0.136***				
2018	0.226***	0.106***	0.110***			
2020	0.191***	0.115***	0.094***	0.007^{ns}		
2022	0.140***	0.130***	0.109***	0.055***	0.026**	
2024	0.203***	0.175***	0.121***	0.056***	0.022**	0.011^{ns}

Vegetation changes over thirteen years, since the base line survey in 2011, was marked by an increase in wetness of many sites and a consequent shift in species composition toward a wetter type. In the species' importance value (IV)-based NMDS ordination space, while 2011 wet prairie sites, represented by a centroid, showed a noticeable shift in position towards increasing wetness, marsh sites did not show much shift in species composition over thirteen years along the hydrologic gradient (Figure 8).

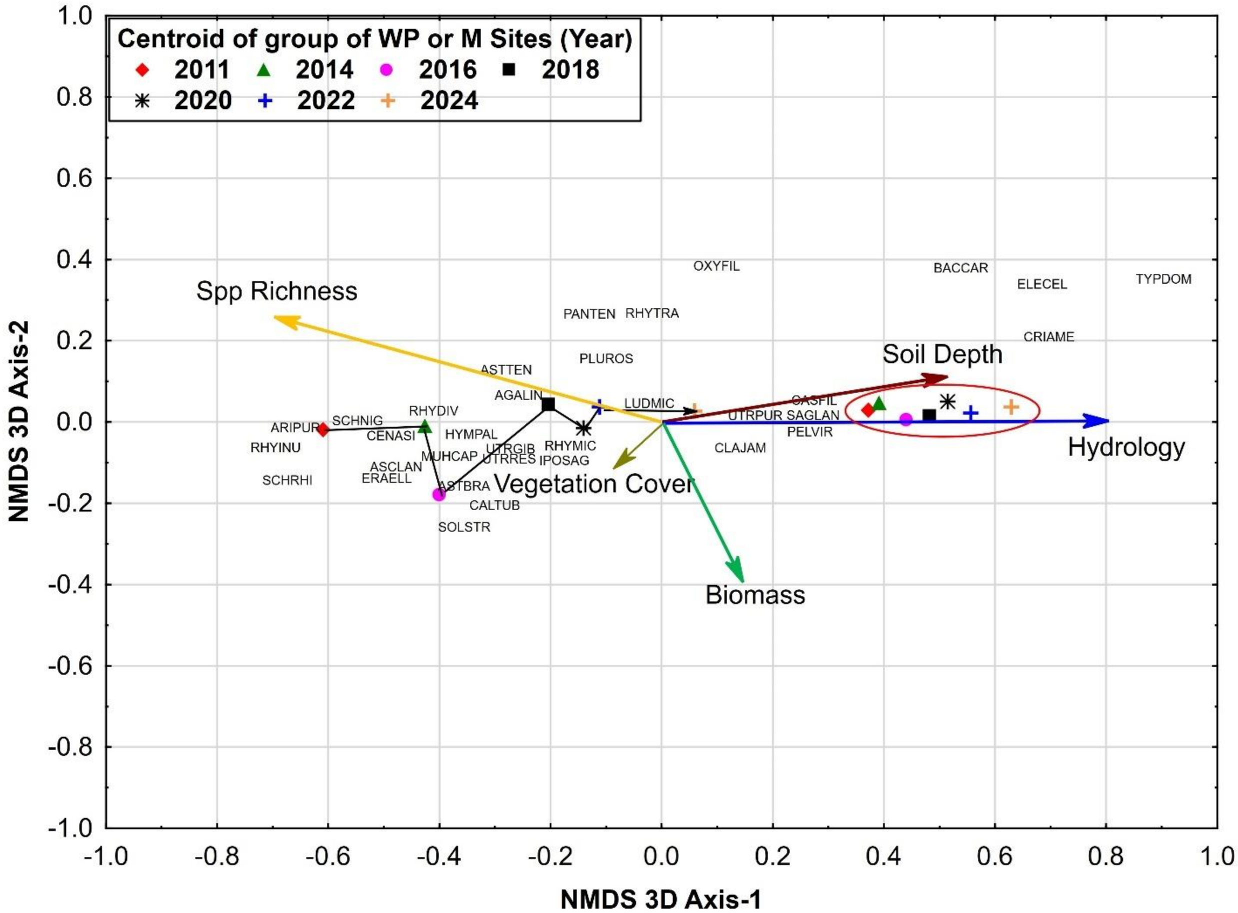

Figure 8: Site scores from species' importance value (IV) based non-metric multidimensional scaling (NMDS) ordination Axis-1 and 2. Points in the ordination space represent centroids of sites grouped by major vegetation category (Wet praire (WP) and Marsh (M)), as sites were classified in 2011, and sampling year (2011, 2014, 2016, 2018, 2020, 2022 and 2024). Centroids of Marsh sites are circled by an oval. Only selected species are plotted to reduce the overlap. Full name of species are given in Appendix 2.

Thirty-six (69.23%) wet-prairie sites of 2011 were classified as marsh sites based on species abundance data collected in 2024 (Appendix 2). In contrast, all of 28 marsh sites from 2011 still had marsh vegetation in 2024. Most of them had *Cladium* Marsh vegetation type, and that stayed the same for the next thirteen years. Trajectory analysis results also revealed that between 2011 and 2024, vegetation composition at 71 (88.75%) sites had shifted toward wetter type (represented by positive delta and slope), and this shift toward a wetter type was statistically significant ($p = < 0.1$) at 45 (63%) sites – 34 wet prairie and 11 marsh sites (Figure 9; Appendix

2). In contrast, vegetation composition at only 7 sites, including five marsh sites, mostly located east of C111 canal, showed a drying trend. Of those seven sites, the shift toward a drier type was significant (delta: p-value = <0.1) at only three (43.5%) sites.

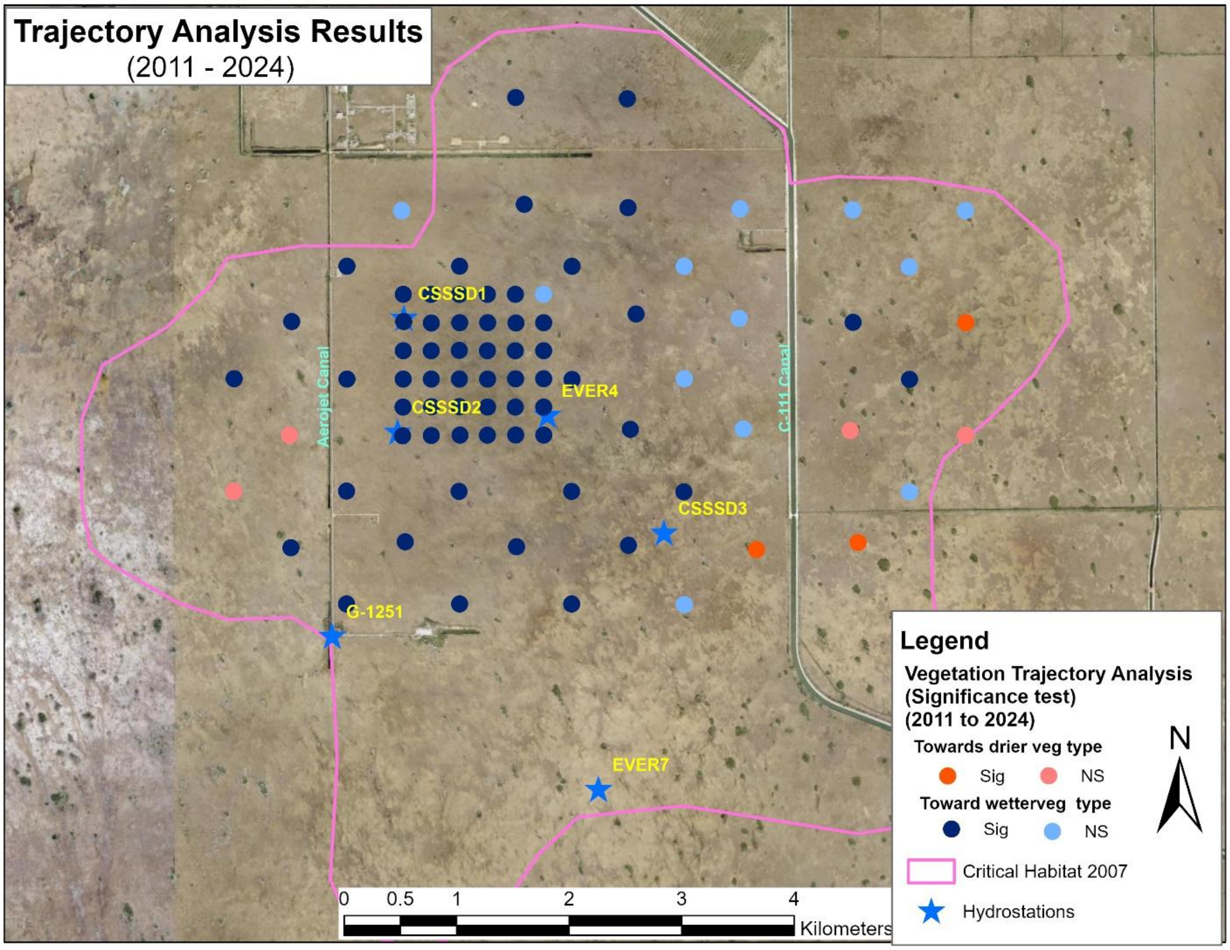

Figure 9: Sites showing a significant shift in vegetation composition between 2011 and 2024 surveys in the habitat of CSSS sub-population D. Significance of site trajectory was obtained by trajectory analysis.

Over thirteen years (2011-2024), 101 species were recorded from the study area (Appendix 3). During that period, the cover value of major species (*Muhlenbergia capillaris* ssp. *filipes*, *Schoenus nigricans* and *Rhynchospora microcarpa*) that are characteristic of marl wet prairie sites, i.e., the dry end of the marl prairie hydrologic gradient, significantly declined (Figure 10a, Appendix 4). The mean cover of muhly grass in 2024 was only one- eighteenth of its cover value in 2011 while the mean cover of blacktop sedge in 2024 was one-fourth of its cover value in 2011. In contrast, differences in mean cover of spikerush (*Eleocharis cellulosa*), which was most abundant at the wet end of the hydrologic gradient in marl prairies (Ross et al. 2006; Sah et al. 2011a), was not statistically significant (Appendix 4). Mean cover of sawgrass (*Cladium jamaicense*) decreased by one-third in 2014, three years after the base surveyed year, but cover then remained the same in the next four years (until 2018 survey). In concurrence with an increase in both four-year average hydroperiod and four-year average water depth between 2018 and 2022 surveys, sawgrass mean cover also significantly (Wilcoxon Matched Pair-test: p <0.001) increased from 23.9±14.1% to 35.9±22.2% during that period, but then remained the same (p = 0.078) in 2024 (Appendix 4). The cover of beakrush sedge (*Rhynchospora tracyi*) varied over time. In 2024, its cover was the lowest since 2011 but was similar to its cover value in 2016 (Appendix 4).

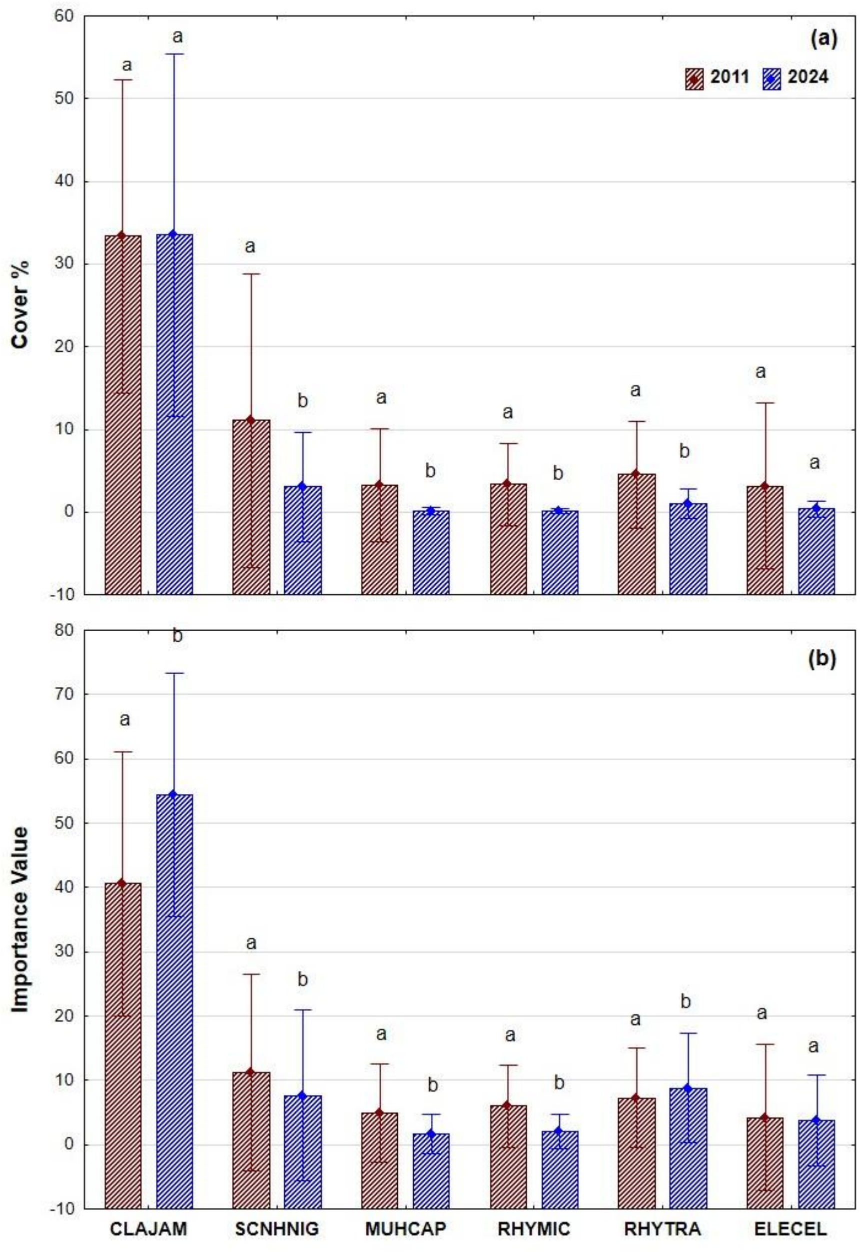

Figure 10: Bar diagram **(**Mean ± 1 S.D.) of percent cover and importance value (IV) of major species averaged over all sites (n = 80) surveyed in 2011 and 2024 within the CSSS sub-population D habitat region. P-values are from non-parametric test, Friedman Analysis of Variance for multiple dependent samples. Different letters in superscript represent the significant difference as determined in non-parametric, Wilcoxon matched-pair test. Full name of species are given in Appendix 3.

Together with the cover value, the importance value (IV) of the species that are characteristic of relatively dry communities, also decreased over time, and IV of *M. capillaris*, *S. nigricans* and *R. microcarpa* in 2024 were significantly lower in 2024 than in 2011 (Figure 10b: Appendix 4). However, the importance values of marsh species were either the same, e.g., IV of spikerush (*E. cellulosa*), or significantly increased, as were the IV of *C. jamaicense* and *R. tracyi* (34% and 20.5%, respectively), suggesting a shift in species composition toward a wetter type.

3.3 Vegetation-inferred hydroperiod

Observed- and vegetation-inferred hydroperiods were well correlated even when data were pooled for all seven sampling years ($r = 0.65$, $p < 0.001$). However, across all the years, the values of inferred hydroperiod were consistently lower than the values of 4-year average hydroperiod. In concurrence with the wetter conditions during the six project-period surveys than the base line survey, the mean (± SD) vegetation-inferred hydroperiod was also significantly (General Linear Mixed Model: Tukey's test, $p < 0.05$) higher in all five post-project surveys than in 2011 (210 ± 47 days) (Figure 11). However, there was no significant difference in vegetation-inferred hydroperiod between 2014 and 2016 or 2018, suggesting that a prevalence of wet conditions during the project period caused a shift in species composition toward a more hydric type, primarily in the first three years after the baseline survey. After 2018, the trend in vegetation changes towards more hydric type continued for the next six years. While there was significant increase (General Linear Mixed Model: Tukey's test, $p = 0.04$) in inferred-hydroperiod between 2018 and 2020, the wetting trend was slow for the next two surveys, as there was no significant difference in vegetation-inferred hydroperiod between 2020, 2022 and 2024. Consistent with the 4-year average hydroperiod (Figure 6), the mean vegetation-inferred hydroperiod was also higher in 2020, 2022 and 2024 (229 ± 38 days, 229±35 and 233 ± 34 days, respectively) than in the first four (2011, 2014, 2016 and 2018) survey years (Figure 11).

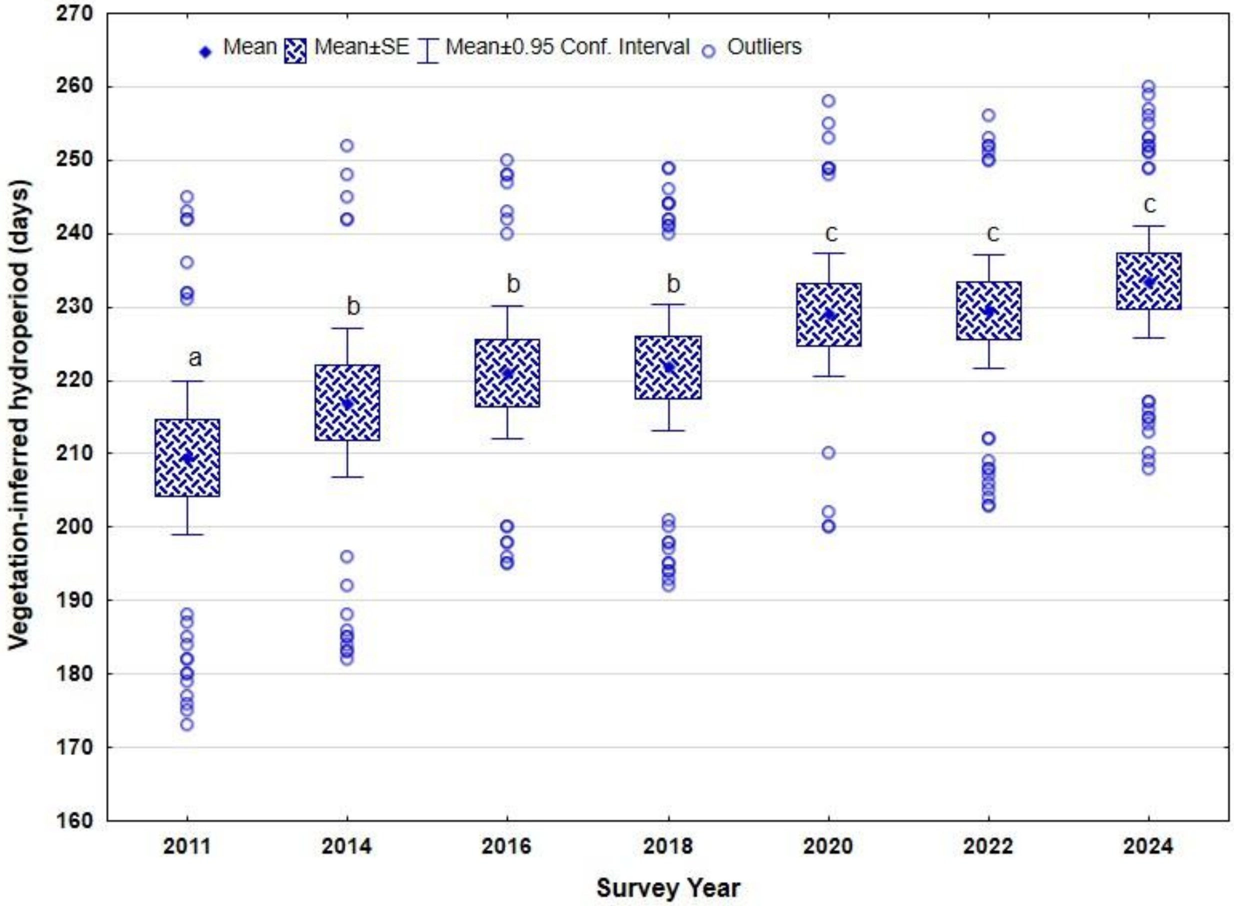

Figure 11: Box-plots (Mean, SE, and mean±1.96*SE) showing vegetation-inferred hydroperiod in different survey years within the habitat of CSSS sub-population D. Vegetation-inferred hydroperiod values were predicted from vegetation composition using Weighted Averaging regression model developed from the vegetation and hydrology data from CSSS vegetation transect D (Ross et al. 2006). Different letters above the whisker represent significant difference (General Linear Mixed Model – Tukey's test with bonferroni correction, $p < 0.05$)

3.4 Vegetation structure and biomass

Vegetation changes over thirteen years (six surveys after the baseline survey in 2011) were also marked by changes in vegetation structure (vegetation cover and height), species richness and aboveground biomass (Figure 12). Change in mean (±SD) vegetation cover varied between surveys, and the cover was significantly lower (General Linear Mixed Model: Tukey's test, $p < 0.05$) during the first post-project survey (2014) (32.6 ± 12.7%) than in 2011 (39.3 ± 17.2%) (Appendix 5). The mean vegetation covers then did not change much until the 2020 survey, as the differences in mean vegetation cover among the first four post-project surveys (2014, 2016, 2018 and 2020) were not significant (General Linear Mixed Model: Tukey's test, $p > 0.05$). However, between 2020 and 2022 surveys, mean vegetation cover significantly increased but did not change much between the last two surveys, 2022 and 2024. Across the survey years, the 2011 wet prairie and marsh sites showed different patterns in changes in vegetation cover. For instance, the sites that were wet prairie in 2011 did not show significant change in vegetation cover between 2011 and 2024, even though 69% of those sites had changed to marsh types over the years. In contrast, marsh sites had higher vegetation cover in 2024 than in 2011 (although not significant), even though those sites remained marsh sites over the years.

In comparison to cover, vegetation height consistently increased over thirteen years. The mean vegetation height was significantly higher in 2018 (61.0 ± 15.5 cm), 2020 (69.4 ± 14.7 cm) and 2022 (69.0 ± 14.6 cm) than the pre-project survey year (2011: 52.9 ± 14.1 cm), whereas the mean height during the first two post-project surveys, 2014 (57.2 ± 11.4) and 2016 (56.4 ± 12.5 cm), were intermediate. In 2024, mean vegetation height slightly decreased (2024: 59.2 ± 15.7) but did not differ significantly from the three previous years (Appendix 5). In the post-project period, the vegetation height primarily increased at only marl wet prairie sites, whereas at the marsh sites, the vegetation height was the same until 2016 but has increased only in recent years (Appendix 5). Mean vegetation height at both 2011 prairie and marsh sites was significantly higher in 2024 than the height during the baseline survey in 2011 (Figure 12B).

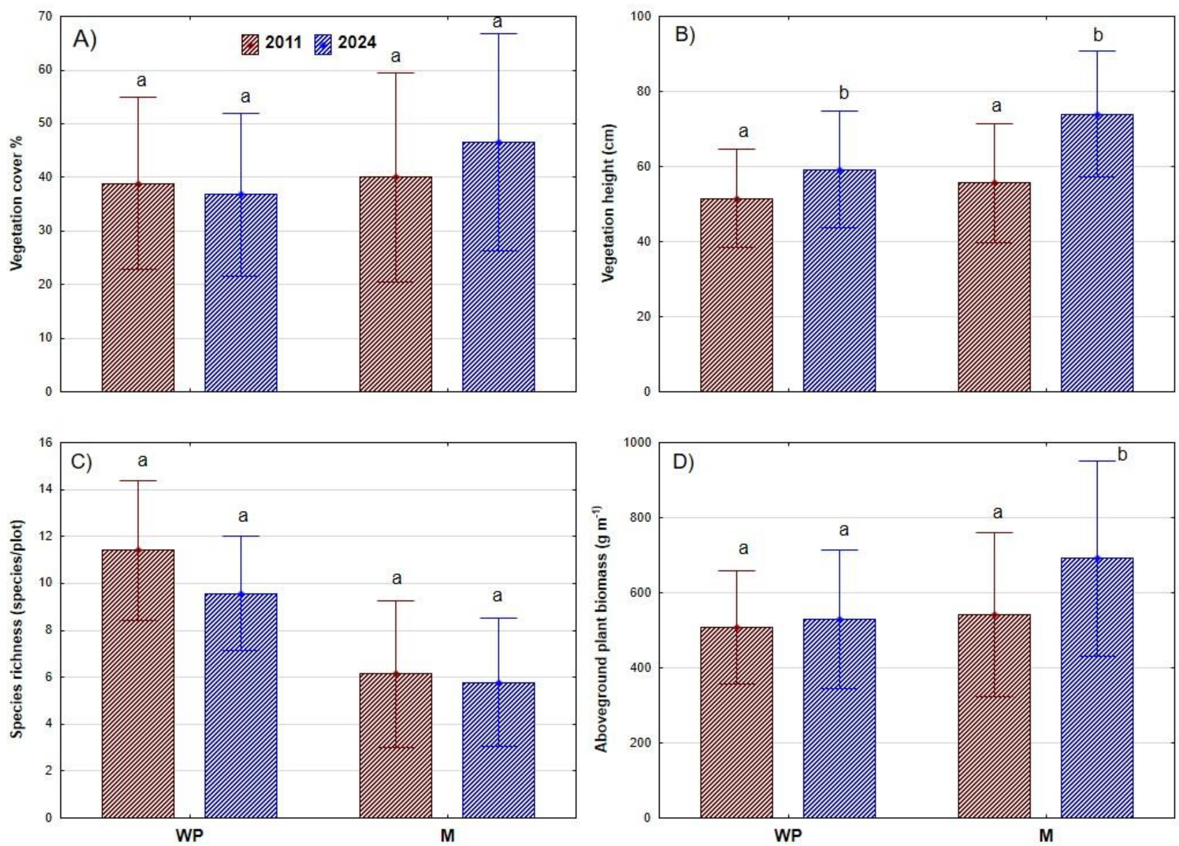

Figure 12: Box-plots (mean, SE, 95% CI) showing the vegetation structure, (a) vegetation cover, (b) vegetation height, (c) species richness, and (d) aboveground biomass in the 2011 baseline survey, and the 2024 survey (n = 80). Different letters represent the significant difference as determined in post-hoc (Tukey's) test using "multcomp" package in R.

Across the survey years, plant species richness was significantly lower at marsh sites than at wet prairie sites (Generalized Linear Mixed Model: $p < 0.001$; Figure 12c; Appendix 5). However, over the survey period (2011-2024), mean plant species richness has not changed much, though the species richness was the lowest in 2018, two years after the year (2016) when sites had remained flooded during the dry season. The mean species richness in three recent surveys, 2020, 2022 and 2024, also did not differ from the richness in any previous survey years, including the baseline survey in 2011 (Appendix 5; Figure 12c).

The aboveground biomass was relatively low in 2014 through 2018 (e.g., 480±137, 490±149, and 467±140 g m^{-1} in 2014, 2016 and 2018, respectively). Thereafter, mean biomass increased with an increase in wetness. However, while above ground biomass at wet prairie sites decreased with further increase in wetness between 2022 and 2024, mean biomass at the marsh sites consistently increased until the most recent survey. At the wet prairie sites, the mean aboveground biomass during the 2024 survey was not significantly different from that in 2011. In contrast, the mean biomass at the marsh sites was significantly higher (27.7%) than the biomass at those sites in 2011 (Figure 12d; Appendix 5).

The observed changes in vegetation structure (cover and height), species richness and aboveground biomass over thirteen years (2011-2024) varied spatially in the study area (Appendix: Figure A-1a, b & Figure A-2c, d). During the 2024 survey, sites in the central portion of study area (CS), where marl wet prairie vegetation types are still dominant, usually had high species richness, medium cover, and medium biomass. In contrast, the sparse sites (SS), especially those located east of C111 and the southern portion of the sub-population D habitat, where marsh vegetation is dominant, had lower species richness than the rest of the area.

4. Discussion

In the Everglades, the marl prairie is a dynamic landscape system where hydrology and fire are important drivers. In this system, vegetation responses to hydrologic alterations may occur rapidly, usually within 3-5 year (Armentano et al. 2006), consequently affecting the CSSS habitat and the sparrow population (Nott et al. 1999; Jenkins et al. 2003). Within the habitat of sub-population D, vegetation has gone through different episodes of change over the past three decades, primarily in response to natural and anthropogenic alterations in hydrologic regimes (Sah et al 2011, 2022). While the vegetation in 1981 was mostly the marl wet prairie type (Pimm et al. 2022), during the early 1990s the vegetation had changed to a sawgrass-dominated marsh type, primarily in response to prolonged hydroperiod and high-water conditions in the area (Ross et al. 2004). These conditions resulted from both high rainfall during the mid-1990s and an increased water delivery into Taylor Slough through the operations of the S-332 pump station (Ross et al. 2004; Armentano et al. 2006). Consequently, the sparrow population sharply declined (Pimm et al. 2002). Marsh vegetation prevailed until the late 1990s, and the sparrow population dropped from sight, as no sparrow was recorded for three consecutive years (2002-2004). During the next few years, the vegetation within the region showed a drying trend, primarily in response to several drought years (Sah et al. 2011a). Consequently, the wet prairie vegetation was more widely spread in 2011 than it was during the period of 2003-2006 when a detailed systematic vegetation survey was first conducted at a network of sites located 1 km apart (Ross et al. 2006; Sah et al. 2011b). Since the baseline survey in 2011, i.e., prior to the implementation of the C-111 Spreader Canal Western project, vegetation composition has shifted back toward a wetter type. For instance, sixty-nine percent of 2011 marl wet prairie sites have changed to relatively wet marl marsh vegetation types in 2024 (Appendix 2). This shift in vegetation might have implications on sparrow occupancy within the area.

In the marl prairies, species richness is negatively correlated with hydroperiod (Ross et al. 2006). Thus, a low richness in three out of six sampling years after the baseline survey was not a surprise, especially when vegetation composition has shifted toward much wetter types in thirteen years. However, the fact that species richness in 2016, 2020 and 2022 was similar to that in 2011 was unexpected. The reason for high species richness in 2016 could be due to the prolonged dry period in 2014 and early 2015, one year prior to the 2016 sampling. In 2016, there was also a high variation in the occurrence of species at the wet prairie sites (Appendix 3). Many of the sites in that particular year had characteristic species from both marl wet prairie and marsh vegetation types, especially due to unusually high-water conditions in the dry season that occurred after a prolonged dry period in 2014-2015. Many species that are usually found at the marl marsh sites, such as *Eleocharis interstincta*, *Ludwigia alata*, *L. curtissii*, *L. repens*, *Utricularia purpurea*, *U. resupinata*, and *U. subulata*, were recorded for the first time in 2016. Nonetheless, by 2018, in conjunction with a change in vegetation composition from wet prairie to marsh types, species richness also declined. In 2018, maidencane (*Panicum hemitomon*), a characteristic species of wet conditions in Everglades was recorded for the first time. An increase in species richness between 2018 and 2020 was possibly the result of alternating dry and wet conditions. While wet conditions observed in 2016 and 2018 continued until 2019, WY 2019/2020, which included the early part of the 2020 survey, was relatively dry. In addition, several sites burned in early 2020 where some ephemeral marsh species were recorded. For instance, Piedmont marshelder (*Iva microcephala*) was recorded for the first time in 2020 from the vegetation survey sites in sub-population D. Even though the plant species richness has shown high variability during the study period (2011-2024), if the marl wet prairie vegetation composition continues to shift towards a wetter type, it is likely that plant species richness in subpopulation D will also decline over time.

A shift in marl prairie vegetation towards wetter types is perceived as deterioration in the available sparrow habitat quality. The foundation for this belief lies in the fact that sparrow occurrence is usually highest in muhly-dominated wet prairie with hydroperiods ranging between 90 and 210 days; concurrently, CSSS occurrence is less frequent in wetter vegetation types ranging from sawgrass-dominated prairie and marsh to beakrush sedge (*Rhynchospora tracyi*) and spikerush (*Eleocharis* sp.) marsh (Nott et al. 1998; Ross et al. 2006; Bencoster and Romañach 2022). In sub-population A, west of Shark River Slough, researchers also attributed a sharp decline in the sparrow population to severe and prolonged flooding in the mid-1990s and the consequent change in vegetation to sawgrass marsh (Nott et al. 1998; Pimm et al. 2002; Jenkins et al. 2003). In Sub-population D too, the sparrow population sharply declined after the first survey in 1981, probably for the same reason (Pimm et al. 2002). However, within this sub-population, a small breeding population of sparrows has consistently been recorded since 2006 by Julie Lockwood (2006-2010) and Tom Virzi (2011-2020) from Rutgers University (Lockwood et al. 2006, 2010; Virzi et al. 2011, 2015; Virzi and Davis 2013, 2014, 2016, 2017; Virzi and Murphy 2018; Virzi and Tafoya 2019, 2020). The bird nests were generally found within an area of high ground in the northwest-central region of subpopulation D (Virzi and Davis 2013, 2014, 2016; Virzi et al. 2015; Virzi and Tafoya 2019, 2020), where ground elevation is relatively high, and WP vegetation was dominant in 2011 (Figure 7a).

In 2013, Virzi and Davis reported that the total extent of occupied habitat was shrinking each year, and they wondered if the decline was in response to changes in vegetation conditions. An analysis of 2014 data had also shown that at the WP sites, the increase in mean vegetation-inferred hydroperiod was 11 days between 2011 and 2014, and such an increase was disproportionately higher than the increase in inferred-hydroperiod at marsh sites (Sah and Ross 2014). Between 2014 and 2018, however, the change in vegetation-inferred hydroperiod was not statistically significant (Figure 11), suggesting that after 2014, the habitat condition showed a wetting trend but at a much slower pace. In fact, WY2015 was drier than average (Figure 5), and within the ENP basin total rainfall in WY2015 was 10.54% less than the long-term average rainfall (Abtew and Ciuca 2016), much less (15.5%) than the average during the 2015 wet season. This prolonged dry condition might have temporarily reversed the trend of change in vegetation composition and helped to improve habitat conditions. This was evident by an increase in the ephemeral sparrow population in both 2014 and 2015, which was attributed to the extended favorable breeding season (Virzi and Davis, 2014; Virzi et al. 2015). The sparrow data from 2016 was incomplete, but in 2017, the sparrow population was slightly lower than in 2014 and 2015 (Virzi and Davis 2017). Thereafter, there was a consistent increase in sparrow numbers for the next two years, i.e., until 2020, when the last detailed survey of sparrows in sub-population D was done (Virzi and Tafoya 2020). A dramatic increase in sparrow numbers between 2017 and 2022 was also recorded in the Park's helicopter survey. In 2017, only 4 sparrows (representing a total population of 64 sparrows) were counted that were present at two sites, whereas in the 2021 and 2022 surveys, 18 and 24 sparrows (i.e., population of 288 and 384) from 7 and 12 sites, respectively were recorded. However, the number of birds recorded during both the 2023 and 2024 sparrow surveys was 45.8% less than the number of birds recorded in 2022 (*Source*: ENP). Moreover, among the 11 bird census sites where sparrow occurred in 2023 and/or 2024 and are within the distance of 250 m from the sub-population D vegetation survey sites, 5 (i.e., 45.5%) sites had wet prairie vegetation type in 2011 and remained so until 2024. Moreover, three (27.2%) sites had WP vegetation type in 2011, but they changed to marsh types in recent years (Appendix Figure A-4). Thus, while CSSS continue to nest successfully in the small subset of wet prairie that remains in sub-population D when local conditions are favorable, these small patches of prairie are dwindling as water levels rise throughout the region.

In the Everglades marl prairies and ridge & slough landscapes, the hydrology-mediated change in vegetation composition is usually visible within 3-5 years (Armentano et al. 2006; Zweig and Kitchens 2008; Sah et al. 2014). However, the lag time could be longer depending on the pattern and magnitude of hydrologic changes, including annual variability in hydrologic regime. In addition, the unusually extreme hydrologic condition may also disrupt the vegetation trajectories. In general, extreme weather events, such as tropical storms, cold events, flooding, and drought, are well recognized as critical drivers of vegetation change in different ecosystems (Allen and Breshears 1998; John et al. 2013; Copeland et al. 2016), including those in South Florida (Miao et al. 2009; Ross et al. 2009). In South Florida, rain events are closely associated with El Nino-Southern Oscillation (ENSO) (Moses et al. 2013). In the WY2016, 2021and 2023 dry seasons, rainfall was higher than the historical seasonal average (Abtew and Ciuca, 2017; Cortez et al., 2022; Cortez, 2024), resulting in high water conditions throughout South Florida, including southern Everglades. In a normal year, water level in the eastern marl prairies drops to 100 cm below the ground during the dry season (Sah et al. 2011b). Conversely, during the WY2016 dry season, the mean water level at vegetation survey sites was 11.8 cm above the ground, which was 11.2 cm higher than the 33-year average of dry season water depths. The water level in the 2016 dry season was high enough to shorten the sparrow study period in that subpopulation (Virzi and Davis 2016).

In the past, unusual high-water conditions during the sparrow reeding season not only caused a crash in sparrow populations, e.g., sub-population A, but also contributed to a vegetation shift from muhly- or bluestem-dominated marl wet prairies to sawgrass-dominated marshes (Pimm et al. 2002; Nott et al. 1998). In the case of sub-population A, however, high water conditions in the area continued for the next 2-3 years, due to both high rainfall and water deliveries through the S12 structures. Thus, unusual dry season flooding followed by higher water level than normal for multiple years was the major cause of habitat degradation within the western marl prairies (Nott et al. 1998; Jenkins et al. 2003). For similar reasons, a decline in the sparrow population and a shift in vegetation composition also occurred in sub-population D (Pimm et al 2002; Ross et al. 2004; Virzi et al. 2011). In this area, the mean annual as well as dry season water depth for the next eight water years (2017-2024) was higher than the 33-year annual and dry season average water depths, respectively. In addition, the 4-year average hydroperiod was significantly higher during the 2020, 2022 and 2024 surveys than in any previous surveys. That might have accelerated the vegetation composition to shift towards a wetter type than it was in 2018 and before. In fact, in 2020, 2022 and 2024 the vegetation-inferred hydroperiod, a metric that has been used to track the shift in vegetation composition in response to hydrologic changes, was significantly higher than in any previous surveys. If the trend continues, it will have an adverse impact on CSSS habitat quality, and ultimately, the sparrow population in the area.

Prior to the implementation of the C111-SC project, a simulation model to assess the potential impacts of the project on habitat conditions indicated that the operations would result in an increased hydroperiod and might have adverse effects on the habitat conditions within the CSSS subpopulation D critical habitat (USFWS 2009). Thus, during the 12-year post-project period, a shift in vegetation composition in response to hydrologic changes towards wetter types that we observed in the sub-population D area is consistent with the model expectations. In general, an increasing trend in wetness in marl prairies beyond 210 days hydroperiod is envisioned as gradual deterioration of sparrow breeding habitat conditions (USFWS 2016). Thus, the increase in 4-year average vegetation-inferred hydroperiods from 210 days in 2011 to 229 days in 2020 and 2022 and 233 days in 2024 (Figure 11) observed during the post-project period could be an indication of deteriorating CSSS habitat condition. However, more successful sparrow nesting during the breeding season in the during 2018-2020 than in previous years (Virzi and Murphy 2018; Virzi and Tafoya 2019, 2020), and relatively high population in 2021 and 2022 (Park's helicopter survey) is contrary to our ecologically-based expectations. In 2020, almost all sparrows' nests and the majority of sparrows were observed within the core area where prairie vegetation is still dominant (Appendix: Figure A-3). However, eleven pairs were detected outside the core area, designated during the baseline survey. Several factors, including the favorable dry seasons, low dispersal barrier, effect of Hurricane Irma on dispersal patterns and habitat quality, etc. might have played important roles in the successful breeding and relative high number of sparrows in sub-population D in those years, but the subpopulation remains small and vulnerable (Virzi and Tafoya 2020). Contrary to the existing view that sparrow occurrences decline with increases in hydroperiod beyond 240 days, the growing number of sparrows in sub-population D needed a more in-depth study of sparrow behavior in relation to habitat conditions within this area. A recent decline in the sparrow population during the last two surveys (2023 and 2024) from the peak in the population in 2022 suggests that the need of more in-depth study is warranted.

Within the habitat of sub-population D, an ongoing wetting trend together with the shift in vegetation from short-hydroperiod marl wet prairies to marsh types may eventually have adverse effects on sparrow success. Thus, it is important to minimize the chances of high-water condition in the coming years, especially in the dry season, so that the observed trend of vegetation shift will not accelerate. Only a continued monitoring of the vegetation as well as sparrow population dynamics can provide a conclusive assessment of the ongoing trend of vegetation shift, probably caused by the synergistic effects of high rainfall and the project activities on the future fate of the existing CSSS population and its habitat. Moreover, the trajectory analysis method used in this study has made it possible to detect a shift in vegetation composition along a vector representative of increasing wetness. This demonstrates that a more sensitive tool based on plant assemblages is available for tracking the outcome of water management decisions on sparrow habitat quality in this sub-population.

Acknowledgements

We would like to acknowledge the assistance in field and lab works provided by the following members of South Florida Terrestrial Ecology Lab (SOFTEL) – Pablo Ruiz, Jesus Blanco, Junio Freixa, Allison Jirout, Alex Martinez-Held, Katherine Castrillon, Rosario Vidales, Christian Prieto and Juliana Alvarez. We would also like to thank Eric Cline for arranging the SFWMD helicopter for field surveys, and Gary Hansen for flying us to the field sites. We would like to thank Field Operation Center (FOC) at Institute of Environment, FIU for providing us the logistic support for fieldwork. The project received financial support from the South Florida Water Management District (PO # 4500147501).

References

Abtew, W. and Ciuca, V. (2016). *South Florida Hydrology and Water Management.* In: 2016 South Florida Environmental Report Volume 1. South Florida Water Management District (SFWMD).

Abtew, W. and Ciuca, V. (2017). *South Florida Hydrology and Water Management*. In: 2017 South Florida Environmental Report Volume 1. South Florida Water Management District (SFWMD).

Allen, C. D. and Breshears, D. D. (1998) Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences, USA*, **95**: 14839–14892.

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.

Bates, D., Mächler, M., Bolker, B. and Walker, S. (2015) Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software* **67**(1): 1–48.

Benscoter, A.M. and Romañach, S.S. (2022) Endangered Cape Sable seaside sparrow ecology: actions towards recovery through landscape-scale ecosystem restoration. Endangered Species Research **49**: 199-215.

Clarke, K.R., Gorley, R.N., Somerfield, P.J. and Warwick, R.M. (2014) Change in marine communities: an approach to statistical analysis and interpretation, 3rd edition. PRIMER-E:Plymouth.

Copeland, S. M., Harrison, S. P., Latimer, A. M. et al. (2016) Ecological effects of extreme drought on Californian herbaceous plant communities. *Ecological Monographs* **56** (3): 295-311.

Cortez, N. A. (2024). *Chapter 2A: South Florida Hydrology and Water Management*. In: 2024 South Florida Environmental Report, Vol. 1, pp. 2A-1 to 2A-59. South Florida Water Management District, West Palm Beach, FL

Cortez, N. A., Qiu, C. and Ciuca, V. (2022). *Chapter 2*: *South Florida Hydrology and Water Management*. In: 2022 The South Florida Environmental Report, Vol. 1, pp. 2A-1 to 2A-46. South Florida Water Management District, West Palm Beach, FL.

Faith, D. P., Minchin, P. R., Belbin, L. (1987) Compositional dissimilarity as a robust measure of ecological distance. Vegetatio **69**, 57–68.

Jenkins, C. N., Powell, R. D., Bass, O. L. and Pimm, S. L. (2003) Demonstrating the destruction of the habitat of the Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*). *Animal Conservation* **6**: 29-38.

John, R., Chen, J., Ou-Yang, Z.-T., Xiao, J., Becker, R., Samanta, A., Ganguly, S., Yuan, W. and Batkhishig, O. (2013) Vegetation response to extreme climate events on the Mongolian Plateau from 2000 to 2010. *Environment Research Letters* **8** (2013) 035033 (12pp)

Juggins. S. (2014) C^2 version 1.7.6. Software for ecological and palaeoecological data analysis and visualization. Newcastle University, Newcastle upon Tyne, UK.

Kantvilas, G. and Minchin, P. R. (1989). An analysis of epiphytic lichen communities in Tasmanian cool temperate rainforest. *Vegetatio* **84**, 99–112.
Lockwood, J. L., Baiser, B., Boulton, R. and Davis, M. (2006) Detailed study of Cape Sable seaside sparrow nest success and causes of nest failure: 2006 annual report, p. 77. Rutgers, The State University of New Jersey, New Brunswick, NJ.
Lockwood, J. L., Virzi, T., Boulton, R. L., Gilroy, J. J., Davis, M. J., Baiser, B. and Fenn, K. H. (2010) Recovering small Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*) subpopulations: breeding and dispersal of sparrows in the Everglades, Report to US Fish and Wildlife Service and National Park Service, Homestead, Florida, USA.
McCune, B and Grace, J. B. (2002) Analysis of ecological communities. MjM Software Design, Gleneden Beach, OR. 300 pp.
McCune B. and M. J. Mefford (2011) PC-ORD. Multivariate analysis of ecological data. Version 6.0 for Windows. MjM Software, Gleneden Beach, OR, USA.
Miao, S., Zou, C. B. and Breshears, D. D. (2009) Vegetation responses to extreme hydrological events: sequence matters. *The American Naturalist* **173** (1): 113-118.
Minchin, P. R. (1998) DECODA: Database for Ecological Community Data. Anutech Pty. Ltd., Canberra, Australia
Minchin, P. R., M. Folk and D. Gordon (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
Moses, C. S., Anderson, W. T., Saunders, C. and Sklar, F. (2013) Regional climate gradients in precipitation and temperature in response to climate teleconnections in the Greater Everglades ecosystem of South Florida. *Journal of Paleolimnology* **49**: 5-15.
Nott, M. P., Bass, O. L. Jr., 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
Pimm, S. L., Lockwood, J. L., Jenkins, C. N., Curnutt, J. L., Nott, M. P., Powell, R. D., 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, USA
R Core Team (2021) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.
Ross, M. S., Sah, J. P., Snyder, J. R., Ruiz, P. L., Jones, D. T., Colley, H., et al. (2004) Effect of Hydrologic Restoration on the Habitat of the Cape Sable seaside sparrow. 2003-2004. Year-2. Final Report submitted to Everglades National Park, Homestead, FL and U. S. Army Corps of Engineers, Jacksonville, FL. November 2004. 36 pp.
Ross, M. S., Sah, J. P., Snyder, J. R. Ruiz, P. L., Jones, D. T., Colley, H., et al. (2006) Effect of Hydrologic 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. 46 pp.

Ross, M. S., Ruiz, P. L., Sah, J. P. and Hanan, E. J. (2009) Chilling damage in a changing climate in coastal landscapes of the sub-tropical zone: a case study from south Florida. *Global Change Biology* 15 (7):1817-1832.

Sah, J. P., Ross, M. S., Snyder, J. R., P. L. Ruiz, Jones, D. T., Travieso, R., et al. (2007) Effect of hydrological restoration on the habitat of the Cape Sable seaside sparrow. Annual Report of 2005-2006. A report submitted to Everglades National Park, Homestead, FL. March 8, 2007. 49 pp.

Sah, J. P., Ross, M. S. and Ruiz, P. L. (2011a) Vegetation structure and composition within sparrow sub-population D. habitat. In Virzi et al. 'C-111 Project & Cape Sable seaside sparrow subpopulation D: Baseline data on sparrows, vegetation and hydrology – Annual Report 2011. pp: 39-60. Submitted to the South Florida Water Management District, West Palm Beach, FL.

Sah, J. P., Ross, M. S., Ruiz, P. L., Snyder, J. R., Rodriguez, D. and Hilton, W. T. (2011b) Cape Sable seaside sparrow habitat – Monitoring and Assessment - 2010. Final Report submitted to U. S. Army Corps of Engineers, Jacksonville, FL. (Cooperative Agreement # W912HZ-10-2-0025). April. 2011. 57 pp

Sah, J. P., Ross, M. S., Saha, S., Minchin, P. and Sadle, J. (2014a) Trajectories of vegetation response to water management in Taylor Slough, Everglades National Park, Florida. *Wetlands* **34** (Suppl. 1): S65-S79.

Sah, J. P. and Ross, M. S. (2014b) Status of Vegetation Structure and Composition within the Habitat of Cape Sable seaside sparrow Subpopulation D. Annual Report submitted to South Florida Water Management District (SFWMD), West Palm Beach, FL. PO # 4500079149. Sept 26, 2014. 25 pp.

Sah. J. P., Ross, M. S., Blanco, J., and Freixa, J. (2015) Evaluation of WDAT vs EDEN Water Surface Elevation data for studying vegetation: hydrology relationship in the southern Everglades. Report submitted to US Army Corps of Engineers. (CA # W912HZ-14-2-0022-P00001 & W912HZ-14-2-0023-P00001). August 2015. pp 31

Sah, J. P., Jirout, A., Stoffella, S., Ross, M. S. (2016) Status of Vegetation structure and composition within the habitat of Cape Sable seaside sparrow Subpopulation D. Annual Report submitted to South Florida Water Management District (SFWMD), West Palm Beach, FL. PO # 4500091267. Sept 30, 2016. 27 pp.

Sah, J. P., Pulido, C., Stoffella, S. and Ross, M. S. (2018) Status of Vegetation structure and composition within the habitat of Cape Sable seaside sparrow Subpopulation D. Annual Report submitted to South Florida Water Management District (SFWMD), West Palm Beach, FL. PO # 4500104597. September 2018. 29 pp.

Sah, J. P., Pulido, C., Stoffella, S., Castaneda, S. and Ross, M. S. (2020) Status of Vegetation structure and composition within the habitat of Cape Sable seaside sparrow Subpopulation D. Annual Report submitted to South Florida Water Management District (SFWMD), West Palm Beach, FL. PO # 4500117438. December 2020. 38 pp.

USFWS (U.S. Fish and Wildlife Service) (2009) Biological Opinion for the C-111 Spreader Canal (SC) Western Phase 1 Project, and its potential effects on the Cape Sable seaside sparrow (CSSS) and designated CSSS habitat. US Fish and Wildlife Service, Vero Beach, FL.

USFWS (U.S. Fish and Wildlife Service) (2016) Biological Opinion for the Everglades Restoration Transition Plan-2006. Submitted to U.S. Army Corps of Engineers, Jacksonville, FL

Virzi, T., Davis, M. J., Sah, J. P., Ross, M. S. and Ruiz, P. L. (2011) C-111 Project and Cape Sable seaside sparrow subpopulation D: Baseline data on sparrows, vegetation and hydrology. Annual Report – 2011. Submitted to South Florida Water Management District, West Palm Beach, FL. 80 pp.

Virzi, T. and Davis, M. J. (2013) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2013. Submitted to South Florida Water Management District, West Palm Beach, FL. 38 pp.

Virzi, T. and Davis, M. J. (2014) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2014. Submitted to South Florida Water Management District, West Palm Beach, FL. 40 pp.

Virzi, T., Davis, M. J. and Slater, G. (2015) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2015. Submitted to South Florida Water Management District, West Palm Beach, FL. 50 pp.

Virzi, T. and Davis, M. J. (2016) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2016. Submitted to South Florida Water Management District, West Palm Beach, FL. 36 pp.

Virzi, T. and Davis, M. J. (2017) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2017. Submitted to South Florida Water Management District, West Palm Beach, FL. 38 pp.

Virzi, T. and Murphy, S. P. (2018) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2018. Submitted to South Florida Water Management District, West Palm Beach, FL. 51 pp.

Virzi, T. and Tafoya, D. (2019) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2019. Submitted to South Florida Water Management District, West Palm Beach, FL. 64 pp.

Virzi, T. and Tafoya, D. (2020) C-111 Project and Cape Sable seaside sparrow subpopulation D. Annual Report – 2020. Submitted to South Florida Water Management District, West Palm Beach, FL. 80 pp.

Zweig, C. L. and Kitchens, W. M. (2008) Effects of landscape gradients on wetland vegetation communities: information for large-scale restoration. Wetlands **28**:1086–1096

APPENDIX

Appendix 1: Annual mean hydroperiod and water depth at the vegetation survey sites (n = 70) for 33 years (1991/92-2023/24 water years: May 1 – April 30). The 33-year (WY) average value is shown at the bottom. Hydroperiod for each site was calculated using field water depth-based ground elevation and water surface time-series data obtained from the Everglades Depth Estimation Network (EDEN) website.

Water Year	Hydroperiod (days)	SD	Water Depth (cm)	SD
1991_92	194	54	-1.35	6.11
1992_93	168	86	-2.23	6.15
1993_94	252	75	4.59	5.93
1994_95	269	59	8.41	5.99
1995_96	275	36	10.20	5.93
1996_97	189	52	0.39	5.98
1997_98	306	46	9.87	5.94
1998_99	232	62	2.02	6.03
1999_00	273	37	7.66	5.74
2000_01	168	40	-5.78	5.75
2001_02	194	40	-0.97	5.59
2002_03	174	50	-2.82	5.57
2003_04	245	55	2.69	5.64
2004_05	133	40	-10.67	5.78
2005_06	210	38	2.38	5.60
2006_07	156	62	-3.44	5.77
2007_08	190	40	-0.81	5.87
2008_09	190	37	-4.61	5.85
2009_10	288	43	5.73	5.65
2010_11	214	53	0.55	5.84
2011_12	179	42	-6.50	5.86
2012_13	276	31	9.45	5.47
2013_14	300	32	9.11	5.49
2014_15	213	43	-1.66	5.72
2015_16	276	49	6.76	5.55
2016_17	276	26	8.83	5.54
2017_18	283	21	10.33	5.45
2018_19	297	35	9.21	5.50
2019_20	272	27	6.00	5.52
2020_21	326	21	16.27	5.48
2021_22	297	36	9.97	5.46
2022_23	307	32	10.86	5.46
2023_24	350	12	20.73	5.43
33-yrs average	242	56	3.97	6.98

Appendix 2: List of CSSS sub-population D habitat vegetation monitoring sites sampled in 2020. Vegetation types are based on 2011 and 2024 species composition data. MWP = *Muhlenbergia* Wet Prairie; SOWP = *Schoenus* Wet Prairie; COWP = *Cladium* Wet Prairie; CM = *Cladium* Marsh; CRM = *Cladium-Rhynchospora* Marsh; RCM = *Rhynchospora-Cladium* Marsh; ERM = *Eleocharis-Rhynchospora* Marsh. Delta and slope (amount and rate of change in the target direction, respectively) were obtained for each site from trajectory analysis in which the base year for vegetation change was 2011 and statistical significance ($p < 0.1$) of delta and slope was tested using Monte Carlo's simulations with 1,000 permutations.

PLOT	X_UTM83	Y_UTM83	Veg. type (2011)	Veg. type (2024)	delta (Δ)	p-value (delta)	slope	p-value (slope)
D-01-02	544353	2801406	CWP	SOWP	0.343	0.152	0.038	0.025
D-01-03	545411	2804404	CM	CM	0.389	0.019	0.022	0.038
D-01-05	546405	2803430	CWP	CM	0.447	0.023	0.047	0.000
D-01-06	546354	2802406	CWP	CM	0.598	0.031	0.053	0.003
D-01-07	547357	2802410	SOWP	SOWP	0.218	0.199	0.019	0.102
D-01-08	547475	2801337	CM	CM	-0.185	0.809	-0.028	0.988
D-01-10	548377	2801401	CM	CM	-0.279	0.866	-0.033	0.974
D-02-01	545335	2805354	SOWP	SOWP	0.477	0.022	0.036	0.009
D-02-02	546327	2805342	CWP	CM	0.565	0.086	0.057	0.010
D-02-03	546334	2804375	CM	CM	0.426	0.099	0.052	0.000
D-02-04	543345	2803363	MWP	SOWP	0.552	0.003	0.043	0.001
D-02-06	547321	2803391	CM	CM	0.000	1.000	0.000	1.000
D-02-07	548307	2802395	CM	CM	-0.088	0.765	-0.009	0.851
D-03-01	547329	2804365	CWP	SOWP	0.007	0.485	0.001	0.497
D-03-02	544322	2804348	CM	CM	0.219	0.172	0.011	0.232
D-03-03	546337	2801375	CRM	CM	0.558	0.005	0.031	0.005
D-03-04	545343	2801363	CRM	CM	0.622	0.000	0.036	0.000
D-04-01	542834	2802855	CM	CM	0.618	0.001	0.049	0.000
D-04-02	542831	2801856	MWP	CRM	0.027	0.469	-0.009	0.718
D-04-03	543326	2802353	SOWP	SOWP	0.042	0.445	-0.004	0.615
D-04-04	543338	2801354	CWP	RCM	0.480	0.015	0.034	0.007
D-04-05	543835	2803855	CWP	CM	0.665	0.011	0.054	0.001
D-04-06	543835	2802853	SOWP	SOWP	1.018	0.002	0.083	0.000
D-04-07	543832	2801857	MWP	CWP	0.845	0.002	0.073	0.000
D-04-08	543832	2800854	CRM	CM	0.813	0.000	0.052	0.000
D-04-09	544836	2803855	SOWP	CM	0.837	0.000	0.071	0.000
D-04-10	544832	2801855	CM	CM	0.091	0.364	0.034	0.030
D-05-01	544836	2800854	SOWP	CRM	0.537	0.025	0.047	0.002
D-05-02	545835	2803854	SOWP	CWP	0.696	0.010	0.063	0.001
D-05-03	545835	2802849	CWP	CM	0.219	0.169	0.028	0.037
D-05-04	545831	2801855	CWP	CRM	0.418	0.057	0.034	0.012
D-05-05	545833	2800854	CM	CM	0.325	0.041	0.026	0.009
D-05-06	546832	2803854	CM	CM	0.253	0.068	0.013	0.147
D-05-07	546833	2802854	CM	CM	-0.014	0.535	0.007	0.235
D-05-08	546830	2801851	RCM	RCM	0.715	0.022	0.034	0.083
D-05-09	546834	2800850	CM	CM	-0.090	0.697	0.010	0.222
D-06-01	548330	2804355	CM	CM	0.398	0.054	0.021	0.107

PLOT	X_UTM83	Y_UTM83	Veg. type (2011)	Veg. type (2024)	delta (Δ)	p-value (delta)	slope	p-value (slope)
D-06-02	548333	2803356	CWP	CM	0.500	0.002	0.037	0.000
D-06-03	548832	2803849	CM	CM	0.050	0.386	0.006	0.284
D-06-04	548834	2802850	CRM	CRM	0.583	0.000	0.036	0.000
D-06-05	548834	2801851	CRM	CRM	0.033	0.377	0.008	0.138
D-06-06	549331	2804349	ERM	CM	-0.092	0.604	0.002	0.461
D-06-07	549336	2803354	CM	CM	-0.131	0.891	-0.012	0.949
D-06-08	549334	2802353	CM	CM	-0.053	0.675	-0.005	0.726
TD-01-01	544337	2803605	MWP	CRM	0.693	0.000	0.044	0.000
TD-01-02	544583	2803606	CWP	CM	0.646	0.011	0.055	0.001
TD-01-03	544835	2803604	SOWP	CM	0.948	0.000	0.069	0.000
TD-01-04	545084	2803606	CWP	CM	0.657	0.002	0.035	0.005
TD-01-05	545333	2803606	SOWP	CWP	0.901	0.001	0.061	0.000
TD-01-06	545582	2803607	CWP	CM	0.400	0.068	0.022	0.106
TD-02-01	544339	2803363	SOWP	CM	0.910	0.002	0.066	0.000
TD-02-02	544585	2803351	CWP	CM	0.767	0.003	0.062	0.000
TD-02-03	544837	2803353	CWP	CM	0.887	0.000	0.055	0.001
TD-02-04	545086	2803354	CRM	CM	0.834	0.004	0.066	0.001
TD-02-05	545337	2803351	CWP	CM	0.746	0.000	0.055	0.000
TD-02-06	545583	2803353	CWP	CM	1.383	0.000	0.087	0.000
TD-03-01	544337	2803104	CWP	CM	1.030	0.000	0.072	0.000
TD-03-02	544584	2803105	CWP	CM	0.613	0.006	0.038	0.003
TD-03-03	544834	2803107	SOWP	CM	0.741	0.002	0.043	0.009
TD-03-04	545084	2803104	SOWP	SOWP	0.544	0.005	0.040	0.002
TD-03-05	545332	2803104	SOWP	CWP	0.516	0.016	0.031	0.021
TD-03-06	545584	2803105	SOWP	CWP	0.490	0.041	0.028	0.079
TD-04-01	544335	2802852	SOWP	SOWP	0.972	0.000	0.066	0.000
TD-04-02	544585	2802853	SOWP	CM	0.844	0.001	0.057	0.000
TD-04-03	544835	2802853	SOWP	CM	0.702	0.000	0.047	0.000
TD-04-04	545085	2802853	CWP	CM	0.564	0.001	0.047	0.000
TD-04-05	545334	2802854	CWP	CM	0.609	0.005	0.039	0.003
TD-04-06	545584	2802856	CWP	CM	0.851	0.001	0.051	0.000
TD-05-01	544334	2802604	SOWP	CWP	1.092	0.000	0.065	0.000
TD-05-02	544587	2802607	SOWP	CWP	0.866	0.007	0.074	0.000
TD-05-03	544833	2802608	CWP	CM	0.803	0.000	0.057	0.001
TD-05-04	545085	2802605	CM	CM	0.541	0.019	0.055	0.000
TD-05-05	545332	2802603	CWP	CM	1.257	0.000	0.091	0.000
TD-05-06	545584	2802603	CM	CM	0.365	0.031	0.044	0.000
TD-06-01	544330	2802349	CWP	CM	0.970	0.001	0.069	0.000
TD-06-02	544585	2802352	CWP	CM	0.857	0.000	0.075	0.000
TD-06-03	544839	2802354	SOWP	CRM	0.824	0.000	0.059	0.000
TD-06-04	545084	2802353	SOWP	CRM	0.839	0.003	0.065	0.000
TD-06-05	545335	2802356	CWP	CM	0.336	0.030	0.014	0.098
TD-06-06	545585	2802355	CM	CM	0.304	0.076	0.023	0.047

Appendix 3: List of species recorded during vegetation samplings in 2011, 2014, 2016, 2018, 2020, 2022, and 2024 in CSSS Subpopulation D within C-111 Spreader Canal West Project area. Species' name in parenthesis are the current name of species accepted by ITIS (Integrated Taxonomic Information System).

SPCODE	Species name	Species Occurrence						
		2011	**2014**	**2016**	**2018**	**2020**	**2022**	**2024**
AESPRA	*Aeschynomene pratensis*							+
AGALIN	*Agalinis linifolia*	+	+	+	+	+	+	+
ALEBRA	*Aletris bracteata*	+		+		+		
AMBART	*Ambrosia artemisiifolia*	+						
ANNGLA	*Annona glabra*			+	+	+	+	
ARIPUR	*Aristida purpurascens*	+		+	+	+	+	+
ASCLAN	*Asclepias lanceolata*	+	+	+	+	+	+	+
ASCLON	*Asclepias longifolia*						+	+
ASTADN	*Aster adnatum* (*Symphyotrichum adnatum*)		+					
ASTBRA	*Aster bracei (Symphyotrichum tenuifolium var. aphyllum)*			+				
ASTDUM	*Aster dumosus (Symphyotrichum dumosum var. dumosum)*	+		+	+		+	+
ASTSPP	*Aster* sp.		+					
ASTTEN	*Aster tenuifolium (Symphyotrichum tenuifolium)*	+	+	+	+	+	+	+
BACCAR	*Bacopa caroliniana*	+	+	+	+	+	+	+
CALTUB	*Calopogon tuberosus*	+	+	+	+	+	+	+
CARSCA	*Carolina scalystem (Elytraria caroliniensis)*				+			
CASFIL	*Cassytha filiformis*	+	+	+	+	+	+	+
CENASI	*Centella asiatica*	+	+	+	+	+	+	+
CHIALB	*Chiococca alba*	+	+	+		+	+	+
CHRICA	*Chrysobalanus icaco*		+		+			
CLAJAM	*Cladium jamaicense*	+	+	+	+	+	+	+
CONERE	*Conocarpus erectus*		+	+		+		+
CRIAME	*Crinum americanum*	+	+	+	+	+	+	+
CYPHAS	*Cyperus haspan*		+					
DICDIC	*Dichanthelium dichotomum (Dichanthelium dichotomum var. ensifolium)*			+				
DYSANG	*Dyschoriste angusta*	+						
ELEBAL	*Eleocharis baldwinii*	+	+			+		+
ELECEL	*Eleocharis cellulosa*	+	+	+	+	+	+	+
ELEINT	*Eleocharis interstincta*			+			+	
ERAELL	*Eragrostis elliottii*	+		+	+	+	+	+
FUIBRE	*Fuirena breviseta*	+		+				
HELPIN	*Helenium pinnatifidum*			+		+	+	

SPCODE	Species name	Species Occurrence						
		2011	**2014**	**2016**	**2018**	**2020**	**2022**	**2024**
HYMPAL	*Hymenocallis palmeri*	+	+	+	+	+	+	+
HYPCIS	*Hypericum cistifolium*	+		+				
ILECAS	*Ilex cassine*	+	+	+	+	+	+	+
IPOSAG	*Ipomoea sagittata*	+	+	+	+	+	+	+
IVAMIC	*Iva microcephala*					+		
JUSANG	*Justicia angusta*		+	+	+	+	+	+
LEEHEX	*Leersia hexandra*	+	+	+		+		+
LINMED	*Linum medium* var. *texanum*	+	+	+		+		
LOBGLA	*Lobelia glandulosa*	+			+			
LUDALA	*Ludwigia alata*	+		+			+	
LUDCUR	*Ludwigia curtissii*			+				
LUDMIC	*Ludwigia microcarpa*	+	+	+	+	+	+	+
LUDREP	*Ludwigia repens*			+				
LUDSPP	*Ludwigia sp.*							+
MAGVIR	*Magnolia virginiana*			+	+			
MIKSCA	*Mikania scandens*	+		+	+			
MITPET	*Mitreola petiolata*	+	+	+		+	+	+
MORCER	*Morella cerifera*	+	+	+	+	+	+	
MUHCAP	*Muhlenbergia capillaris*	+	+	+	+	+	+	+
OXYFIL	*Oxypolis filiformis*	+	+	+	+	+	+	+
PANDIC	*Panicum dichotomiflorum*							+
PANHEM	*Panicum hemitomon*	+			+		+	
PANTEN	*Panicum tenerum (Coleataenia tenera)*	+	+	+	+	+	+	+
PANVIR	*Panicum virgatum*	+	+	+	+	+	+	+
PASMON	*Paspalum monostachyum*	+		+		+	+	
PELVIR	*Peltandra virginica*	+	+	+	+	+	+	+
PERBOR	*Persea borbonia*	+	+					
PHYNOD	*Phyla nodiflora*	+	+	+	+		+	+
PHYSTO	*Phyla stoechadifolia*	+						
PLUROS	*Pluchea rosea*	+	+	+	+	+	+	+
POLGRA	*Polygala grandiflora*	+	+	+		+		
PROPAL	*Proserpinaca palustris*	+						
RHYDIV	*Rhynchospora divergens*	+	+	+	+	+	+	+
RHYINU	*Rhynchospora inundata*	+		+				+
RHYMIC	*Rhynchospora microcarpa*	+	+	+	+	+	+	+
RHYSPP	*Rhynchospora* sp.			+				
RHYTRA	*Rhynchospora tracyi*	+	+	+	+	+	+	+
SABGRA	*Sabatia grandiflora*			+		+		
SABSTE	*Sabatia stellaris*	+	+			+		

SPCODE	Species name	Species Occurrence						
		2011	2014	2016	2018	2020	2022	2024
SAGLAN	*Sagittaria lancifolia*	+	+	+	+	+	+	+
SALCAR	*Salix caroliniana*	+		+				
SAMEBR	*Samolus ebracteatus*	+		+				
SARCLA	*Sarcostemma clausum (Funastrum clausum)*	+						
SCHALB	*Schoenolirion albiflorum*						+	+
SCHNIG	*Schoenus nigricans*	+	+	+	+	+	+	+
SCHRHI	*Schizachyrium rhizomatum*	+	+	+	+	+	+	+
SETPAR	*Setaria parviflora*	+						
SOLSTR	*Solidago stricta*	+	+	+	+	+	+	+
TAXDIS	*Taxodium distichum*		+	+	+	+	+	+
TEUCAN	*Teucrium canadense*		+					
TYPDOM	*Typha domingensis*	+	+	+	+	+	+	+
UNK22_25	*Unknowrn 2225_herb*						+	
UNK22_34	Unkwown 2234_herb						+	
UNK22_55	*Unknown 2255_herb*						+	
UNK24_62	*Unknown 2462_herb*							+
UNKD21	Unknown D02-01			+				
UNKS2252	Unknown 2252_seedling						+	
UNKSEED	Unknown seedling					+		
UNKTD25	Unknown TD02-05	+						
UNKTD56	Unknown TD05-06		+					
UNKWP16	Unknown WP16			+				
UTRCOR	*Utricularia cornuta*	+	+	+	+	+	+	
UTRFOL	*Utricularia foliosa*	+		+	+	+	+	+
UTRGIB	*Utricularia gibba*	+		+	+		+	+
UTRPUR	*Utricularia purpurea*			+	+	+	+	+
UTRRES	*Utricularia resupinata*			+			+	
UTRSPP	*Utricularia* sp.			+		+	+	+
UTRSUB	*Utricularia subulata*			+		+		
VICACU	*Vicia acutifolia*	+						

Appendix 4: Mean (± 1 S.D.) value of percent cover and importance value (IV) of major species averaged over all sites (n = 80) surveyed in 2011, 2014, 2016, 2018, 2020, 2022 and 2024 within the CSSS sub-population D habitat region. P-values are from non-parametric test, Friedman Analysis of Variance for multiple dependent samples. Different letters in superscript represent the significant difference as determined in non-parametric, Wilcoxon matched-pair test.

Plant species	**Survey years**							**p-value**
	2011	**2014**	**2016**	**2018**	**2020**	**2022**	**2024**	
	Mean Cover							
Cladium jamaicense	33.3±18.9^{a}	21.9±14.0^{b}	22.7±14.5^{b}	23.9±14.1^{b}	27.9±16.6^{c}	35.9±22.2^{a}	33.5±21.9^{a}	<0.001
Schoenus nigricans	11.1±17.8^{a}	6.0±10.5^{b}	5.3±9.6^{bc}	4.0±6.7^{cd}	2.8±6.3^{e}	3.9±7.4^{d}	3.0±6.6^{e}	<0.001
Muhlenbergia capillaris ssp. *filipes*	3.2±6.9^{a}	1.7±2.7^{b}	1.0±1.8^{c}	0.5±1.1^{d}	0.8±1.7^{ce}	0.4±1.1^{de}	0.2±0.4^{d}	<0.001
Rhynchospora microcarpa	3.3±5.0^{a}	1.5±1.9^{b}	0.6±1.5^{c}	0.4±0.9^{de}	0.3±0.6^{d}	0.6±1.0^{ce}	0.1±0.2^{c}	<0.001
Rhynchospora tracyi	4.5±6.5^{a}	3.5±3.7^{a}	1.8±3.4^{bc}	3.0±4.2^{ad}	2.8±4.8^{bd}	2.2±3.0^{bd}	1.0±1.8^{c}	<0.001
Eleocharis cellulosa	3.2±10.0	2.3±7.0	1.2±4.7	1.0±5.4	0.8±2.1	0.5±1.7	0.3±0.9	0.427
Importance Value (IV)								
Cladium jamaicense	40.6±20.6^{a}	42.1±19.2^{a}	48.6±20.5^{b}	50.0±19.2^{bc}	52.2±19.0^{cd}	52.3±19.5^{d}	54.4±19^{e}	<0.001
Schoenus nigricans	11.2±15.3^{a}	10.0±14.1^{ab}	10.7±15.4^{ab}	10.0±14.4^{b}	7.0±11.9^{c}	7.5±12.0^{c}	7.6±13.3^{c}	<0.001
Muhlenbergia capillaris ssp. *filipes*	4.8±7.6^{a}	4.5±6.3^{a}	4.1±6.2^{a}	2.3±4.0^{b}	3.5±5.3^{a}	2.3±3.7^{b}	1.7±3.1^{b}	<0.001
Rhynchospora microcarpa	6.0±6.4^{ab}	6.3±6.0^{a}	5.0±4.6^{b}	2.9±3.4^{c}	3.4±3.9^{cd}	3.9±4.1^{d}	2.1±2.7^{e}	<0.001
Rhynchospora tracyi	7.3±7.8^{a}	10.1±8.9^{bc}	8.5±11.0^{ade}	12.8±9.7^{f}	10.6±9.8^{b}	10.0±8.5^{bd}	8.8±8.6^{ce}	<0.001
Eleocharis cellulosa	4.2±11.4	5.1±13.3	3.6±9.9	3.3±9.2	4.0±9.2	3.8±8.0	3.8±7.1	0.645

Appendix 5: Mean (± 1 S.D.) value of vegetation structural measurements and species richness for two groups of sites, wet prairie (WP) vs marsh (M) surveyed in 2011, 2014, 2016, 2018, 2020, 2022 and 2024 within the CSSS sub-population D habitat region. Grouping of sites as WP and M is based on the 2011 site classification. Different letters in superscript represent the significant difference as determined in post-hoc (Tukey's) test with beefaroni corrections using "multcomp" package in R.

Vegetation structural variables	Vegetation type (2011)	Sampling years						
		2011	2014	2016	2018	2020	2022	2024
Vegetation cover (%)	WP	38.9±16.0^{a}	32.4±12.1^{abc}	34.3±12.6^{abc}	28.8±12.0^{b}	31.4±13.4^{b}	39.0±17.5^{ac}	36.7±15.1^{ac}
	M	40.0±19.4^{ac}	33.0±14.0^{ab}	33.7±15.6^{ab}	29.4±11.6^{b}	33.2±18.1^{ab}	44.9±18.3^{c}	46.6±20.2^{c}
Vegetation height (cm)	WP	51.5±13.1^{a}	58.1±11.5^{b}	56.9±12.3^{b}	60.2±13.9^{b}	67.0±13.6^{c}	66.1±13.1^{c}	59.2±15.7^{b}
	M	55.6±15.8^{a}	55.6±11.2^{a}	55.8±13.1^{ab}	62.1±18.2^{b}	73.7±15.9^{c}	74.4±15.8^{c}	74.0±16.8^{c}
Species richness (species/plot)	WP	11.4±3.0^{a}	9.8±2.4^{ac}	12.2±3.9^{ab}	9.5±2.5^{c}	10.8±3.2^{abc}	10.8±2.8^{abc}	9.6±2.4^{c}
	M	6.1±3.1^{a}	5.9±3.3^{a}	6.2±3.4^{a}	5.2±2.9^{a}	6.1±2.9^{a}	6.1±3.1^{a}	5.8±2.8^{a}
Aboveground plant biomass ($g\ m^{-1}$)	WP	509±150^{ab}	483±133^{ab}	493±142^{ab}	463±140^{a}	516±173^{ab}	582±211^{b}	529±184^{b}
	M	542±218^{a}	476±145^{a}	484±164^{a}	474±141^{a}	565±221^{a}	672±226^{b}	692±260^{b}

Appendix: Figures

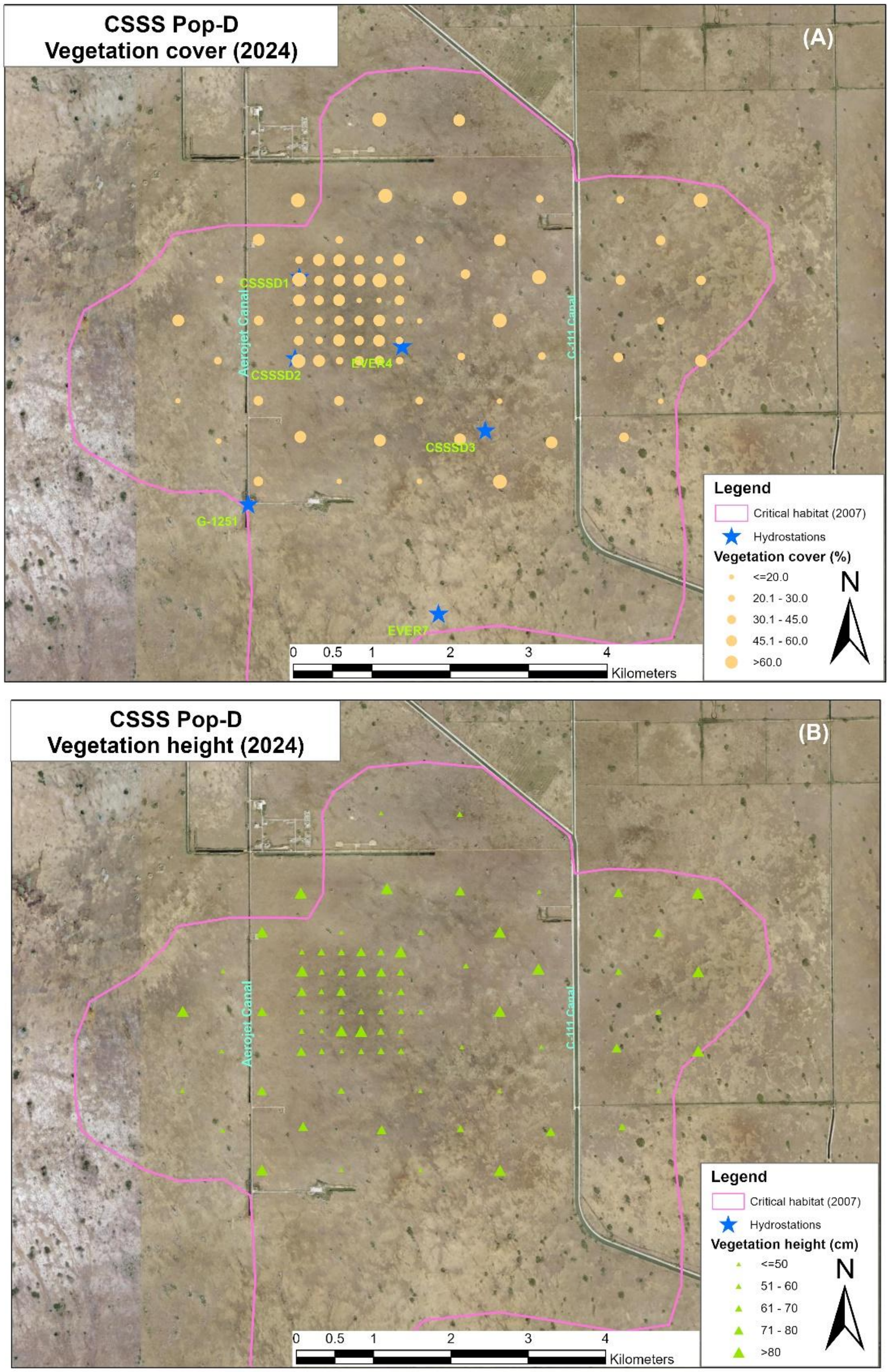

Figure A-1: Mean total vegetation cover and height at 80 sites surveyed during 2024 in CSSS Sub-population D habitat within C-111 Spreader Canal Western Project area.

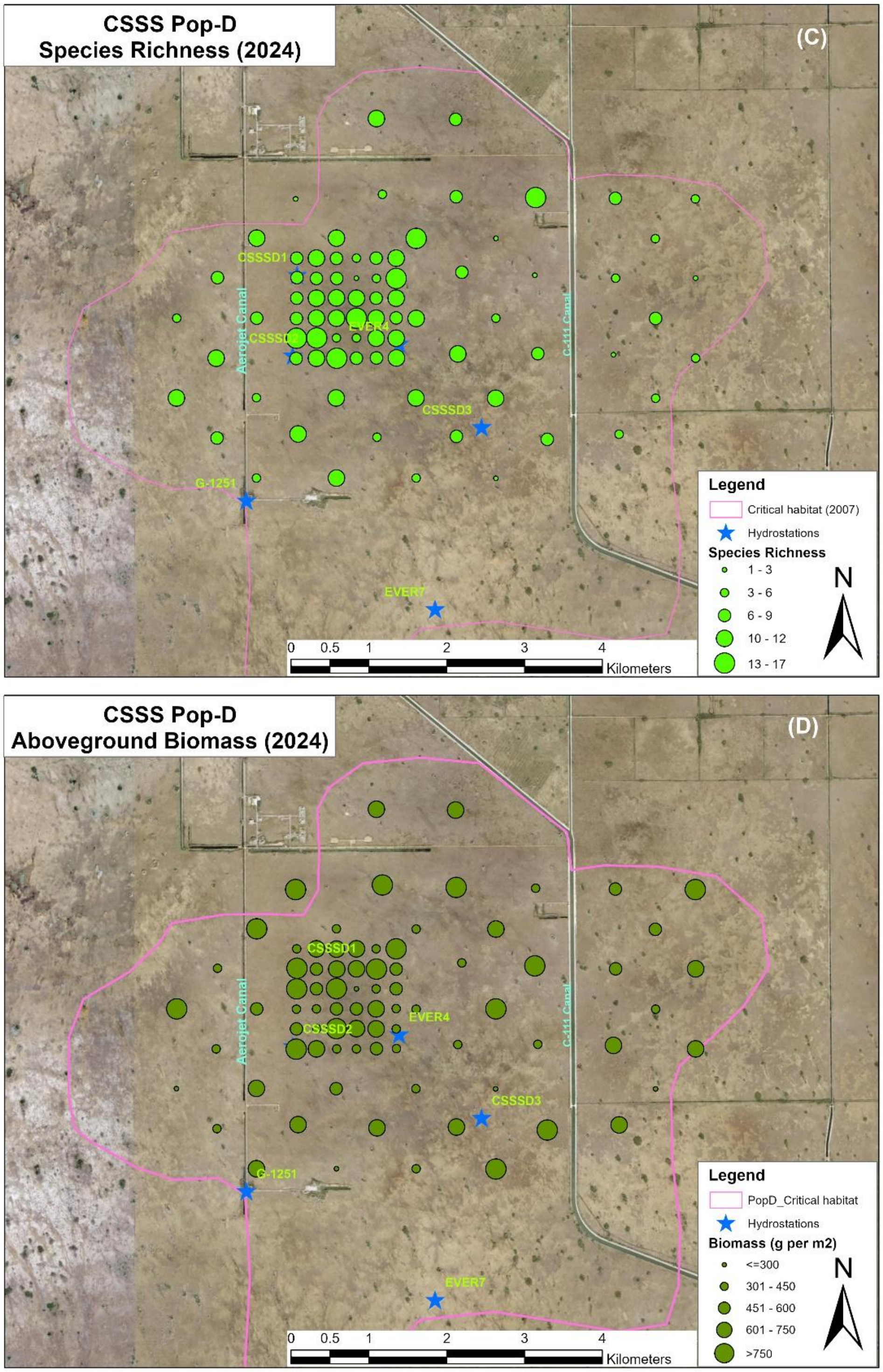

Figure A-2: Mean species richness and aboveground biomass at 80 sites surveyed during 2024 in CSSS Sub-population D habitat within C-111 Spreader Canal Western Project area.

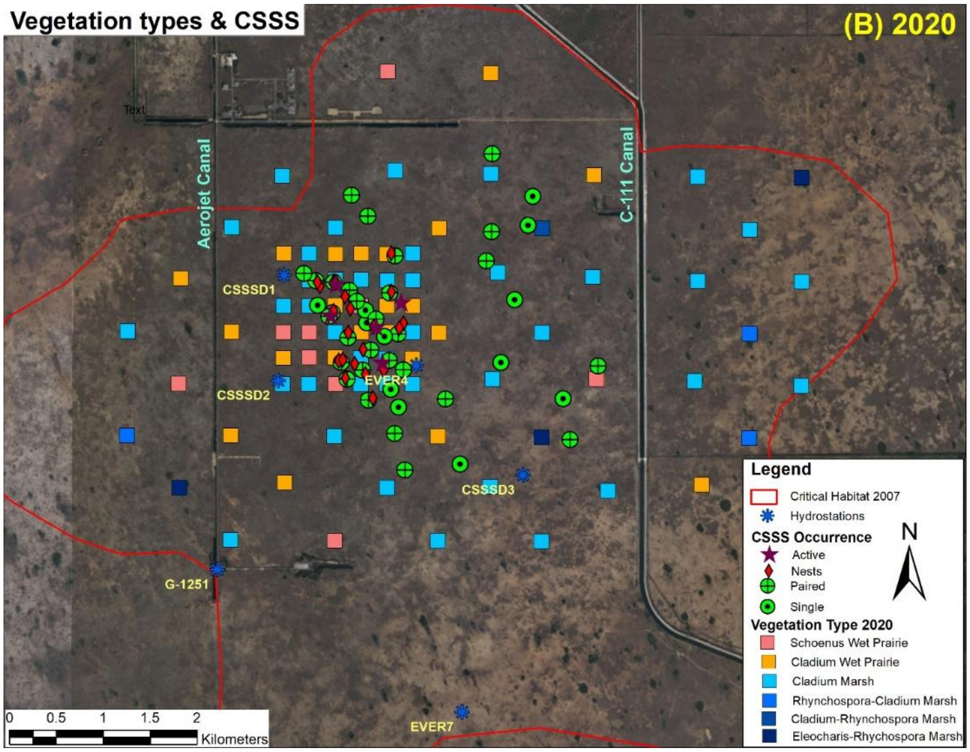

Figure A-3: Vegetation types at 79 sites surveyed during 2020 study and sparrow occurrence, as recorded by Virzi and Tofoya (2020) within the habitat of CSSS sub-population D within C-111 Spreader Canal Western Project Area.

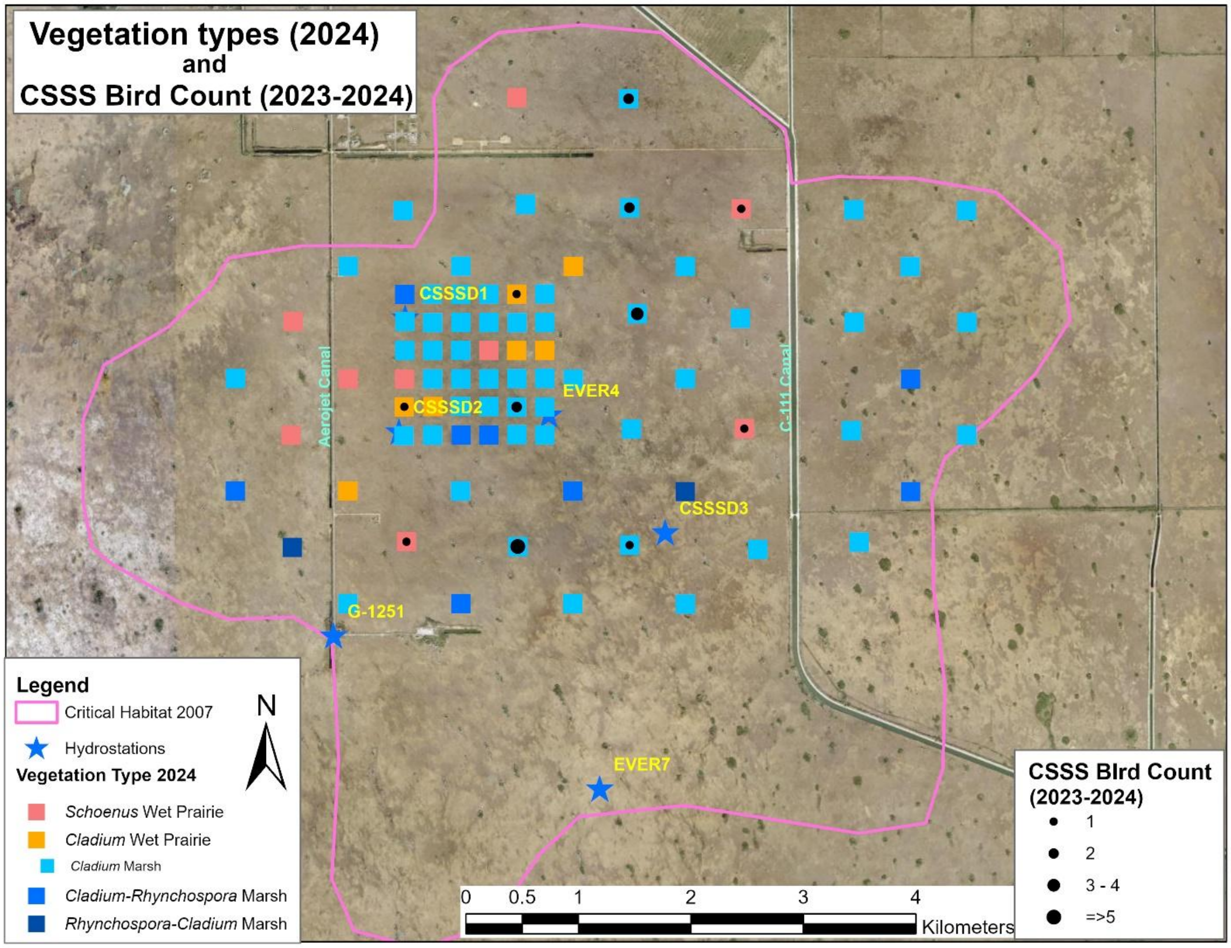

Figure A-4: Map showing the vegetation types at the C111 Sub-population D sites surveyed in 2024 and the number of birds at the sparrow census points with at least one bird recorded during the annual sparrow survey in any of two years (2023 and 2024).