



**KNOWLEDGE  
SYNTHESIS**

*of*

**Cape Sable  
Seaside Sparrow  
Science**

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Cover photograph by Lori Oberhofer, National Park Service

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# TABLE OF CONTENTS

iv Purpose  
iv Infographics  
v Acronyms

## 1

### BACKGROUND

1 Historical Context  
3 Population Trends  
4 Recent Trends  
6 Life History  
6 Life Span  
7 Reproduction  
10 Genetics  
11 Spatial Ecology  
11 Territory and Home Range  
12 Population Density  
12 Juvenile Dispersal  
14 Adult Movement  
15 Flocks

## 17

### ECOLOGY

17 Trophic Interactions  
17 Diet  
17 Predation  
18 Habitat  
20 Critical Habitat  
21 Nesting Habitat  
22 Habitat Avoided  
23 Habitat Threats  
24 Predicted Change  
26 Hydrometrics  
26 Hydrology and Habitat  
28 Hydrology and Breeding Impacts  
29 Hydrology and Occurrence  
33 Fire  
33 Fire Types  
34 Timing of Fire  
34 Fire Frequency

## 39

### SURVEY DATA

39 Overview  
39 Population Estimation  
41 Data Challenges  
44 Risk

## 46

### POTENTIAL SUITABLE CONDITIONS

46 Land Cover  
50 Decision Making Tools

## 54

### CLIMATE CHANGE

56 References



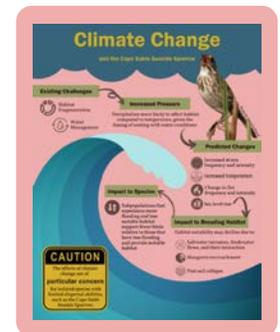
## PURPOSE

This report represents a literature review of science conducted on the Cape Sable Seaside Sparrow (*Ammodramopsitta maritima mirabilis*, hereafter “CSSS” or “sparrow”). This information can be used as a foundation for the upcoming Species Status Assessment and for updating the CSSS Recovery Plan. This report focuses on areas of interest relative to CSSS management such as habitat, hydrology, fire, and population estimates.

We include peer-reviewed scientific literature, synthesis reports, and some field reports. Many field reports to funding agencies and other documents exist beyond what is reviewed here; however, we include the most relevant documents relative to the focus of CSSS management. All documents reviewed are listed in the References section.

## Infographics

To support this synthesis, a summary infographic is included at the end of each chapter. These visuals are meant to share concise information about the CSSS to broader audiences. Click on an thumbnail below to see the full infographic within the report.





## ACRONYMS

<b>BCNP</b>	Big Cypress National Preserve
<b>CJS</b>	Cormack-Jolly-Seber
<b>COP</b>	Combined Operational Plan
<b>CSSS</b>	Cape Sable Seaside Sparrow
<b>EDEN</b>	Everglades Depth Estimation Network
<b>ENP</b>	Everglades National Park
<b>ESA</b>	Endangered Species Act
<b>ITIS</b>	Integrated Taxonomic Information System
<b>MHHW</b>	mean higher high water
<b>mtDNA</b>	mitochondrial DNA
<b>NPS</b>	National Park Service
<b>NSM</b>	Natural System Model
<b>RSM</b>	Regional Simulation Model
<b>SFNRC</b>	South Florida Natural Resource Center
<b>SLR</b>	sea level rise
<b>SRS</b>	Shark River Slough
<b>TS</b>	Taylor Slough
<b>USFWS</b>	U.S. Fish and Wildlife Service
<b>USGS</b>	U.S. Geological Survey
<b>WCA</b>	Water Conservation Area
<b>WP</b>	wet prairie



## BACKGROUND

### Historical Context

The Cape Sable Seaside Sparrow (*Ammospiza maritima mirabilis*, hereafter “CSSS” or “sparrow”) was originally listed as an endangered species in 1967 (32 FR 4001) in the first issued list of endangered species under the Endangered Species Preservation Act of 1966 (Public Law 89-669; 80 Stat 926). Protection was continued under the Endangered Species Conservation Act of 1969 (Public Law 91-135), until it was listed under the Endangered Species Act (ESA; ESA 1973, as amended). The taxonomic nomenclature for the CSSS has changed multiple times, although regardless of its taxonomic classification, it has always been either its own species or subspecies, distinct from other seaside sparrows. The CSSS was originally described as its own species (*Thryospiza mirabilis*; Howell 1919) and later termed *Ammospiza maritima* (Howell 1932, Stimson 1968). Afterwards it was relegated to subspecific status in 1973 (*Ammospiza maritima mirabilis*; Eisenmann 1973), later assigned to the genus *Ammodramus* (*Ammodramus maritimus mirabilis*; USFWS 1983, 1999a). Recent genetic research indicates that *Ammodramus* is polyphyletic, with the recommendation that *Ammospiza* be reinstated for the clade that includes *maritimus* (Klicka et al. 2014). Current genetic studies classify the CSSS as *Ammospiza maritima mirabilis* (Davis et al. 2021, Beaver et al. 2019), which agrees with the current classification for seaside sparrows by the American Ornithological Society’s North American Classification Committee (Chesser et al. 2018). At the time of writing, the Integrated Taxonomic Information System is in the process of changing the genus of the CSSS to *Ammospiza*.

The geographic range of the CSSS is completely within the southern Everglades of south Florida (Lockwood et al. 2001). The CSSS is endemic to south Florida,

non-migratory, relatively sedentary, and is presently found in short hydroperiod freshwater prairies primarily in Everglades National Park (ENP; and adjacent areas) and Big Cypress National Preserve (BCNP). The CSSS is geographically and ecologically distinct from other sparrows (Curnutt et al. 1998); it differs from other extant subspecies of seaside sparrow in that it inhabits inland freshwater marshes as compared to congeners that inhabit mainly coastal brackish habitats (Post and Greenlaw 1994, 2000).

The range of the CSSS has historically comprised a larger extent than it presently occupies. The historic range extended from Cape Sable to Ochopee (Big Cypress Basin), and east to Taylor Slough (ENP; Werner 1975). Presently, the known distribution is primarily restricted to two areas of marl prairies within ENP and BCNP, including some areas directly adjacent to ENP (the majority of subpopulation D is on state-managed land; Fig. 1). The locations where the CSSS is found are east and west of Shark River Slough and flanking Taylor Slough (USFWS 1999a) in areas with *Muhlenbergia* (Werner and Woolfenden 1983, Curnutt et al. 2000, Fletcher et al. 2000, Davis et al. 2005; *Muhlenbergia capillaris* var. *filipes*, hereafter referred to as “*Muhlenbergia*” or “muhly grass,” note the accepted scientific name has changed multiple times [ITIS 2021, Wunderlin et al. 2021]).

Historically, the CSSS occupied additional areas and habitats that it does not currently occupy (USFWS 1983, Werner and Woolfenden 1983, Post and Greenlaw 2000). First described and recorded by Howell in 1919, it was described as “only moderately numerous” on Cape Sable (Monroe County) at the time of recording (Howell 1919). It was thereafter reported north of

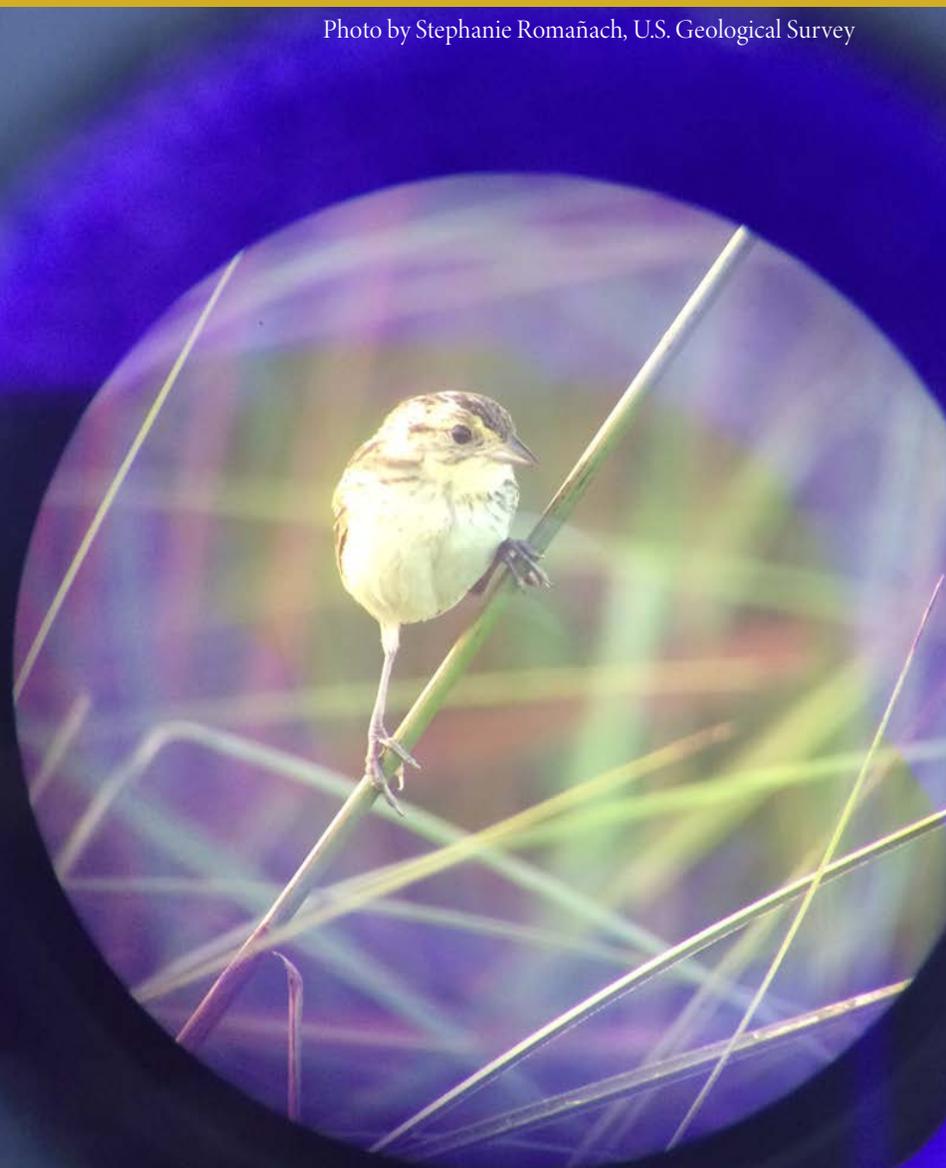
Cape Sable in Monroe County and in southern Collier County, near what is presently named Everglades City (southern Big Cypress Swamp, southern BCNP; Nicholson 1928, Stimson 1956), and Ochopee (southern Big Cypress Basin; Anderson 1942, Stimson 1956).

The CSSS is no longer found at its earliest known sites, including the Ochopee prairies in Big Cypress Basin, the Everglades City area, and the clumped *Spartina* marshes of Cape Sable (Stimson 1968, Kushlan et al. 1982, Werner and Woolfenden 1983, USFWS 1999a, Post and Greenlaw 2000, Walters et al. 2000). A hurricane in 1935 may have contributed to the demise of the CSSS on Cape Sable, via direct effects such as mortality (Stimson 1956, 1968) and indirect effects such as habitat alteration (Werner and Woolfenden 1983).

The vast stands of *Spartina* (cordgrass) are no longer present on Cape Sable, and the coastal marl prairie has turned into a very narrow strip and has changed in species composition (Werner and Woolfenden 1983). The original marshes in which the sparrow was found (once extensive cordgrass marsh) changed after the hurricane of 1935 to being dominated by mangroves, bare mud flats, and halophytic forbs (Kushlan et al. 1982, Kushlan and Bass 1983). The CSSS has not been observed on Cape Sable during the breeding season since the 1935 storm (Stimson 1956; although see Semple [1936] for one possible observation of the CSSS on Cape Sable in April 1936), save for a small number of observa-

tions in the 1970s (Werner 1971, 1975). No birds were observed on Cape Sable in 1979, 1980, or 1981 after a fire that burned what little cordgrass was left on Cape Sable (Kushlan et al. 1982). While there were some observations of the CSSS in the Ochopee marshes of the Big Cypress Basin in the first half of the 20<sup>th</sup> century (Anderson 1942), very few sparrows have been found there since the mid-1980s. The decline in birds in the Big Cypress Basin area (Ochopee) has been attributed to fires and salinity changes associated with altered hydrology and mangrove encroachment (Stimson 1956, Kushlan et al. 1982, USFWS 1983). The decline in CSSS numbers from the western edge of the southern Everglades (e.g., Everglades City area) has been attributed to widespread fires (Stimson 1968).

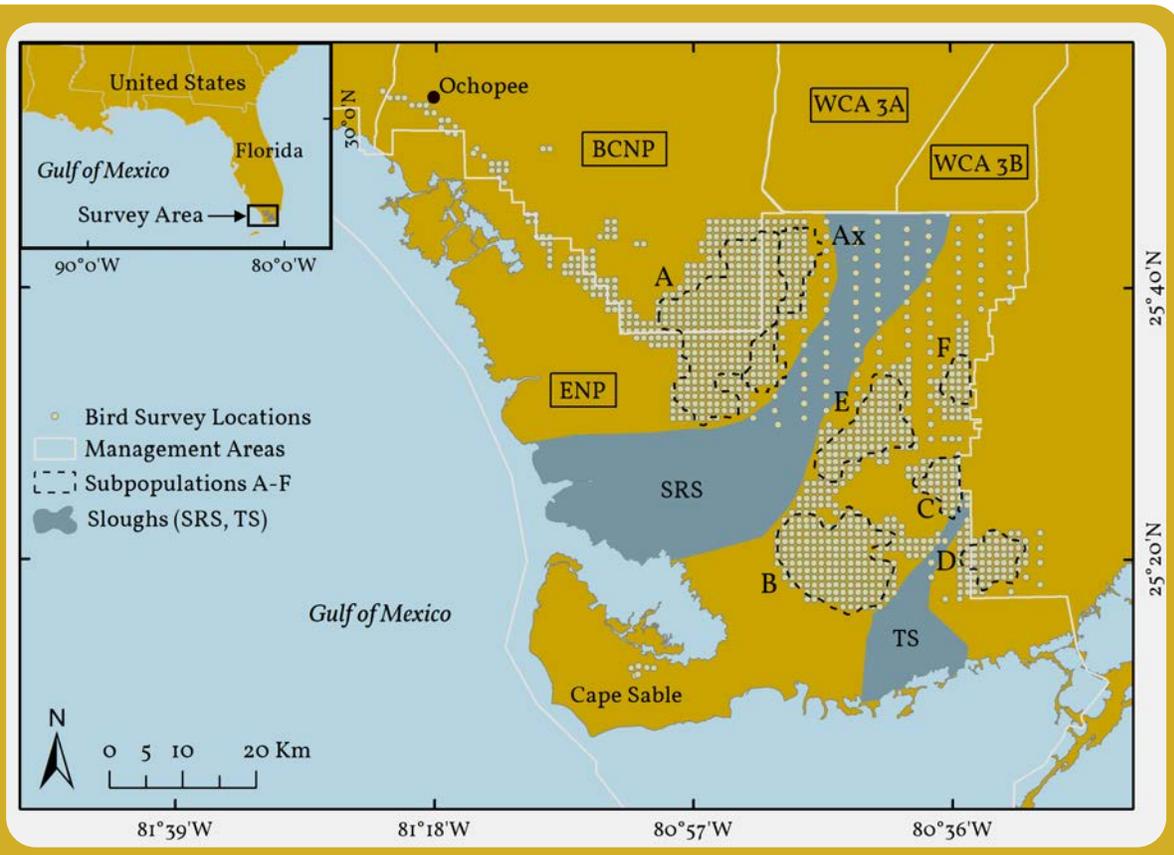
Photo by Stephanie Romañach, U.S. Geological Survey



## Population Trends

Since the CSSS range-wide surveys began in 1981, the population has declined by an estimated 63%, from approximately 6,600 birds to an estimated 2,448 birds in 2021 (personal communication, Tylan Dean, July 19, 2021). Although the CSSS range is delineated by six subpopulations (A–F; subpopulation delineations derived from USFWS [2020]), only two support CSSS subpopulations numbering in the hundreds or thousands of birds, subpopulations B and E (see USFWS [2020] for a complete table showing the bird counts and corresponding population estimates from the range-wide helicopter surveys from 1981–2019; Fig. 1). Subpopulation A declined markedly in the 1990s, and

the survey in 2019 was the first to count zero birds in that area. Subpopulations C, D, and F are much smaller now than they were in 1981 (USFWS 2000, Walters et al. 2000). At the time of writing, the preliminary 2021 population estimate from NPS based on the traditional method of multiplying the number of birds counted by 16 (data are subject to review and revision by ENP prior to release) is 2,448 birds, with each subpopulation estimate at: A at 0 birds, B at 1,488 birds, C at 112 birds, D at 288 birds, E at 528 birds, and F at 32 birds (personal communication, Tylan Dean, July 19, 2021). For information on population estimation methods, see *SURVEY DATA* chapter.



**Figure 1.** There are six spatially delineated subpopulations (A–F; subpopulation boundaries derived from USFWS [2020]) of the Cape Sable Seaside Sparrow (CSSS; *Ammospiza maritima mirabilis*). Subpopulation A is located on the western side of Shark River Slough (SRS; Ax represents an expanded area of subpopulation A containing potential future suitable habitat for the CSSS), and subpopulations B–F are located on the eastern side of SRS; Taylor Slough (TS) is located east of SRS. The portion of subpopulation D outside of Everglades National Park (ENP) is located within the Southern Glades Wildlife and Environmental Area. Abbreviations: BCNP=Big Cypress National Preserve; WCA=Water Conservation Area.

## Recent Trends

Virzi and Tafoya (2021) report on the status of subpopulations A–E based on analysis of recent field data from in-depth demographic study plots through 2020. They do not provide detailed information on subpopulation F, though range-wide survey population estimates presented above show consistently low annual counts for subpopulation F. Summaries of their findings are provided below:



Photo by Ulf Gotthardson

### Subpopulation

# A

Data from range-wide surveys and in-depth demographic monitoring (Virzi and Tafoya 2021) indicate that subpopulation A has not recovered to pre-1990 occupancy levels. Data from the in-depth study plots from 2012–2017 indicate that the sex-ratio was somewhat male biased at 0.61, and from 2018–2020 the sex-ratio was highly male-biased at 0.75. Multi-brooding is important for population viability in the CSSS, and from 2012–2017 only two cases of multi-brooding were observed. They report that CSSS numbers in subpopulation A since Hurricane Irma in 2017 have been very low, that subpopulation A is vulnerable to extirpation, and that it is unlikely to recover on its own. No birds were observed in 2019 via the range-wide surveys, however four adult sparrows were observed in the in-depth study plots. Additionally, in 2020, four adult sparrows and two juvenile sparrows were observed in the in-depth study plots, and a breeding pair was documented, suggesting that successful breeding occurred. Although sparrow numbers in subpopulation A have been very low in recent years, successful breeding has been recorded each year from 2008–2020 (Virzi and Tafoya 2021).

# B

### Subpopulation

Subpopulation B supports the largest number of sparrows. Virzi and Tafoya (2021) report declines in both the number of individuals observed and the number of nests from 2012–2017 in multiple in-depth study plots in subpopulation B, along with increasing trends of male biased sex-ratios. These study plots are on the periphery of subpopulation B, and the establishment of additional in-depth study plots in other areas of subpopulation B was recommended, possibly near the core area of B, to determine if these declines are local occurrences or indicative of a more widespread trend. The mean sex ratio at one in-depth study plot was 0.58 from 2012–2015, 0.69 in 2016, and 0.89 in 2017. Only one (unsuccessful) breeding pair was observed in this study plot in 2017, and no breeding was observed there from 2018–2020. In August of 2017, this study plot experienced a wildfire that burned approximately 75% of potentially suitable sparrow habitat. Another in-depth study plot in subpopulation B that was monitored from 2017–2020 (and was set up in response to declines in the other in-depth study plot) also showed highly male-biased sex ratios, and some successful breeding was observed there with a downward trend in observed birds and breeding numbers; however, hatching and fledging rates for birds that did breed were high (Virzi and Tafoya 2021).

## Subpopulation

C

Subpopulation C has one of the smallest population sizes, along with subpopulations D and F (USWFS 2020). Subpopulation C has experienced over-drying since the 1990s from water management but is expected to benefit from restoration efforts to increase hydroperiod. Over-drying and a high frequency of human-caused fires contributed to almost yearly fires in subpopulation C before 1999. Starting in 1999, ENP suppressed many fires in this area and burned buffer strips along the eastern boundary of ENP to provide protection from human-caused fires. Subpopulation C has showed a substantial recovery in numbers since 1992; however, sparrow populations have not recovered to 1981 numbers. Although this subpopulation has suffered from male-biased sex ratios and a relatively small subpopulation size, evidence exists from the in-depth demographic monitoring that this subpopulation has exhibited some positive trends in the past. Multi-brooding was observed each year from 2006–2009, afterwards which no in-depth monitoring was conducted until 2017. In 2017, higher densities of sparrows were observed, the sex ratio was 0.59, and multi-brooding was observed. Limited in-depth monitoring was conducted in subpopulation C from 2018–2020. Both the in-depth monitoring and the helicopter point count surveys indicate that the distribution of sparrows is limited to the northern portion of the subpopulation (Virzi and Tafoya 2021).

D

## Subpopulation

Subpopulation D has shown substantial declines in numbers since the range-wide surveys began in 1981. This subpopulation has regularly exhibited very high male-biased sex ratios and low population size, but starting in 2018 through 2020, substantial growth was observed in subpopulation D, including more even sex ratios, greater levels of successful breeding, expansion of area used within the subpopulation, and higher densities of sparrows observed via in-depth monitoring. The sex ratio before Hurricane Irma in 2017 was very male-biased, with a mean of 0.79 from 2011–2017; however, the sex ratio was reported as 0.67 in 2018, 0.59 in 2019, and 0.54 in 2020. Additionally, very high levels of multi-brooding were observed in subpopulation D in 2019. Virzi and Tafoya (2021) state that although recent trends are positive for subpopulation D, habitat is anticipated to change towards more marsh-like communities with sawgrass which are not suitable for sparrow breeding.

## Subpopulation

E

Subpopulation E is one of the two largest and most stable subpopulations. Sex ratios reported in 2018 were 0.61 and 0.69 at two in-depth study plots in subpopulation E, and limited demographic monitoring in 2019 indicated an increase in the male-biased sex ratio. Additionally, recent hydrologic changes have likely led to increased hydroperiods in subpopulation E, and more hydrologic changes are expected in the future. Increased hydroperiods have led to areas with less suitable, wetter habitat for the CSSS. Based on in-depth monitoring in study plots across a gradient of wetter habitat to drier habitat (from northwest to southeast), sparrows were observed moving eastward away from the wetter areas as the breeding season advanced, and some sparrows moved out of the study area completely, concerning because this is the only subpopulation outside of subpopulation B that is large and stable. Virzi and Tafoya (2021) highly recommend additional in-depth monitoring of subpopulation E in the future as restoration and water management actions are implemented.

## Life History

### ***Life Span***

The CSSS is relatively short-lived, with a mean life-span range of two to five years (Lockwood et al. 2003, Boulton et al. 2009); however banded males have been observed at eight years of age (Boulton et al. 2009).

Mean individual annual apparent survival estimated using the Cormack-Jolly-Seber (CJS) method was reported as 0.66 for adult males with territories by Lockwood et al. (2001; subpopulations A, B, and E from 1994–1998), 0.56 for subpopulation A by Boulton et al. (2009; from 1997–2007), and also varied by sex for subpopulations B and E, ranging from 0.47–0.83 for males and from 0.29–0.69 for females (pooled estimate across both subpopulations from 2002–2006, calculated annually; Boulton et al. 2009). Apparent annual survival was variable from year to year in subpopulations B and E, females had 14–19% lower survival than males, and juveniles had lower survival rates than adults (juvenile annual survival estimates ranged from 0.09–0.47; Boulton et al. 2009). Gilroy et al. (2012a) report CSSS juvenile survival rates of 0.34 using a Bayesian modification of the standard CJS model that allowed for incorporation of emigration and habitat-dependent dispersal; by comparison, they also document apparent juvenile survival of 0.15 using the standard CJS model that does not incorporate emigration effects. Pimm et al. (2002) report the minimum annual adult survival rate at  $>0.50$  (based on data from 1994–1998).



Photo by Danny Bales

## Reproduction

Nesting for the sparrow is observed during the late dry season, from approximately March–June, but can start earlier or extend longer (Dean and Morrison 1998, Davis et al. 2005, Boulton et al. 2009, USFWS 2020) depending on biotic and abiotic factors; the peak of nesting season is approximately April–May (Walters et al. 2000, Lockwood et al. 2003, La Puma et al. 2007) and can vary from year to year. Both sexes initiate breeding in the first spring after they hatch (Post and Greenlaw 1994). Behaviors such as defending territories, courtship, and mating may begin in February (Werner and Woolfenden 1983, Lockwood et al. 1997) or as early as late January (Pimm et al. 2002). It is reported that the length of the nesting season each year is associated with the length of the dry season in that year (Lockwood et al. 1997, 2001), where the beginning of the breeding season coincides with receding water levels at the onset of the dry season, and the end of the breeding season generally coincides with the onset wet season (Lockwood et al. 1997, Elder and Nott 2008). Dean and Morrison (2001) report that the cessation of breeding activities for each individual coincides with the onset of molting, which can vary substantially among individuals (molting can begin in July, and most individuals have completed molting by the end of October).

A full nesting cycle for the CSSS takes approximately 35–45 days to complete, although shorter and longer durations are reported (e.g., 30 days and 60 days; Lockwood et al. 1997, 2001, Pimm et al. 2002). Pairs may remain mated for successive nesting cycles (Werner and Woolfenden 1983, Pimm et al. 2002) and the CSSS is generally monogamous (Post and Greenlaw 1994), although mating between different individuals within a breeding season is reported for both males and females (Pimm et al. 2002); polygyny is also observed (Pimm et al. 2002). Sex ratios are reported as male-biased and are more imbalanced in small subpopulations (A, C, D, F) compared to larger subpopulations (B, E; Virzi

et al. 2016). The CSSS usually produces two broods per breeding season (Lockwood et al. 1997, 2001), and third or fourth brood attempts, while possible, can be rare in a single breeding season (Werner and Woolfenden 1983, Curnutt et al. 2000, Walters et al. 2000, Lockwood et al. 2001). Females may start new nests prior to fledging of the previous brood (Werner and Woolfenden 1983). Nests are not re-used in subsequent nesting attempts (Lockwood et al. 1997, Pimm et al. 2002).

The mean clutch size for the CSSS is approximately three eggs per nest (Lockwood et al. 2001, Pimm et al. 2002). Sometimes, only two eggs are laid (Werner and Woolfenden 1983), but as many as five are reported (Howell 1932). Pimm et al. (2002) report no difference in number of eggs laid per nest across years or subpopulations (n=160 nests from 1996–2000, mean clutch size=3.1 eggs/nest). Lockwood et al. 2001 report a mean clutch size of 3.2 eggs per nest (SD=0.6, n=126 nests; data set overlaps with Pimm et al. [2002]) and that clutch size did not differ between early and late season nests. Lockwood et al. (2001) report that sparrows averaged 2.9 nestlings per nest (SD=0.9, n=179 nests), which also did not vary between early and late season nests. However, the numbers of nestlings in each nest varied by year and subpopulation (n=176 nests from 1996–2000; Pimm et al. 2002). Hatching success for 24 nests monitored in eastern ENP in 1996 and 1997 was 0.88 (Lockwood et al. 1997). The eggs in second or third clutches (only a single third clutch was observed) either did not hatch or were abandoned (Lockwood et al. 1997). Boulton et al. (2011) report a hatching success rate between 0.66 and 1.00 for the CSSS. Causes for the loss of eggs or young in the nest are attributed primarily to apparent predation, nest flooding, failure to hatch, and starvation (Lockwood et al. 1997). *For more information, see Nesting Habitat section.*



Photo by Ecostudies Institute

Nest success is the proportion of nests that successfully fledge young. Nest success values reported here were calculated using Mayfield techniques, unless otherwise stated; the Mayfield method relies on calculating daily nest survival probabilities and avoids the assumption that each nest was observed during the entire nest cycle (Mayfield 1975). Lockwood et al. (2001) show evidence of spatial and temporal variation in nest success for nests in subpopulations A, B, and E monitored from 1996–1999. For example, by year, nest success varied from a minimum of 0.12 in 1999 and a maximum of 0.40 in 1998. By subpopulation, E had the highest nest success at 0.53, whereas population A was only 0.12. Subpopulation B had a large enough sample size to examine early and late season nest success, defined as hatching before or after June 1, where early season nest success was 0.26 and late season nest success was 0.10 (Lockwood et al. 2001). Baiser et al. (2008) report nest success of 0.18 (95% CI 0.14–0.23,

subpopulations B and E) for 429 nests in ENP from 1996–2006, and nests initiated earlier in the breeding season had greater success rates than nests initiated later in the breeding season. Walters et al. (2000) report nest success for the first and second nests of the breeding season for the CSSS and found that success declined as the breeding season advanced; nest success for the first nest of the season ( $n=40$ ) was 0.43, while success for the second nest ( $n=11$ ) was 0.16 (Walters et al. 2000). Overall nest success from 2002–2008 in subpopulation E was reported as 0.40 for first nesting attempts, 0.19 for second nesting attempts, and 0.08 for third nesting attempts (Boulton et al. 2011). La Puma et al. (2007) report nest success as 0.41 ( $n=7$  nests) and 0.36 ( $n=14$  nests) in two adjacent plots in subpopulation E in 2002. Lockwood et al. (2001) also report fledging success for the CSSS, or the proportion of eggs that successfully fledge from the nest, as 0.42 (Lockwood et al. 1997).



Photo by Stephanie Romañach, U.S. Geological Survey

Virzi and Tafoya (2021) conducted a nest survival analysis from 2012–2020 on 272 nests across all subpopulations and found a nest success of 0.29 (95% CI=0.21–0.38). They also calculated apparent nest success on the same data set, which is calculated differently from the Mayfield method, and produced an estimated apparent nest success of 0.50. Apparent nest success is frequently biased high and not directly comparable to nest success calculated using the Mayfield method. For example, nests that fail before being found are not included in apparent nest success calculations, which can bias apparent nest success to higher values. Virzi and Tafoya (2021) showed that nest survival decreased with both age of the nest and day in the breeding season. Virzi and Tafoya (2021) also report that subpopulation did not have a substantial effect on nest success, except for in subpopulation D, where nest success was higher towards the end of the study period.

Lockwood et al. (2001) used demographic modeling to determine that both more frequent late-season nesting and low nest success constrained CSSS population growth. In this study, the percent of late-season nests (those hatching after June 1) observed in subpopulations A, B, and E from 1996–1999 was 10, 33, and 27%, respectively. Further, high nest success was critical for achieving positive population growth rates (Lockwood et al. 2001). Walters et al. (2000) developed a simple demographic model to assess the effect of multiple brooding on population dynamics. Their model indicates that the ability for a second breeding attempt is very important because the population growth rate ( $\lambda$ ) only exceeded one (indicating positive population growth) with multiple broods and high survival rates (55% survival for both adults and juveniles; although juvenile survival is presumed lower, the model did not separate adult and juvenile survival; see Table 3 of Walters et al. [2000] for more details).

## Genetics

There are nine recognized subspecies of the seaside sparrow (seven extant) that range along the Atlantic Ocean and Gulf of Mexico coasts of the United States (Nelson et al. 2000, Davis et al. 2021), although taxonomy, subspecific status, and synonymy of subspecies has changed throughout the years (Davis et al. 2021). For example, when the CSSS was originally listed under the Endangered Species Preservation Act, it was listed as a species, *A. mirabilis* (32 FR 4001), but later designated as a subspecies (*A. m. mirabilis*; Eisenmann 1973). There are other examples of changes to the nine recognized subspecies, including evidence for grouping the two western Florida subspecies (*A. m. juncicola* and *A. m. peninsulae*) and evidence for three genetic groupings along the Atlantic coast (as opposed to two, described in Davis et al. [2021]). Davis et al. (2021) analyzed mitochondrial DNA (mtDNA) and nuclear DNA and found that there were five to seven genetic clusters of seaside sparrows, which represent distinct population segments for conservation and management. They found three groups along the Gulf of Mexico coast, two to three along the Atlantic coast (with low to moderate levels of genetic differentiation among Atlantic coast groups), and a sixth (or seventh) group that represents *A. m. mirabilis* from southern Florida, although data are too limited for the southern Florida group to draw conclusive results. Davis et al. (2005) also found that *A. m. mirabilis* grouped most closely with individuals from the Atlantic coast and that *A. m. mirabilis* showed low levels of genetic diversity. Nelson et al. (2000) report that the CSSS is a part of the Atlantic matrilineal clade of seaside sparrows along the Atlantic coast of the U.S. and divergent from other Gulf of Mexico seaside sparrows, using mtDNA.

Genetic diversity and connectivity were evaluated by Beaver et al. (2019) via mtDNA and nuclear microsatellite DNA markers from feather samples collected from 80 birds for five of the six CSSS subpopulations. The mtDNA showed all samples showed high levels of relatedness. Nuclear markers revealed low levels of inbreeding in three of the five subpopulations studied (Beaver et al. 2019).

### *A. m. mirabilis* showed low levels of genetic diversity

Gilroy et al. (2012b) did not detect evidence for Allee effects (decreases in individual fitness with declining population density) for the CSSS, using demographic analyses with nest survival data. They report that environmental variables best explained variation in nest survival across subpopulations, not population size or density, specifically seasonal water table depth (stage), the timing of nesting activity related to onset flooding, and to a lesser degree the number of days of flooding during the nesting period. Environmental variables influenced nest success in fundamentally similar ways for both small and large subpopulations (Gilroy et al. 2012b).

## Spatial Ecology

### *Territory and Home Range*

Male sparrows are highly philopatric (Lockwood et al. 1997, Dean and Morrison 1998, Pimm et al. 2002), and often show site fidelity to territory areas, both within a breeding season and from one breeding season to the next (Werner and Woolfenden 1983, Pimm et al. 2002). Males are reported to use overlapping territory areas for two successive years, and the use of the same territory area for three years in a row is documented (Werner 1975). Virzi et al. (2012) conducted a study using a sparrow song playback system, and their results suggest that sparrows employ a mixed strategy in selecting territories, incorporating philopatry and social conspecific cues. During the breeding season, males establish and defend territories (Werner and Woolfenden 1983) for mating, nesting, feeding, and sheltering. Establishment of territories usually begins in early February (Pimm et al. 2002). Territories typically do not overlap, but sometimes slight overlap occurs (Pimm et al. 2002, Cassey et al. 2007).

Territory and home range sizes, which represent the core use area and overall area of activity, respectively, are generally larger in small CSSS subpopulations with lower sparrow densities compared to larger CSSS subpopulations (Virzi et al. 2016). Additionally, single males (called “floaters”) are reported to have larger territory and home range sizes compared to mated males, especially in small subpopulations; single males are also more common in small subpopulations (Virzi et al. 2016). Virzi et al. (2016) report CSSS territory sizes and home ranges calculated using a minimum of 10 GPS points gathered over multiple days for each individual (n=373 males from all six subpopulations from 2006–2015). The territory represents the core area where the animal is located and corresponds to an area with a 50% probability of locating the animal in a given survey (50% kernel density estimate or isopleth), whereas the home range (95% kernel density estimate)



Photo by Danny Bales

is the area where an individual spends a majority of its time (Virzi et al. 2016). Virzi et al. (2016) report smaller and less variable male territory sizes in larger CSSS subpopulations (B, E; mean=1.7 ha, SE=0.1) compared to small subpopulations (A, C, D, F; mean=10.3 ha, SE=1.5). Territory sizes also varied substantially between paired (mean=6.7 ha, SE=1.0) and unpaired (mean=14.6 ha, SE=3.1) males in the small subpopulations, especially for subpopulation A. Similarly, the overall home range size was smaller and less variable in the large subpopulations (B, E; mean=7.2 ha, SE=0.5) compared to small subpopulations (A, C, D, F; mean=42.1 ha, SE=5.6), and paired males had smaller home ranges than single males in the small subpopulations (A, C, D, F; Virzi et al. 2016).

Werner and Woolfenden (1983) report territory sizes from 0.3–6.8 ha at *Muhlenbergia* prairie sites around Taylor Slough (n=52 territories, mean=1.4 ha) and from 0.3–6.6 ha in the Big Cypress area (n=10 territories, mean=3.6 ha); territories were measured by observing and recording locations of marked individuals on field maps (Werner and Woolfenden 1983). Pimm et al. (2002) report a mean territory size of 2.36 ha, for territories measured from 1993–1996 by marking perch locations of singing males; the largest territory was 38 ha and the smallest territories were <1 ha (Pimm et al. 2002).

During the non-breeding season (e.g., August–February), sparrows increase the area they use compared to the breeding season territory (Dean and Morrison 2001), but they are primarily sedentary and stay in the area of their breeding territory (Dean and Morrison 2001), moving only up to a few kilometers from their breeding territories (e.g., see Elderd and Nott [2008]). The mean non-breeding season home range was reported as approximately 17.1 ha (range: 5.7–55.5 ha, n=45 [included adults, juveniles, and individuals of unknown age]) and determined via 90% probabil-

ity contours of adaptive kernel home range estimation by Dean and Morrison (2001); the size of non-breeding season home ranges varied substantially. The mean size of the home ranges during the non-breeding season were similar for male and female adult sparrows, and juvenile home ranges (mean=20.90 ha) were on average slightly larger than adult home ranges (mean=15.70 ha; Dean and Morrison 2001). Home ranges overlapped during the non-breeding season (Dean and Morrison 1998). Juveniles were also more likely to use multiple home ranges during the non-breeding season compared to adults.

## **Population Density**

Gilroy et al. (2012b) report that CSSS densities are lower in small subpopulations (e.g., A and C; 1–4 males/km<sup>2</sup>/yr) than in a larger subpopulation (e.g., E; 13–26 males/km<sup>2</sup>/yr). Although the effort and area are variable based on the study plot, Virzi et al. (2021) report density estimates for the CSSS from 2012–2020 that range from 0.4–4.4 birds/km<sup>2</sup> for subpopulation A (mean=1.8 birds/km<sup>2</sup>) and range from 1.8–30.9 birds/km<sup>2</sup> for subpopulation B (mean=11.6 birds/km<sup>2</sup> for areas with birds observed; data are from two different in-depth

study plots). Virzi et al. (2021) also report CSSS density estimates from 2017–2020 for subpopulation C that range from 5.2–12.6 birds/km<sup>2</sup> (mean=8.1 birds/km<sup>2</sup>), density estimates from 2011–2020 for subpopulation D that range from 2.5–14.4 birds/km<sup>2</sup> (mean=7.0 birds/km<sup>2</sup>), and density estimates from 2018 for subpopulation E across three in-depth study plots that range from 8.5–15.8 birds/km<sup>2</sup> (mean=11.4 birds/km<sup>2</sup>). Pimm et al. (2002) found higher sparrow densities in areas of high muhly grass.

## **Juvenile Dispersal**

Data from resighted or recaptured juveniles from hatching locations show a mean dispersal distance of 577 m (SD=980, n=15) and that juveniles are capable of dispersing distances greater than 1 km (natal dispersal; Lockwood et al. 2001). Lockwood et al. (2001) report that they never documented movement of a banded sparrow between subpopulations, although genetic data suggests some movement between subpopulations (Beaver et al. 2019). Werner (1975) reports that one male established a territory 400 m from the location where it was banded as a fledgling. Boulton et al.

(2009) observed eight inter-subpopulation movements, three made by juvenile birds and the rest by adult birds (observed via capture, mark, and resight data from three subpopulations from 1997–2007). Virzi et al. (2012) report a distance in subpopulation C of 984 m between the location where a CSSS was marked as a nestling and the area where it established a territory as an adult. Sparrows usually settle in an area within 6 months of fledging (Dean and Morrison 2001).

Van Houtan et al. (2010) modeled dispersal and found that, for adult and juvenile sparrows, most birds travel

short distances, but on rare occasions some birds travel long distances. Juvenile dispersal (natal dispersal) was commonly farther compared to adult movement. For adults, half of the individuals were recaptured within 211 m of their previous breeding location. The model showed that 50% of juveniles dispersed almost 1.34 km in their first year. There were nine cases of CSSS recaptures more than 10 km from the original location, five beyond 20 km, and two beyond 30 km (the last representing almost the entire length of their distribution). Eight birds were documented moving to subpopulations other than their natal subpopulation, and three of those individuals flew across Shark River Slough (Van Houtan et al. 2010).

Dean and Morrison (2001) applied color bands and radio-transmitters to investigate non-breeding season movement in subpopulation B during two non-breeding seasons (1997–1998 and 1998–1999). They found that juveniles moved around broadly after independence but switched to sedentary movement behavior at the onset of molt; all of the sparrows stayed sedentary

throughout the rest of the non-breeding season and had home ranges that were larger than breeding season territories. Longer distance movements for juveniles occurred more frequently at the beginning of the non-breeding season, in August, and long distance movements for juvenile sparrows that represented the upper 10% and 5% of the distribution of movement lengths between locations were 345 and 519 m, respectively. Juveniles made more frequent and longer distance movements compared to adults, and in one case a juvenile sparrow moved 4.92 km between locations. Of the six color-banded individuals that represented recruits during the study period, they were observed 206, 407, 447, 706, 1673, and 2702 m from where they were originally banded as nestlings or fledglings. Long distance movements were always abruptly when individuals came into contact with the margins of the prairie habitat (Dean and Morrison 2001). The mean rate of travel for radio-marked sparrows in the first year of the non-breeding season study was 6.4 m/hr (n=20 birds; Dean and Morrison 1998).

Photo by Ecostudies Institute



## Adult Movement

Adult sparrows exhibit properties of sedentary species, are faithful to breeding locations, and generally stay within 1 km of their breeding grounds (Lockwood et al. 2001). However, some birds may make exploratory movements away from the vicinity of their breeding territories and may occasionally relocate their territories and home ranges before resuming a sedentary movement pattern (Dean and Morrison 2001). It is reported that the area that a typical CSSS travels during its lifetime is under 50 ha, and that the CSSS does not move far from breeding areas during the non-breeding season (Pimm et al. 2002). For example, sparrows were found a mean distance of 277 m (n=14 birds, range: 77–986 m) from the location where they were banded in the preceding breeding season (Pimm et al. 2002). Lockwood et al. (1997) report that three males nest-

Owing to their high site fidelity, birds are reported to use areas that may represent unsuitable habitat if these areas were previously suitable, or if they border areas of densely populated suitable habitat (Jenkins et al. 2003b). Additionally, low dispersal capabilities can prevent birds from re-colonizing suitable habitat areas where CSSS numbers were previously depleted (Jenkins et al. 2003b).

In the Dean and Morrison (2001) study, where they applied color bands and radio-transmitters to investigate non-breeding season movements in subpopulation B, adult sparrows were relatively sedentary, and non-breeding season home ranges comprised the area of the breeding season territory. They found that longer distance movements by adults were rare, and often

**Owing to their high site fidelity, birds are reported to use areas that may represent unsuitable habitat if these areas were previously suitable, or if they border areas of densely populated suitable habitat.**

represented the diameter of the average home range for the CSSS. Long distance movements for adult sparrows that represented the upper 10% and 5% of the distribution of movements between locations were 276 and 362 m, respectively. The longest distance movement by an adult was from a female that traveled 5.99 km between locations. They also report

ed a mean distance of 40 m from the location of their nests the previous year. Lockwood et al. (2001) report that adult sparrows moved a mean distance of 212 m (SD=131, n=30) from the location where they were banded in a previous year, and most of the birds traveled less than 300 m. Because territories are an average of 2 ha (representing a circular diameter of 160 m), this means most of the birds do not adjust the locations of their territories across years, and if they do, only small spatial deviations are present. Virzi et al. (2012) report the movement of 128 m from one year to the next for a marked adult male CSSS in subpopulation C. Within a breeding season, nests from second clutches were 8–95 m from the location of the first clutch in the same year (Lockwood et al. 1997). Werner (1975) reports a female caught 940 m from its original capture site.

a long-distance movement by a banded adult female that was resighted approximately 4 km from its previous territory location. Longer distance movements were rare; birds often returned to their previous home range, and the movements were terminated when birds reached the end of the short hydroperiod marl prairie habitat. However, they recorded three individuals that did not return to the location of their previous home ranges and settled in new home ranges 450, 6,000, and 7,000 m away (Dean and Morrison 2001). Dean and Morrison (1998) found a male that nested in one area and then moved during the same breeding season to establish a new territory about 3 km away. Following molt, adults displayed sedentary movement behavior (Dean and Morrison 2001).

Virzi and Tafoya (2021) observed a pair of birds leave the location of a failed nesting attempt, an apparently rare occurrence for this species, in an in-depth study plot in subpopulation B in 2020; the cause of the nest failure was presumed predation by a nocturnal mammal. The pair was observed near the location of the nesting attempt after the nest failure, but one week later the female left the area. The male sang for three weeks at the location and then departed. The male was observed just over three weeks after the nest failure event 3.3 km from its original territory in another area of subpopulation B where numerous other sparrows were nesting (Virzi and Tafoya 2021).

Restoration of the Everglades is expected to change the location of suitable sparrow habitat (RECOVER 2020). Although the CSSS has been documented to move long distances, these are reported rarely, and long-distance re-colonization has not been documented (Sustainable Ecosystems Institute 2007). The inability of sparrows to re-colonize habitats as they become suitable may pose challenges for population persistence and recovery.

Translocation could help mediate the generally low dispersal capabilities observed in the CSSS and has been considered as a potential emergency conservation measure (Slater et al. 2009). Translocation has been implemented successfully for the conservation of another sparrow in Florida, the Florida Grasshopper Sparrow (*Ammodramus savannarum floridanus*; USFWS 2019). If translocation were considered for the CSSS, further study might include: 1) analysis to understand a threshold population size that would trigger translocation, 2) the population size that could pose a risk to the donor population, and 3) age structure for translocation (USFWS 2019). For Florida Grasshopper Sparrow translocations, two developmental stages were released, juveniles soon after reaching nutritional independence ( $\geq 40$  days post-hatching) and pre-breeders, just prior to the development of territorial behavior. Additional considerations are suitable habitat for translocation, other risks at the translocation site (e.g., predators, disease), and risk tolerance of the decision maker.

## Flocks

Flocks are observed at the end of the breeding season, are generally comprised of two or more individuals that stay close to each other (e.g.,  $\leq 5$  m), and show synchronization of behaviors from flock members (Dean and Morrison 2001, Pimm et al. 2002). These aggregates were typically comprised of several juveniles and 1–3 adults and usually only contained a few individuals, but a flock size of up to 26 individuals was observed (Dean and Morrison 2001); membership in flocks changed regularly (Dean and Morrison 2001). Pimm et al. (2002) report that flocks did not represent family groups and adults in the flock were not observed with their offspring but that nestmates may have been found in the same flock. Most of the adults observed in flocks were males (86% of 28 banded adults observed in flocks), and they held territories close to the observed flock (within approximately 200 m), but juveniles traveled much farther from the nests they fledged from, within about a 1 km range (Pimm et al. 2002). 

Photo by Ecostudies Institute



# Background on the Cape Sable Seaside Sparrow

*Ammospiza maritima mirabilis*



Only found in

## South Florida

and does not migrate

### 6 subpopulations

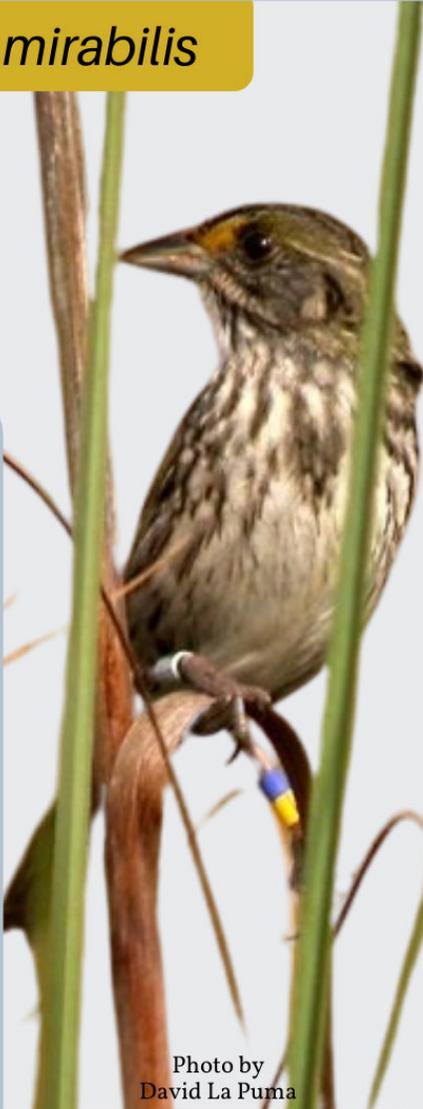
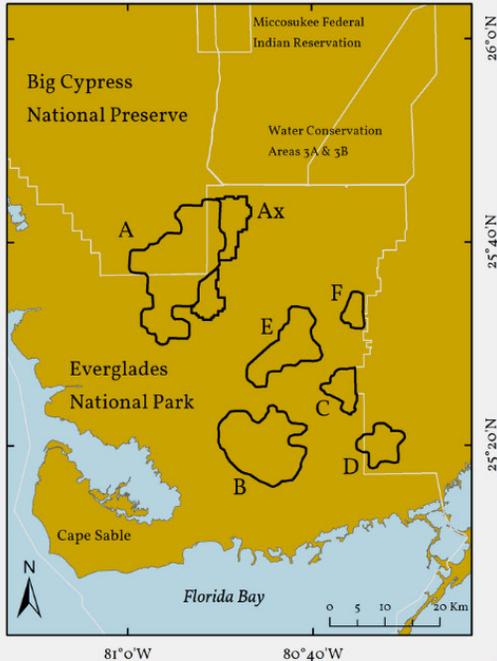
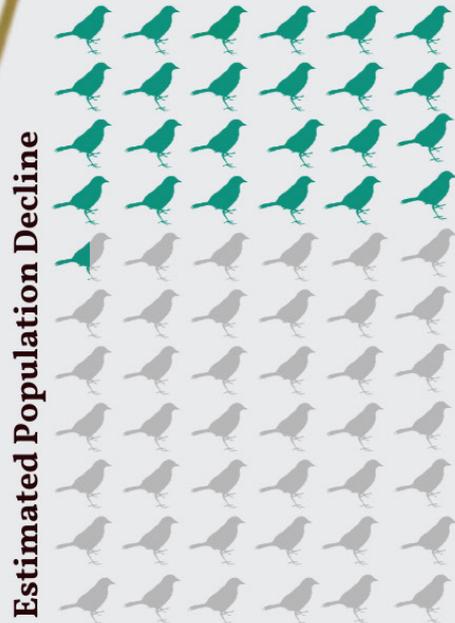


Photo by David La Puma



The Endangered Species Preservation Act lists it as **Endangered Species**

**63%**  
6,600 birds in 1981  
2,448 birds in 2021\*



Estimated Population Decline

\*preliminary 2021 estimate = 100 birds

Occupies freshwater marl and mixed-marl prairie, which are **flooded 4-6 months** of the year



**Generalist diet** of insects, mollusks, spiders, and amphibians

### Declines in numbers due to:

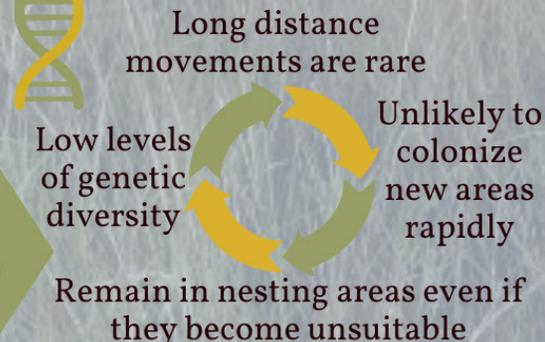
- increased flooding during breeding and non-breeding season
- over-drying, increased fire, woody encroachment
- decline in habitat suitability

## During dry periods...

- Breeding can occur from February to August
- Can have >1 brood/season and 3 eggs/nest
- Nests are not reused



## Challenges





## ECOLOGY

### Trophic Interactions

#### Diet

The CSSS is reported as mainly insectivorous during nesting (Stimson 1968). Howell (1932) examined stomachs of 15 specimens, which contained almost entirely insects and spiders, a few small mollusks and amphibians, and beetles, which were the most numerous. Diet analysis from fecal samples indicated that the CSSS diet consists of insects, snails, and a very small percentage of seeds or seed coats (only 4%; Dean and

Morrison 2001). The primary feeding tactic for the CSSS was reported as gleaning, a tactic that involves plucking items from the ground and vegetation; they also eat more active or flying insects (Kushlan et al. 1982). Lockwood et al. (1997) report that nestlings were fed insects and spiders (mainly Orthoptera, Lepidoptera, and Odonata), according to nest monitoring data from 24 nests in eastern ENP from 1996–1997. It

is reported that the CSSS is a dietary generalist, as the proportion of diet items (e.g., species of insect or spider) can vary over time (across years) and among sites, and the CSSS shifted the proportions of prey items with their availability (Pimm et al. 2002).

**Stomachs contained almost entirely insects, spiders, a few small mollusks, and amphibians. Beetles were the most numerous.**

#### Predation

The main documented cause of nest failure is predation (Lockwood et al. 1997, Pimm et al. 2002, Baiser et al. 2008, Boulton et al. 2009, Virzi et al. 2009, Slater et al. 2014), and apparent predation is reported to account for more than 75% of nest failures (Lockwood et al. 1997, Baiser et al. 2008). A complete list of nest predators is not known but may include raptors, small mammals such as raccoons (*Procyon lotor*), rice rats (*Oryzomys palustris*), and snakes, including non-native pythons (Ogden 1972, Lockwood et al. 1997, Post and Greenlaw 2000, Dean and Morrison 2001, Baiser et al. 2008). Predation by a short-tailed hawk (Ogden 1972) and predation on nestlings by ants have been observed (Werner and Woolfenden 1983). Dean and Morrison (2001) used radio-telemetry to track movements and habitat use of 45 sparrows (both juvenile and adult) during two non-breeding seasons and

observed five mortalities, four of which were caused by predation. Three predations showed evidence of avian predation, while the fourth predation was by a cottonmouth in which the snake consumed the bird and the transmitter (Dean and Morrison 2001). It is possible that non-native tegus may become nest predators for the CSSS if tegus continue to broaden their range (Mazzotti et al. 2015). Pimm et al. (2002) report average risk of predation per day to be 0.034 (range: 0–0.118). Predation pressure was also found to increase after the onset of the rainy season, and high water in general caused an increase in predation pressure independent of timing (Pimm et al. 2002).

**Predation increases with high water levels.**

## Habitat

The current habitat of the CSSS is primarily the freshwater marl prairie (Davis et al. 2005, Elder and Nott 2008), a short hydroperiod community that is seasonally inundated during the wet season generally for 4–6 months out of the year (Davis et al. 2005) but can range from 3–7 months (Nott et al. 1998, Hanan et al. 2010) or even 2–9 months as determined by the Natural System Model (NSM; Davis et al. 2005). The marl soils of this habitat are created by benthic mats of calcitic algae (Davis et al. 2005). The marl prairie is situated on an elevation gradient between deeper marsh and drier scrub and forest habitat (Lockwood et al. 2003, Elder and Nott 2008), which can be described as an intermediate zone between permanently flooded sloughs and drier pine-dominated habitat (Walters et al. 2000). Marl prairie is composed of a diverse community of grasses and forbs and frequently dominated by grasses, sedges, and rushes (Walters et al. 2000, Davis et al. 2005), in particular *Muhlenbergia* (Davis et al. 2005, Elder and Nott 2008). Other common species in the marl prairie include *Schizachyrium rhizomatum* (Florida little bluestem, hereafter “*Schizachyrium*”), thin stands of *Cladium jamaicense* (hereafter “*Cladium*” or “sawgrass”; note that sawgrass is a sedge), and *Schoenus nigricans* (black-topped sedge, hereafter “*Schoenus*”; Fletcher et al. 2000, Davis et al. 2005, Elder and Nott 2008, Sah et al. 2015). The CSSS is also found in the mixed-marl prairie habitat, which can be wetter than marl prairie, but is still on the dry end of the hydroperiod scale in the Everglades and contains similar species such as *Muhlenbergia*, *Cladium*, and *Schoenus*.

Marl prairie communities have a greater diversity of plants than the adjacent deeper-water marsh (Ross et al. 2004, Sah et al. 2009, 2015). There are more than 160 species documented in CSSS habitat (Ross et al. 2004). Grasses and graminoids are often short (e.g.,

less than 1 m high; Werner and Woolfenden 1983, Nott et al. 1998), clumped, and moderately dense (Werner and Woolfenden 1983, Stevenson and Anderson 1994), and there is open space that allows for ground movement (Stevenson and Anderson 1994). When hydroperiods are slightly longer, taller grasses and sedges dominate (e.g., sawgrass), and when hydroperiods are slightly shorter, the prairie remains only when fire pushes back encroachment by woody species. Marl and mixed-marl communities experience intermediate disturbance including flooding, drying, and fire (Kushlan et al. 1982), are heterogenous, and are spotted with occasional trees and tree islands, solution holes, and rocky outcrops (Werner and Woolfenden 1983). Higher-elevation areas can support tropical hammock tree islands within marl prairies (Davis et al. 2005).

**When hydroperiods become longer, taller grasses and sedges dominate, and when hydroperiods are shorter, the prairie remains only when fire pushes back encroachment of woody species.**

Vegetation community composition in the marl prairie wetland is linked to hydrology, where vegetation groups are situated along a hydrologic gradient based on hydroperiod (Lockwood et al. 2003, Davis et al. 2005, Ross et al. 2006, Sah et al. 2015). Areas with 1–2 month hydroperiods are dominated by *Schizachyrium*, areas with 3–5 month hydroperiods are dominated by muhly grass, and areas with 6–8 month hydroperiods are dominated by sawgrass (Davis et al. 2005). More specifically, sites where sparrows were observed had the following vegetation community groups, named by their dominant vegetation and listed from shortest to longest vegetation-inferred hydroperiod (driest to wettest): *Muhlenbergia* wet prairie (WP), *Schoenus* WP, *Schizachyrium* WP, *Cladium* WP, *Paspalum-Cladium* marsh, *Cladium* marsh, *Cladium-Rhynchospora*

marsh, *Rhynchospora-Cladium* marsh, and *Eleocharis-Rhynchospora* marsh; vegetation-inferred hydroperiods overlap for many of the vegetation groups (Lockwood et al. 2003, Ross et al. 2006). The percent of sites where the CSSS was observed generally followed this hydrologic gradient from shortest to longest vegetation-inferred hydroperiod, with the greatest percentage of sites with birds at WP community types (concentrated in the eastern half of the CSSS range), from *Muhlenbergia* WP, *Schoenus* WP, *Schizachyrium* WP, and *Cladium* WP, and the least percentage of sites with birds at marsh community types (concentrated on the western and southeastern areas of the CSSS range), *Cladium-Rhynchospora* marsh, *Rhynchospora-Cladium* marsh, and *Eleocharis-Rhynchospora* marsh (Ross et al. 2006). The vegetation-inferred hydroperiod was determined by analyzing vegetation composition data at sites with known topography and water levels to establish a relationship between vegetation community groups and hydroperiod (Ross et al. 2006). Vegetation community groups were determined using detailed vegetation surveys and cluster analyses (Ross et al. 2006).

Kushlan and Bass (1983) described three habitats used by the sparrow: muhly prairie (1–6 month hydroperiod, *Muhlenbergia* is dominant, can also include areas dominated by *Schoenus*, *Cladium*, *Schizachyrium*, or *Rhynchospora*), mixed prairie (equivalent to sparse sawgrass community described in Werner and Woolfenden [1983], lacks *Muhlenbergia*, contains *Cladium*, *Rhynchospora*, and *Panicum*, has a longer hydroperiod compared to muhly prairie), and cordgrass marsh (*Spartina* is dominant). Lockwood et al. (2003) report that the CSSS is more common in areas with relatively short hydroperiods and where sawgrass is not dominant. Pimm et al. (2002) report that sites occupied by the CSSS were more likely to be muhly-dominated and species rich, whereas sites not occupied by the CSSS were more likely to be sawgrass-dominated with low species richness. Taylor (1983) found that the amount of dead biomass and total biomass were related to CSSS presence, where areas with a high amount of dead biomass (e.g., >550 g/m<sup>2</sup>) and a high amount of total biomass (e.g., >700 g/m<sup>2</sup>) were not occupied by the CSSS. However, Pimm et al. (2002) did not observe this trend using the same methodology. Pimm et al.

Photo by Lori Oberhofer, National Park Service



(2002) also found that sparrows were selecting habitats on a scale <50 ha within marl prairies.

Ecologically, the CSSS and the Dusky Seaside Sparrow (*Ammodramus maritimus nigrescens*; extinct) are the only seaside sparrows reported to inhabit freshwater wetlands (Walters et al. 2000); however, the CSSS was historically found in other vegetation types, including brackish areas (Werner and Woolfenden 1983). The CSSS was originally recorded in *Spartina* (cordgrass; Howell 1932). Nicholson (1928) reports the CSSS in *Spartina* and *Distichlis* habitats. Stimson (1956, 1968) reports the CSSS in brackish habitats including *Spartina* marshes. Stimson also observed the CSSS in *Cladium* (Stimson 1956, 1968). Sparrow habitat surveyed from 1970–1975 included four distinct vegetation types: clumped *Spartina* prairie (only on Cape Sable), non-clumped *Spartina* prairie (a narrow band between the Big Cypress area and mangrove swamps that has since been replaced by vegetation less desirable to sparrows), sparse *Cladium* prairie (between the Big Cypress area and Shark River Slough and edges of Shark River and Taylor Slough), and *Muhlenbergia* prairie (primarily east of Shark Slough in ENP; Werner and Woolfenden 1983).

Dean and Morrison (2001) radiomarked 45 individuals during the non-breeding season from 1997–1999 in subpopulation B and recorded the CSSS in open muhly and sawgrass prairie habitats. Pimm et al. (2002) also recorded habitat use during the non-breeding season (wet season), and within the marl prairie habitat the CSSS was observed in thick sawgrass clumps (sawgrass sways) and occasionally perched in small trees in hammocks (near their nesting location), saw palmettos, or on roadside vegetation.

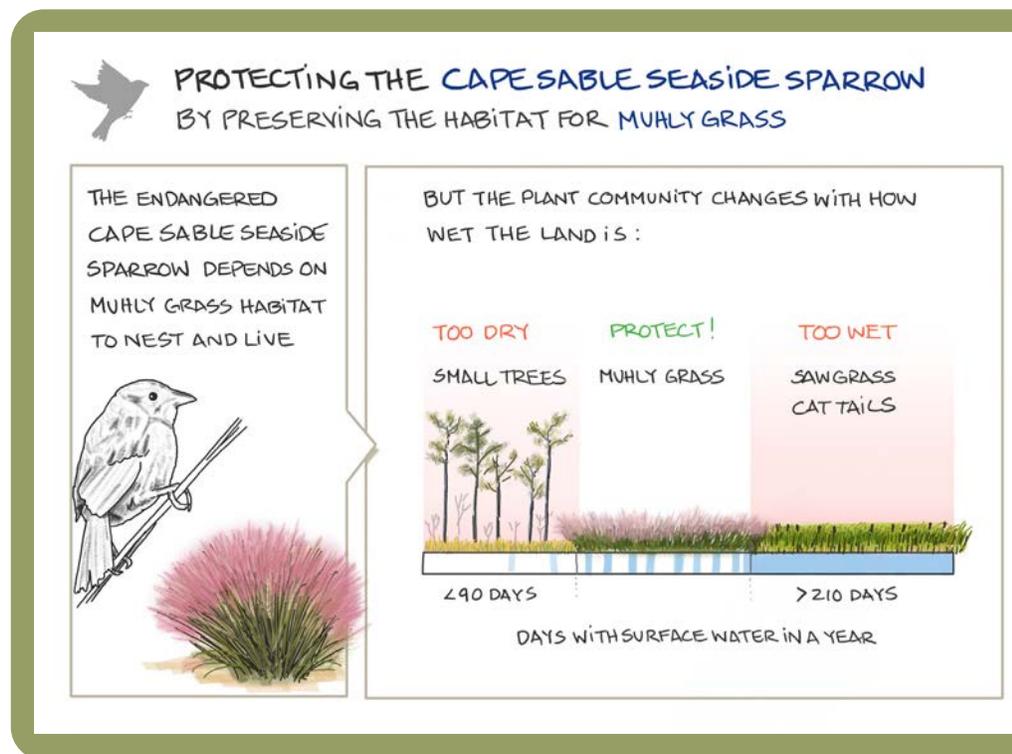


Image provided courtesy of U.S. Fish and Wildlife Service

## Critical Habitat

The most recent revision on the designation of critical habitat was in 2007, where current critical habitat designated for the CSSS includes subpopulations B–F (72 Federal Register 214 [6 November 2007], pp. 62735–62766). Critical habitat for the CSSS is located within ENP and the neighboring state-owned lands (subpopulation D crosses the border between federal

and state lands). Critical habitat for subpopulation A was previously proposed (71 Federal Register 210 [31 October 2006], pp. 63980–64002) but not included in the 2007 designation (72 Federal Register 214 [6 November 2007], pp. 62736–62766).

## Nesting Habitat

The CSSS builds nests close to the ground in clumps of grass, often in *Muhlenbergia* or other graminoid vegetation (Werner 1975, Lockwood et al. 1997, Davis et al. 2005) such as *Cladium* (Werner and Woolfenden 1983) or *Panicum* (Stimson 1968). Elder and Nott (2008) compared known breeding observations of the CSSS from 1993 in subpopulation A and report sparrows breeding in cells with a minimum of 55% preferred vegetation and at least 10% open area. Pimm et al. (2002) found that sparrows will select nest sites with high cover of muhly, *Rhynchospora* spp., and *Schizachyrium*, litter, and high effective and maximum vegetation height relative to habitat available within the territory. Nests are both cupped and domed shaped (Werner and Woolfenden 1983, Pimm et al. 2002), woven into clumps of vegetation, well-hidden (Post and Greenlaw 1994), and built away from standing shrubs and trees.

Lockwood et al. (2001) report that the CSSS nests near the ground between 16 and 21 cm above the soil surface and that mean nest height was lower for early vs. later season nests (before vs. after the onset of summer rains in June). Mean nest height earlier in the breeding season was 17 cm off the ground, while mean nest height later in the breeding season was 21 cm (Lockwood et al. 2001). Mean nest height also varied by year, where mean nest height was lower during the 1996 and 1997 breeding seasons (16 and 15 cm, respectively), and higher during the 1998 and 1999 breeding seasons (21 and 19 cm, respectively). Werner and Woolfenden (1983) report that nests near Taylor Slough were 6 to 37 cm above substrate, with a mean height of 16 cm (n=16 nests).

Pimm et al. (2002) report microhabitat characteristics for nest sites and territory areas associated with 74 nests in subpopulation B. Four dominant species were identified in close proximity to CSSS nests (within 1 m<sup>2</sup> quadrat): *Cladium*, *Muhlenbergia*, *Rhynchospo-*

*ra* spp., and *Schizachyrium*. The CSSS selected nest sites with high muhly grass, *Rhynchospora* spp., and *Schizachyrium* coverage, higher percent litter cover (25–50%), and high vegetation heights relative to the habitat within the territory. Greater percent cover of muhly grass and higher maximum vegetation height were also associated with successful nests. The density of males increased with greater percent coverage of muhly grass and higher maximum vegetation height in the territory. This study showed that although the CSSS chose territories with habitats that were dominated by muhly grass and were relatively species rich, once a territory was established, areas with higher vegetation height (e.g., sawgrass is used for perching) were used for nest placement (Pimm et al. 2002).

Photo by Ecostudies Institute



## Habitat Avoided

There are numerous habitats found near short hydroperiod freshwater marl prairie communities that are not used by the CSSS. Although the sites where sparrows are present may include intermittent cypress domes (*Taxodium* spp.), hardwood hammocks, and tree islands (Fletcher et al. 2000), the CSSS avoids areas containing shrubs and trees (Davis et al. 2005), especially during nesting (Kushlan and Bass 1983, Werner and Woolfenden 1983, Curnutt et al. 2000). Trees and shrubs can serve as refugia for CSSS predators such as raptors (Elder and Nott 2008). Sparrows avoid habitat that is within 40 m of hardwood or pine vegetation types (Lockwood et al. 2001, Jenkins et al. 2003a, 2003b) and do not nest near woody vegetation (Pimm et al. 2002). Using 235 sites of known CSSS nests and aerial photography, Jenkins et al. (2003a) found very few nests within 29 m of a bush. They also found fewer nests located less than 87 m from a bush compared to 235 random points in the same study areas (Jenkins et al. 2003a).

Areas that are too wet are also avoided because they lack required structure for CSSS nesting or foraging (Lockwood et al. 2001, Jenkins et al. 2003a, 2003b). Communities identified that the CSSS did not use included longer hydroperiod habitats such as very tall and dense *Cladium* prairies (Kushlan and Bass 1983,

Werner and Woolfenden 1983, Nott et al. 1998, Pimm et al. 2002), coastal spikerush (*Eleocharis* spp.) marshes (Werner and Woolfenden 1983, Nott et al. 1998, Pimm et al. 2002) near mangrove swamps, extensive *Typha* monocultures (Kushlan and Bass 1983, Werner and Woolfenden 1983), black needlerush marsh dominated by *Juncus* (Kushlan and Bass 1983), and slough prairies (Werner and Woolfenden 1983); vegetation that is too tall and too dense, or too short and too sparse, was not occupied by the CSSS (Werner and Woolfenden 1983). Pimm et al. (2002) report that sparrows do not occupy habitat dominated by sawgrass (*Cladium*) or that is species-poor, and sparrows do not cross forested habitats or go into the interior of tree islands, even though these habitats are imbedded in or adjacent to marl prairie habitat.

Dean and Morrison (2001) radiomarked 45 individuals (both adults and juveniles) during the non-breeding season from 1997–1999 in subpopulation B, and they report that birds avoided large tree islands, tree lines, densely forested areas, and mangroves but did not avoid isolated tree islands within an open prairie matrix. They also report the CSSS in old agricultural areas that were not shrub-dominated, however old-grove agricultural areas with abundant shrubs were not used (Dean and Morrison 2001).

Photo by Big Cypress National Preserve

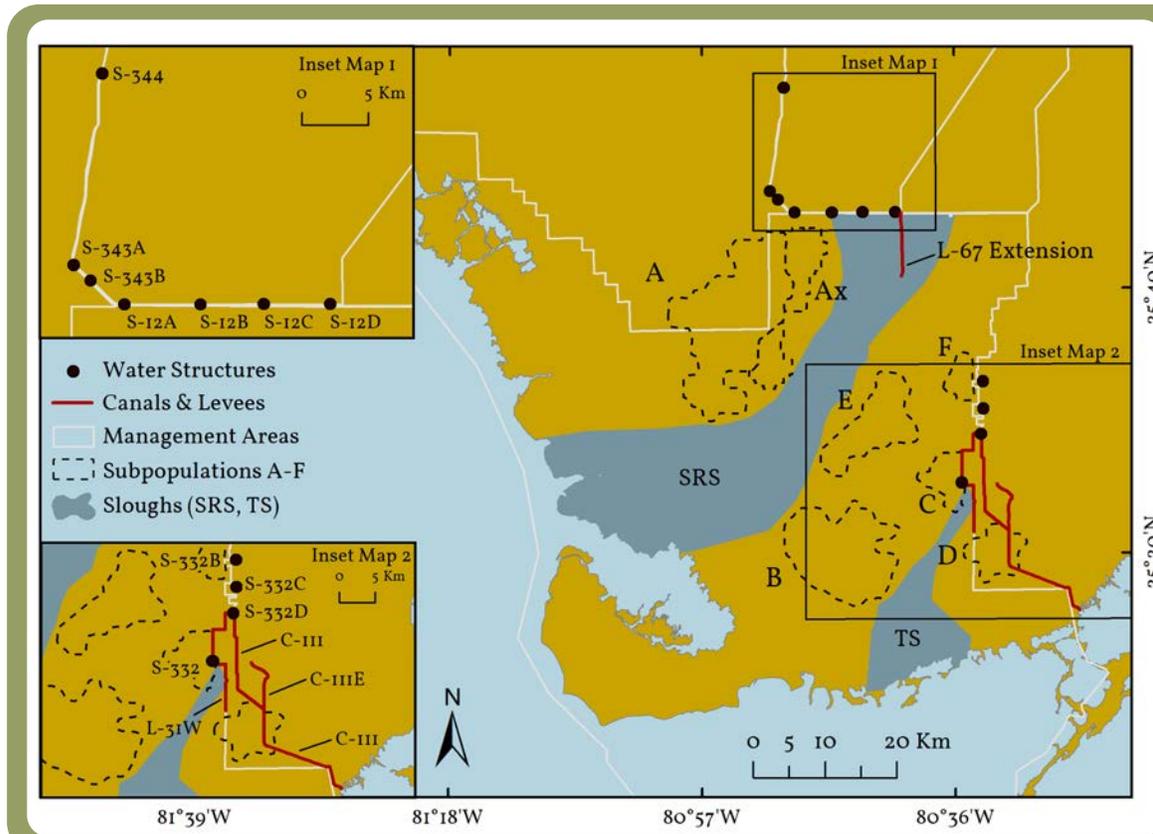


## Habitat Threats

Habitat changes and reduced habitat suitability are the crux of the threats to the CSSS, as determined by a Conservation Committee formed by the American Ornithologist's Union (Walters et al. 2000). The main human-induced changes to marl prairie are related to water management, agriculture, and urbanization (Kushlan et al. 1982, Davis et al. 2005, Elderd and Nott 2008). All of these factors have led to stressors on the marl prairie, including reductions in spatial extent and connectivity, extended hydroperiods and dry season water reversals in some areas, shortened hydroperiod and increased drought severity in other areas, and conditions in some areas that allow for intense fires and the invasion of non-native trees (Davis et al. 2005). In the early 2000s, spatial coverage of the marl prairie was estimated to have decreased by 30% since pre-drainage (Pimm et al. 2002). Human-induced changes to marl prairie habitat, such as water management, can have

direct effects on the CSSS (e.g., nest loss from flooding, mortality from high-intensity dry season fires) or indirect impacts (e.g., changes in habitat, vegetation, or long-term fire regimes).

The marl prairie habitat in the southern Everglades is comprised of two main areas separated by Shark River Slough, one area located west (subpopulation A) and one area located east (subpopulations B–F) of Shark River Slough (Curnutt et al. 1998, Nott et al. 1998). The marl prairies are at a higher elevation than the deeper-water slough (Davis et al. 2005). The marl prairie in southern Florida once extended farther northwards (Davis 1943). In the mid-1960s, Water Conservation Area 3A (WCA 3A; Fig. 1) was created via a system of levees which formed a hard division between ENP and the WCAs to the north.



**Figure 2.** Water structures, canals, and levees are located in and around the Cape Sable Seaside Sparrow (*Ammodramus maritimus mirabilis*) subpopulations (A–F; subpopulation delineations derived from USFWS [2020]).

The western area of marl prairie is located in ENP and extends west into portions of BCNP (subpopulation A; Nott et al. 1998). The east side of the western area of marl prairie is flanked by Shark River Slough. The northern portions of subpopulation A and Ax have the highest elevations in the range of the sparrow (USGS 2011). There are four gated spillways, the S-12 structures (S-12A–D; Fig. 2), on the western side of WCA 3A that control the release of water from WCA 3A into ENP; the location of these structures results in a water flow that is diverted mainly to the western side of Shark River Slough and creates longer hydroperiods, reversing natural drying patterns in this area (Curnutt et al. 1998, Nott et al. 1998, Davis et al. 2005). Water that historically flowed through the area on the east side of Shark River Slough has been diverted west of Shark River Slough (Curnutt et al. 1998, Walters et al. 2000, Davis et al. 2005). In the late 1990s, the USFWS (1999b) concluded that continued existence of the CSSS was jeopardized, and one immediate emergency measure was to lower water depths during the breeding season to increase the number of nesting days for the completion of two nesting cycles in subpopulation A. Additionally, the L-67 extension, a levee running south from the WCA 3A and ENP border, limits water flow diverted west from traveling back eastward (Nott et al. 1998, Walters et al. 2000). This re-distribution of water flows leads to deeper water than what historically occurred on the western side of Shark River Slough and to an increase in the flooding duration of the western marl prairies compared to historic conditions (Nott et al. 1998).

## **Predicted Change**

The implementation of the Combined Operational Plan (COP) in August 2020 was the first major system-wide change in water management in 20 years (USFWS 2020). Water operations under COP will impact the entirety of the geographic range of the CSSS. One advantage of COP over historical operational plans is that it allows some flexibility such as closures and openings of the S-12 structures which directly impact subpopulation A immediately downstream

The eastern marl prairies are flanked by Shark River Slough on the western side, and the Atlantic coastal ridge, agricultural areas, and urban development on the eastern side (Curnutt et al. 1998, Nott et al. 1998); the agricultural and urban areas to the east are drained by a large network of canals (Nott et al. 1998). Taylor Slough is much smaller than Shark River Slough and is located in the eastern area of the marl prairie (Nott et al. 1998). Many of the eastern portions of the marl prairie (north-eastern ENP) became drier than they were historically, namely subpopulations E, F and most of C (Kushlan et al. 1982, Curnutt et al. 1998, Nott et al. 1998, Walters et al. 2000), attributed in part to the diversion of water westward to the S-12 structures (Nott et al. 1998, Walters et al. 2000, Davis et al. 2005). Water that would have flowed from WCA 3B into the eastern side of the Shark River Slough can now be moved westward (Walters et al. 2000). There are other structures in the eastern area that also affect the timing and flow of water, including the L-31W and C-111 canal systems and the S-332s (located along the eastern border of ENP near Taylor Slough; Nott et al. 1998, Armentano et al. 2006) that affect the amount of water entering Taylor Slough and adjacent areas and allow for flood protection of areas east of ENP (Armentano et al. 2006). The lower portions of subpopulations C and D became wetter than they were historically (Nott et al. 1998), owing to flood protection of the adjacent agricultural and urban lands (Nott et al. 1998) and additional water released into Taylor Slough in the southeastern area of ENP (Walters et al. 2000).

of those structures. Historically, seasonal closures of the S-12A, S-12B, S-343A, S-343B, and S-344, have improved hydrologic conditions to some extent and allowed sparrows to persist in subpopulation A (USFWS 2020), yet still conditions have remained too wet. After a few years under COP, water flow is expected to reduce hydroperiods in and around subpopulation A and improve nesting conditions.

Water flows expected from COP are anticipated to benefit sparrows (USFWS 2020). The COP is expected to yield discontinuous hydroperiods (the number of days water is above ground each year) that are more favorable to the CSSS and also toward the USFWS target (i.e., a 4-year average discontinuous hydroperiod range of 90–210 days, with no two consecutive years failing to meet this target). The USFWS dry nesting days target of 90 consecutive dry days between March 1 and July 15 (which is during the breeding season) is expected to improve under COP as well. Subpopulation A is expected to experience lengthened nesting season conditions. Subpopulation F should become wetter. Subpopulations B and C are not expected to experience major changes in meeting USFWS targets compared to existing conditions. Subpopulation D may have reductions in the number of years that it meets the discontinuous hydroperiod target. Subpopulation E, the second largest subpopulation, is expected to be the most impacted and it is expected that sparrows will move toward the eastern portion of the subpopulation area as hydrologic conditions become wetter and less suitable for sparrows at the western edge.

Interim Goals and Targets were developed to assess restoration progress using important indicators such as marl prairie habitat suitability. Using scenarios that include expected hydrologic conditions with planned restoration projects in place by 2026 and 2032 (for more details see RECOVER [2020]), increased water flow into northeast Shark River Slough is predicted to decrease marl prairie habitat suitability in portions of subpopulations D and E. However, habitat suitability is expected to increase in subpopulation C, subpopulation A and the western prairies, and in areas southeast of subpopulations E and F (RECOVER 2020).

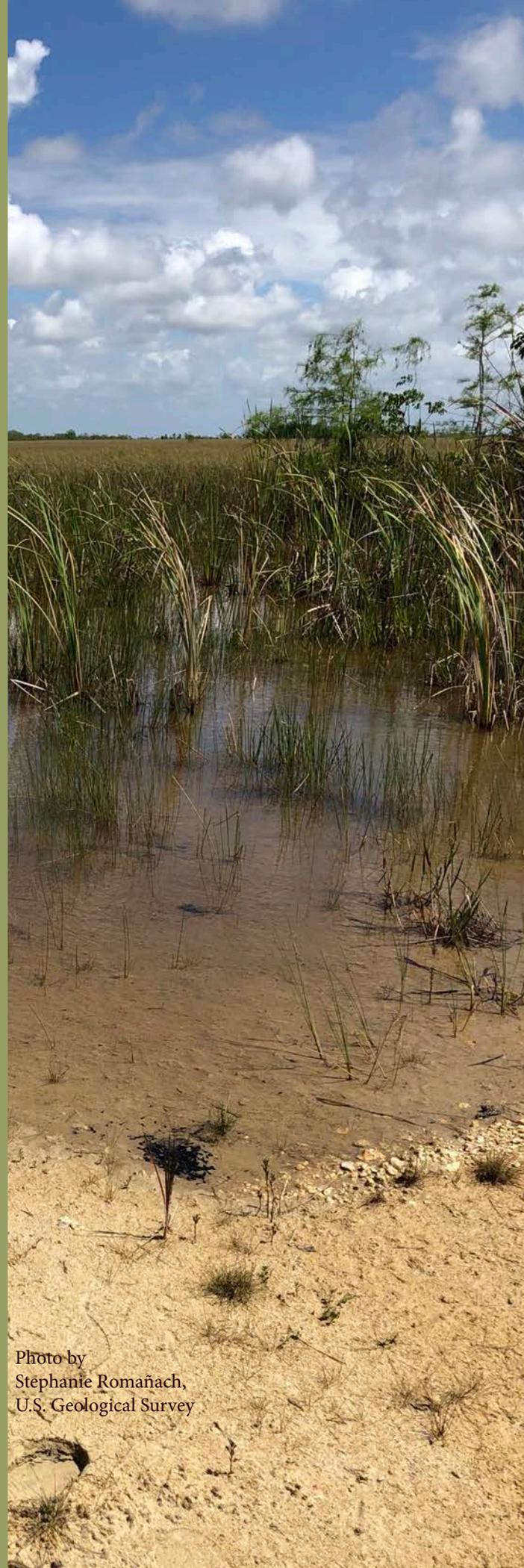


Photo by  
Stephanie Romañach,  
U.S. Geological Survey

## Hydrometrics

### Hydrology and Habitat

Hydrologic change, in the form of persistent flooding or drying beyond the hydrologic regime of the marl prairie, can alter vegetation communities (Curnutt et al. 1998, Nott et al. 1998, Jenkins et al. 2003a, Ross et al. 2006) and lead to habitat change that is unsuitable for sparrows (Jenkins et al. 2003a). Sah et al. (2020) show that Everglades vegetation communities, including the marl prairie, can shift in species composition with changes in hydrologic conditions in as little as 3–4 years (shift from prairie to marsh, or from suitable to unsuitable CSSS habitat), based on detailed vegetation surveys along transects that span the hydrologic gradient from marl prairie to slough over a 13-year period. Armentano et al. (2006) report vegetation shifts to more hydric (wetter) species in as little as 3–4 years along with wetter conditions, such as from *Muhlenbergia* dominated vegetation to *Cladium* dominated vegetation, or from *Cladium* dominated vegetation to *Eleocharis cellulosa* (spikerush). However, after conditions became less hydric (drier), the vegetation over a six-year period only partially changed back to less hydric species (Armentano et al. 2006). Not only has it been shown that it takes longer to shift from unsuitable CSSS habitat (e.g., marsh) back to suitable habitat (e.g., wet prairie), Sah et al. (2009) show that flooding after fire led to longer vegetation recovery times in addition to altered species composition recovery trajectories, especially when a site flooded immediately after fire. Overly wet conditions can cause breeding habitat degradation (Nott et al. 1998) and lead to submerged habitat conditions typical of slough habitat and areas dominated by thick stands of *Cladium*, *Eleocharis*, and *Rhynchospora* (Kushlan et al. 1982, Nott et al. 1998, Ross et al. 2003).

On the other end of the hydrologic spectrum, overly dry conditions can lead to increased fire frequency or woody encroachment (Pimm et al. 2002, Elderd and Nott 2008). Drought and shortened hydroperiods, in

the absence of fire, can increase hardwood (e.g., willow [*Salix caroliniana*]) and pinewood encroachment (Davis et al. 2005, Elderd and Nott 2008) into CSSS habitat, including encroachment of non-native trees (Davis et al. 2005). Non-native trees that encroach into marl prairie include species such as Melaleuca (*Melaleuca quinquenervia*), Brazilian pepper (*Schinus terebinthifolius*), and Australian pine (*Casuarina equisetifolia*), which can displace native plant communities (Davis et al. 2005). Shortened hydroperiods can also lead to higher intensity fires in marl prairie habitat (Gunderson and Snyder 1994). Woody encroachment is reported in some of the eastern subpopulations of the CSSS (east of Shark River Slough; Kushlan et al. 1982) in addition to high fire frequencies (e.g., in subpopulations C, D, F, 1982–1996; Curnutt et al. 1998).

Over the years, the USFWS has refined a discontinuous hydroperiod target and now uses a 4-year average range of 90–210 days to represent habitat suitability for the CSSS with no two consecutive years failing to meet this target (USFWS 2020). The USFWS (2020) found that subpopulation C met the target of 90–210 days in just over half of the years evaluated (1991–2019), while subpopulations B, D, E, and F met this target in approximately one third of the years, and A (including Ax) only met the target in two years. Fewer periods of time met the two consecutive years part of this target. It is important to note that this target measures only one element of habitat suitability for the sparrow.

Jenkins et al. (2003a) estimated changes in potential CSSS habitat from year to year using spectral signatures from satellite imagery at known CSSS nesting sites, coupled with filters to exclude areas within approximately 58 m from a bush and habitat patch sizes <2 ha (approximate territory size of the CSSS); they report that flooding reduced potential CSSS habitat in subpopulation A from 82 km<sup>2</sup> in 1992 to 7 km<sup>2</sup> in 1993

(an approximately 90% decrease). Jenkins et al. (2003a) show that potential habitat in subpopulation A started to recover slowly after 1995 but had not fully recovered in 1999, indicating that the negative effects of flooding on CSSS habitat persisted for much longer than the actual flooding. Jenkins et al. (2003a) also report that portions of subpopulation E were affected by flooding in 1993 and 1995, decreasing estimated available habitat from 68 km<sup>2</sup> in 1994 to 34 km<sup>2</sup> in 1995. Subpopulation D also experienced flooding in 1993 and declined in potential habitat from 30 km<sup>2</sup> in 1992 to only 3 km<sup>2</sup> in 1993 (Jenkins et al. 2003a).

Vegetation studies by Nott et al. (1998) show that an area downstream of the Taylor Slough drainage changed from a high diversity *Muhlenbergia* dominated community to a lower diversity *Cladium* dominated community (Werner 1975, Nott et al. 1998, Pimm et al. 2002). This area has also become wetter (fewer dry days) since 1980 due to the implementation of the S-332s at the boundary of ENP and Taylor Slough, which altered the hydrology downstream of the station to become wetter (Nott et al. 1998, Pimm et al. 2002). Areas upstream of the S-332s in the northern Taylor Slough drainage exhibited a shortening of hydroperiods, specifically subpopulation F and most of subpopulation C, attributed to the dredging of the L-31W canal (Kushlan et al. 1982, Curnutt et al. 1998, Nott et al. 1998).

The marl prairie west of the Shark River Slough changed from 1992–1996 from a drier vegetation type (muhly dominated and mixed marl prairie) to a wet vegetation type (sawgrass dominated), except for the highest elevation areas located in the farthest northwestern areas west of Shark River Slough (Nott et al. 1998). Sites in the northwestern area west of Shark River Slough stayed drier, while sites in the southeastern corner west of Shark River Slough became wetter (Nott et al. 1998, Pimm et al. 2002). Similarly, Ross et al. (2006) report that the muhly grass dominated communities west of Shark River Slough

were nearly absent when the vegetation was surveyed from 2003–2005.

Sah et al. (2020) evaluated changes in vegetation (and associated vegetation-inferred hydroperiod) from 2006–2019 along a series of transects that crossed through both marl prairie and slough habitats. Alterations in vegetation composition from water management followed trajectories in line with restoration objectives. These objectives include increased water deliveries to some areas of marl prairie east of Shark River Slough, particularly in the northeastern areas of ENP (e.g., subpopulation F) and areas west of ENP’s Main Park Road (e.g., northwest of subpopulation B) and concomitant shifts towards wetter vegetation types. Additionally, water delivery west of Shark River Slough (e.g., subpopulation A) was regulated (through the S-12 structures) to produce more consistent lower water levels, and vegetation in the northeastern portion of subpopulation A has shifted toward a drier type (Fig. 5; Sah et al. 2018, 2020). However, areas in the southeast (e.g., hS area; Fig. 5) and western portions of subpopulation A show wet vegetation types, marsh vegetation, and longer vegetation-inferred hydroperiods (Fig. 5; Sah et al. 2018). Despite these vegetation changes in subpopulation A in the northeastern portion towards a drier type and potentially more suit-

able for sparrows, Sah et al. (2018) also state that the overall hydrology of subpopulation A has not changed much from 2003 to 2017, and Beerens and Románach (2016) show that very little area of subpopulation A met the FWS target hydroperiod of 90–120 days from 2000–2015.

**The negative effects of flooding on CSSS habitat occur for much longer than the actual flooding.**

## **Hydrology and Breeding Impacts**

Hydrology can have numerous direct effects on the CSSS, including effects on reproduction. Flooding is reported to cause sparrows to cease courtship and nesting activities (Lockwood et al. 1997, 2001) and studies indicate that nest survival for the CSSS is related to hydrology (Lockwood et al. 2001, Baiser et al. 2008, Boulton et al. 2011). High water events during the breeding season (e.g., dry season water level reversals) are reported to limit CSSS nesting because they nest near the ground (Nott et al. 1998). Flooding during the breeding season can cause nest failure because of nest inundation (Nott et al. 1998, Lockwood et al. 2001). Nests that are located too low are at risk of flooding, but nests placed too high may not be supported well enough by the vegetation (vegetation too thin) or not protected well enough from predators or the weather (Walters et al. 2000). Gilroy et al. (2012b) found that variation in nest survival across all CSSS subpopulations was best explained by environmental variables, predominantly seasonal water level, and the timing of nest activity relative to onset of flooding. Nott et al. (1998) report that a substantial rise in water level in CSSS breeding areas can lower fecundity.

Lockwood et al. (1997) report that CSSS nesting does not begin when water depth is higher than 10 cm during the breeding season. Similarly, water levels above 10–20 cm are reported to flood CSSS nests and restrict breeding (Pimm et al. 2002). Lockwood et al. (2001) estimate that in locations where water levels are greater than 29 cm, almost 70% of active CSSS will fail because of nest flooding. Curnutt et al. (2000) report that surface water levels during the breeding season affected breeding potential, where breeding and nesting activity was interrupted when water levels exceeded 15 cm and led to failed nesting attempts. However, Dean and Morrison (1998) found that sparrows were able to begin nesting when water levels were greater than or equal to 10 cm, and the water depth at some nest sites was up to 20 cm. Additionally, they report successful nesting in late July and August at times with high water (Dean and Morrison 1998), indicating at least some

ability for the CSSS to nest during moderately high water conditions during the dry season.

It is generally reported that the length of the breeding season for the CSSS is related to the length of the dry season (Lockwood et al. 1997, 2001). The CSSS breeds in conditions that are not too dry, but also not too wet. It is reported that approximately 40 days of nearly dry conditions (without fire) are needed for successful CSSS nesting and 60 days are needed for the initiation of a second clutch (Pimm et al. 2002). The peak of the breeding season for the CSSS is during the dry period (approximately mid-March through mid-June), where the water table can be as low as 1 m below the surface (Pimm et al. 2002). Typically, the dry period in the marl prairie allows time for the rearing of two broods per season (Lockwood et al. 1997, 2001), although four broods are possible in rare cases. However, Boulton et al. (2011) report that drier conditions before and during the early nesting season delayed the initiation of nesting and limited clutch size; therefore conditions that are too dry can have adverse effects on the CSSS. The timing of initiation of breeding is associated with the amount of rainfall just before the breeding season, when nesting was initiated up to one month earlier in years with higher mean rainfall two months prior to breeding (Boulton et al. 2011). Drier conditions at the onset of breeding were associated with smaller clutch sizes, especially for first nesting attempts (Boulton et al. 2011), which may be driven by food availability. In general, the end of the breeding season for the CSSS is signaled by the beginning of the rainy season (Lockwood et al. 1997). Lockwood et al. (1997) report that even when drying occurred after high water during the breeding season and water levels substantially decreased, breeding activities did not resume for the CSSS (water levels rose to 18 cm in late June, then fell to 4 cm in mid-July). Boulton et al. (2011) report a negative association between annual nest survival and high average rainfall late in the breeding season.

## Hydrology and Occurrence

Flooding can also affect CSSS survival and population size. Nott et al. (1998) report that when water levels were greater than 10 cm during range-wide surveys in 1981 (Bass and Kushlan 1982) and from 1992–1997 (>3,000 total survey locations), singing birds were consistently not heard (Nott et al. 1998). Nott et al. (1998) also state that dramatic increases in water levels in CSSS breeding habitat can decrease local population size in subsequent years. Based on analysis of satellite imagery, Jenkins et al. (2003a) state that subpopulations that are not flooded maintain more sparrows relative to available suitable habitat (e.g., subpopulation B), and subpopulations that experienced flooding supported fewer birds (e.g., subpopulations A, D, E). The mechanism for CSSS population declines attributed to extended hydroperiod include lowered reproduction and unsuitable changes in vegetation (Walters et al. 2000).

In a study evaluating CSSS population numbers and hydrology from 2000–2015, the greatest number of sparrows were observed where mean subpopulation water depth values were between -25–50 cm and 16–18 cm water depth heterogeneity (i.e., SD of depth; Beerens and Románach 2016). From a detailed vegetation study in CSSS habitat (from 2003–2005), Ross et al. (2006) show that CSSS occupancy was close to 50% at sites with vegetation-inferred hydroperiods between 90–120 days and >30% at sites with inferred hydroperiods between 150–240 days. Last, occupancy was <20% at sites with inferred hydroperiods between 60–90 and 240–270 days, and the CSSS was very rarely observed at sites with inferred hydroperiods >270 days (Ross et al. 2006).



Photo by Ecostudies Institute

High water levels, extended hydroperiods, and flooding are attributed as causes for the population declines in CSSS subpopulation A (Nott et al. 1998, Walters et al. 2000, Pimm et al. 2002, Cassey et al. 2007, Elder and Nott 2008). In subpopulation A, long hydroperiods of greater than 300 days prevailed from 1993–1996, where water released during the dry season through the S-12A and S-12B structures from WCA 3A led to water level reversals (Nott et al. 1998, Walters et al. 2000). These high water levels in subpopulation A were also evident via analysis of satellite imagery, where in 1992 the area west of Shark River Slough contained large areas of dry marl prairie, but in 1993 and 1995 the same areas were covered in water. This inundation also occurred, but to a lesser extent, in both 1994 and 1996. Sparrow counts showed a drastic decline in subpopulation A beginning in 1993 (Nott et al. 1998, Pimm et al. 2002, USFWS 2020), with a decline of 84% from 1992–1993 (Curnutt et al. 1998, Pimm et al. 2002). Declines in subpopulation A because of high water was corroborated by Jenkins et al. (2003a). They report reductions in the area occupied by sparrows that were associated with reductions in potential habitat because of flooding in subpopulation A, for example from 1992–1993 (Jenkins et al.

1992–1993 (Jenkins et al.

**Subpopulations that are not flooded maintain more sparrows relative to suitable habitat available.**

2003a). Although the potential habitat in subpopulation A began to recover after 1995 and continued to recover slowly and consistently, the sparrow population showed little recovery and occupied only about 7% of the potential habitat available by the end of the study period in 1999 (Jenkins et al. 2003b).

Nott et al. (1998) evaluated trends in water levels in subpopulation A and found that in the wet years of 1993–1996, there were <40 dry days during the breeding season, with 1993 and 1995 containing zero dry days during the breeding season (Nott et al. 1998). Approximately 40 days is the minimum dry duration the CSSS requires to complete one breeding cycle (Lockwood et al. 1997). In 1992 when sparrows were abundant, 95% percent of the total area in subpopulation A was dry enough to support at least one brood (dry for >40 days) but showed a sharp decline starting in the wet year of 1993 (Nott et al. 1998). The causes for wetter conditions in subpopulation A from 1993–1996 were teased apart and mainly attributed to water releases from the S-12 structures, with a small contribution from increased rainfall (Nott et al. 1998).

Elder and Nott (2008) developed a spatially explicit individual-based model informed by field data for subpopulation A and report that an increase in mean water depth during the breeding season was associated with declines in final population size in subpopulation A and elevated quasi-extinction risk. The population declines and higher extinction risk occurred with as little as a 4 cm increase in mean water levels. An increase in mean water level by 12 cm resulted in a quasi-extinction risk >75%. The modeled population decline and increased extinction risk were related to lowered recruitment from nest abandonment and decreased breeding site availability from the elevated water levels (Elder and Nott 2008).

Declines in CSSS population numbers in areas besides subpopulation A are also attributed to high water levels. Declines in CSSS numbers in subpopulation D and the lower portions of subpopulation C have been associated with the S-332s upstream in Taylor Slough, increased flooding, conversion of suitable

Photo by Sabrina Diaz, National Park Service



**Sparrows tend to occupy sites with an estimated hydroperiod between 90 and 120 days.**

habitat to sawgrass dominated marshes, and inhibition of reproduction (Lockwood et al. 1997, Curnutt et al. 1998, Nott et al. 1998, Walters et al. 2000). Nott et al. (1998) report increased hydroperiods in areas of subpopulation D, attribute them to water releases from the S-332s, and also present evidence of vegetation change from *Muhlenbergia* to *Cladium* in these areas. Pimm et al. (2002) also attribute subpopulation D declines to higher water levels. A small but well-studied area of subpopulation C shifted from muhly-dominated habitat to sawgrass under wetter conditions south of the S-332 structures and experienced declines in sparrow population numbers (Pimm et al. 2002). Pimm et al. (2002) also attribute declines in the southern portion of subpopulation E to flooding. Even with the implementation of emergency management actions to lower water levels during the breeding season (Walters et al. 2000), sparrow numbers in some of the subpopulations, including A and D, did not show recovery from past declines by 2005 (Cassey et al. 2007), although recent range-wide survey efforts (since 2018) indicate that subpopulation D may be on a path towards recovery (USFWS 2020).

Declines in upper reaches of Taylor Slough (subpopulation F and most of C) are attributed to drier conditions from the dredging of the L-31W canal (Nott et al. 1998). The reduced habitat quality in C and F are due to reduced hydroperiods and resulting abnormally high fire frequency (Walters et al. 2000), which negatively affect the CSSS. Subpopulation F also contains high densities of native and non-native shrubs (Curnutt et al. 1998), which are unsuitable for the CSSS. Conditions that are too dry can create an increase in fire risk; if fires occur too frequently, or during the breeding season, they can negatively impact CSSS population numbers.

Using the North American Breeding Bird Survey data for similar grassland bird species, Curnutt et al. (1998) determined that the probability was very low

that the temporary local extinctions that occurred in subpopulations C, D, and F in the early 1990s were due to natural variation alone. These temporary location extinctions were more likely driven by other factors. In August of 1992, Hurricane Andrew may have contrib-

**Declines in CSSS numbers have been related to the release of water through water management structures, increased flooding, conversion of suitable habitat to sawgrass dominated marshes, and inhibition of reproduction.**

uted to the CSSS declines (via factors such as affecting the duration of breeding season and mediating habitat changes), but the spatial pattern of bird decline from 1992–1993 did not match the spatial pattern of areas hit hardest by the hurricane (Curnutt et al. 1998).

Using radiomarked individuals (n=45) in subpopulation B, Dean and Morrison (2001) recorded sparrows in open muhly and sawgrass prairie habitats. They found that the distribution of sparrows did not change drastically between the breeding and non-breeding seasons, but instead the selection of microhabitat characteristics changed with time and was related to hydrology. When water was higher in the non-breeding season, they observed sparrows using old agricultural areas that were not shrub dominated, isolated tree islands within the open prairie matrix, low shrubs, and tall sawgrass areas (sawgrass sways). However, the CSSS did not perch above 1 m high in bushes or trees. During times of low water, non-breeding sparrows used areas with higher effective vegetation heights and higher coverage of *S. rhizomatum*. During high water events, non-breeding sparrows used areas with lower coverage of *S. rhizomatum* and higher sawgrass coverage. Dean and Morrison (2001) also note that considerable variation in small-scale topography existed within CSSS habitat in subpopulation B, with substantial variation in elevation (e.g., 20–30 cm) within an area <1 km<sup>2</sup>. Walters et al. (2000) cautioned against

making interpretations about suitable water levels for the CSSS on a grid scale of 0.5–1 km. Although general landscape patterns can be measured at scales from 0.5–1 km, variation of factors such as topography and vegetation at smaller scales should be considered.

Cade and Dong (2008) evaluated the changes in quantiles of breeding CSSS counts from 1992–2005 as a function of water depth using a quantile regression count model and report flooding as an adverse factor on CSSS populations. They found that the proportion of sites with counts of at least one sparrow declined quickly when water depths increased from 0 to 15 cm; at least 25% of the sites contained a sparrow when water depths increased to 20 cm, and at least 5% of the sites contained a sparrow when water depths increased to 30 cm. Additionally, when water depths increased from 0 to 10 cm CSSS counts decreased between 33%–75%, and when water depth increased from 0 to 20 cm CSSS counts decreased between 50%–83% (Cade and Dong 2008). However, they also observed a large

proportion of sites with little flooding that contained no sparrows, indicating that a simple interpretation of drier as better for sparrows is not sufficient and that factors such as the timing of flooding and the duration of drying are important.

Rising water levels can also increase predation rates, which affect CSSS survival. Predation has been attributed as the primary cause of loss for eggs and young and has been associated with increasing water levels (Pimm et al. 2002, Baiser et al. 2008). Sparrow nest failure has been primarily attributed to predation of eggs or young or to flooding (Lockwood et al. 1997). The predation rate for nests monitored in eastern ENP from 1996–1997 increased later in the breeding season with higher water levels (Lockwood et al. 1997).

Photo by Lori Oberhofer, National Park Service



## Fire

Fire can both help and harm sparrows and their habitat, directly and indirectly. The effects of fire depend on characteristics such as the type, timing, frequency, size, patchiness, whether the fire consumes an entire patch of suitable habitat, juxtaposition of nearby suitable unburned habitat, and post-fire conditions such as rapid flooding. Although several studies (described below)

have reported impacts of fires on sparrows, information about the type or severity of fire is often unknown and may not be recorded in the ENP database (Smith et al. 2015), presenting challenges for drawing firm conclusions.

### **Fire Types**

There are three main sources of fire that occur in CSSS habitat: lightning strikes, non-prescribed human-induced fires (including arson), and prescribed fire (Curnutt et al. 1998), which often differ in the time of year they occur and how they burn. Lightning-strike fires usually occur from March–September (peak in July), non-prescribed human-induced fires usually

occur in the dry season from December–May and are often ignited in the northeast corner of ENP (peak in April), and prescribed fires usually occur from November–March (Curnutt et al. 1998). Dry season fires tend to burn more completely, and wet season fires tend to burn in a pattern that is more patchy (Curnutt et al. 1998).

Photo by National Park Service



## Timing of Fire

The time of year a fire occurs can affect the degree of its impact (Sah et al. 2009). If fires burn sparrow habitat during the breeding season (i.e., dry season fires), they can cause nest loss (Lockwood et al. 2003, Davis et al. 2005). Successful nesting requires 45–60 days of nearly dry conditions without fire (Davis et al. 2005). It is reported that fire prohibits breeding at a site the same year the fire occurs and additionally one breeding season afterwards if the area is burned in the late dry season or early wet season (the late winter or spring; Werner and Woolfenden 1983). Fire can cause direct mortality to birds when birds are present at the site of a fire, which can lead to short-term negative effects on sparrow population size (Werner 1975, Werner and Woolfenden 1983, Curnutt et al. 1998). During the

sparrow breeding season, fires can decrease fecundity and reduce local population size in ensuing years (La Puma et al. 2007). Although the occurrence of dry season fires was rare in their data set (fire records in ENP from 1982–1996), Curnutt et al. (1998) found that CSSS numbers were lower at sites with the most recent fire during the dry season. Fires that do not overlap with the breeding season have been recommended for sparrows (e.g., August–November; Kushlan et al. 1982, Werner and Woolfenden 1983). Late wet season and early dry season fires (late summer and early winter) are reported to have fewer negative effects on sparrows and their habitat (Werner and Woolfenden 1983).

## Fire Frequency

Fire is identified as a factor that affects breeding habitat for the CSSS (Post and Greenlaw 1994). The optimal time since fire for the CSSS is unclear, although it is reported that fire suppresses detrimental woody encroachment (Werner and Woolfenden 1983). Intermittent fires constrain encroachment of hardwood and dense grass and the accrual of dead plant material, all of which are undesirable for CSSS nesting (Pimm et al. 2002). Too infrequent fires can allow for the density of dead vegetation and ground litter to increase to a level unsuitable for the CSSS (Taylor 1983).

Previous reports recommend burning approximately every five years on dense *Muhlenbergia* grass sites and approximately every 8–10 years at less densely vegetated *Muhlenbergia* grass sites (Kushlan et al. 1982). Too frequent fires may be detrimental to suitable habitat (Curnutt et al. 1998) because vegetation cannot grow to CSSS nesting height (e.g., 14–18 cm; Werner 1975, Lockwood et al. 1997). Sah et al. (2010) showed that vegetation recovered to two-thirds the biomass that was present the year before a fire within three years since a fire (Mustang Fire of 2008) on the eastern side of Shark River Slough (subpopulation F). Other re-

ports indicate that vegetation can recover in two years for prairies with deep soils and in four years for prairies with shallow soils (recovery defined as reaching pre-fire vegetation biomass) and that CSSS occupancy after fire followed the response of vegetation recovery (Taylor 1983).

Hydrology can interact with both vegetation recovery after fire and also with fire frequency and the ability for areas to burn. For example, rapid flooding after fire may inhibit the ability of vegetation cover and composition to recover and can lead to altered vegetation recovery trajectories (Sah et al. 2009). A site that experienced rapid flooding post-fire showed only a 2.1% vegetation recovery per year for three years after the fire, and vegetation species assemblages differed from the pre-fire community, leading to unsuitable habitat conditions for the CSSS (Sah et al. 2009). Sah et al. (2009) also report that the type of vegetation pre-fire (e.g., marsh vs. wet prairie vegetation) can play a role in vegetation recovery after fire. Fire frequency also interacts with hydroperiod and is related to vegetation-inferred hydroperiod (hydroperiod calculated based on relationships between vegetation commu-

nities, topography, and water levels, *described above in Hydrology and Habitat section*), where higher fire frequencies are observed in drier habitats and lower fire frequencies are observed in wetter habitats (based on ENP fire data from 1980–2005; Ross et al. 2006).

Jenkins et al. (2003a) used satellite imagery to determine the spectral signatures of areas with known nests to examine variation in potential habitat for the CSSS from 1992–1999 and show that fire is a potential factor affecting habitat suitability. Subpopulation C had a relatively constant amount of potential habitat from 1992–1999 but very few birds, and between 1990–1995 almost the entire area of C burned at least once per year or more. Birds were at an undetectable level in 1993 and were not detected again until 1996. Subpopulation F also showed a small but constant amount of habitat from 1992–1999 but also burned annually from 1981 to 1994, as well as in 1996 and 1998. Sparrows

for a fire may be too short to support CSSS breeding (Lockwood et al. 2003). Several authors agree that annual or biannual fire return frequencies over large areas of the sparrow’s eastern habitats are directly linked to reduced hydroperiods and are most likely the cause of declines and failure to recover in subpopulations C and F (Curnutt et al. 1998, Nott et al. 1998).

The high frequency of human-induced fires in northeastern ENP is associated with inhibited sparrow population growth (subpopulations C and F), and dry conditions in this area are attributed to water control structures restricting water flow to the east (Curnutt et al. 1998, Nott et al. 1998, Lockwood et al. 2003). Walters et al. (2000) report that sparrows in northeast ENP (e.g., subpopulation F) are susceptible to dry season arson fires (Curnutt et al. 1998, Walters et al. 2000). Similarly, Pimm et al. (2002) report that declines in subpopulations C and F were caused by fire. Further,

Lockwood et al. (2003) note that too frequent fires caused the local extinction of sparrow populations (pre-1981) along the eastern boundary of ENP. The risk of large fires can be reduced by increasing water delivery through northeastern Shark River Slough (Walters et al. 2000). Lastly, frequent large fires are reported to have a negative effect on sparrow presence, especially in the early breeding season, and are attributed to the historic decline of sparrows in the Big Cypress area (Stimson 1968, Werner and Woolfenden 1983).

There are numerous studies that report varied patterns on the relationship between fire and the CSSS. Werner and Woolfenden (1983) report that the highest sparrow densities were found three years after a spring fire in a densely vegetated *Muhlenbergia* prairie site in upper Taylor Slough and that CSSS population density declined as vegetation became more dense and the ratio of living to dead plant matter decreased. They report that fire can improve habitat but also can restrict nesting opportunities and destroy birds at the time of the fire, for example no sparrows were observed at the site the same year of the fire, but the following

**Burned prairies are typically unsuitable for sparrows for at least two years.**

did not occupy most of the habitat in subpopulation F during the study period from 1992–1999. The results of Jenkins et al. (2003a) show a pattern between potential availability of habitat, fire, and CSSS occupancy (relationships between fire and CSSS occupancy explained in more detail below).

Fire frequency is related to sparrow occupancy and abundance patterns, where fire can eliminate sparrows from an area. Sparrows do not occupy areas that burn frequently (Curnutt et al. 1998) and frequent human-induced fires are considered a major threat (Kushlan and Bass 1983). It is reported that burned prairies are unsuitable for sparrows for at least two years (Pimm et al. 2002). It is also possible that fires burn short hydroperiod areas too often for vegetation to grow tall enough for CSSS nesting; vegetation that meets the required minimum fuel load (plant biomass)

year sparrows were observed (Werner and Woolfenden 1983). Similarly, Werner (1975) reports that CSSS densities were sparse during the first year post-burn but increased up to three years post-burn (end of study was at three years post-burn). There was also direct observation of individually marked sparrows fleeing the flaming front of a winter wildfire into adjacent unburned areas and flying in circles in areas of smoke and flames (Werner 1975). La Puma (2010) report that sparrow occupancy was lower at sites that had burned one or two years prior compared to sites that were burned three to six years ago, and after an initial decline in occupancy after fire, occupancy increased with time since fire up to 10 years since fire, followed by a decline (quadratic relationship between occupancy and time since fire). Pimm et al. (2002) report sparrow densities in the area of a 1994 fire, in plots categorized as burned, control (unburned), and edge. In this study, burned sites showed approximately one-third the number of sparrows as the unburned and edge sites for two years, after which the densities of burned, unburned, and edge sites were indistinguishable (Pimm et al. 2002). Virzi et al. (2016) report that birds did not re-colonize a burned area in subpopulation A even seven years after a fire occurred and the vegetation had recovered.

Using fire records in ENP from 1982–1996 and sparrow occupancy records from the range-wide surveys during the same time period, Curnutt et al. (1998) found a positive relationship between the mean number of sparrows and the time since fire for up to ten years, for areas east of Shark River Slough. Sparrow numbers were greatest when one or two fires had occurred over the previous 10 years, very low when five or six fires had occurred in the last 10 years, and absent when seven or more fires had occurred within the last 10 years (Curnutt et al. 1998).

Taylor (1983) reports that on burned sites with deeper soils (>40 cm), vegetation recovery was more rapid and sparrow populations recovered and peaked earlier than on sites with shallower soils. The former populations re-colonized rapidly, beginning the second year



Photo by Ecostudies Institute

after fire, and sparrow numbers increased through the fourth year after fire. In burned sites with shallow soils (<20 cm), plant biomass recovery was slower, sparrows did not re-colonize until about four years post-fire, and densities were low for up to 10 years. In addition, post-fire breeding territories in this study were clumped, presumably because birds were forced to use marginal areas following large fires. Fires created long edges where birds concentrated during the first breeding season post-fire and created a mosaic of unburned patches in which birds nested (Taylor 1983). The ability for birds to use unburned refugia is important for vulnerable species such as the sparrow (Slocum et al. 2007). Fire regimes shorter than 8–10 years could be detrimental to populations (Taylor 1983).

Benscoter et al. (2019) evaluated the effect of fire on CSSS relative bird count on a landscape-scale, incorporating data from all subpopulations. They show that the CSSS relative bird count was highest at a fire interval of five to eight years and increased linearly with the percent of cell burned (400 m resolution, 16 ha cells). More specifically, the mean predicted bird count was 0.12 birds at one to two years since fire, 0.16 birds at three to four years since fire, 0.19 birds at five to eight years since fire (mean was 0.19 birds at both five to six years since fire and seven to eight years since fire), and 0.17 birds at nine to ten years since fire. The increase in bird count from one to two years since fire to five to eight years since fire represented a 56.4% increase in the mean expected bird count (Benscoter et al. 2019).

However, evidence also exists that sparrows may not be dependent on fire or that they are capable of inhabiting sites that have not been burned in a long time. Walters et al. (2000) state that the effects of fire during the 1990s were less pronounced than the effects of water levels on the CSSS, although they emphasized that the potential for fires to impact the CSSS is large. Curnutt et al. (1998) report that many birds were observed in subpopulation B (the most productive population) during the mid- to late-1990s in areas that

had not been burned since the 1989 Ingraham Fire. Similarly, the CSSS has been reported on sites that burned 10 years prior (Taylor 1983). Furthermore, La Puma et al. (2007) report that sparrow densities and nest success in the southern portion of subpopulation E were not enhanced by fire (the Lopez Fire, a human-ignited fire), where sparrow density declined for two years after fire but then returned to levels similar to adjacent unburned areas three years after fire (La Puma et al. 2007). The Lopez Fire in the La Puma et al. (2007) study was an early dry season fire not typical of an average fire season, but it provided an opportunity to study the effect of fire in adjacent plots that were burned, unburned, and at the edge of a burn using the before-after-control-impact design. The control plot had not burned in six years when the Lopez Fire occurred nor burned in 11 years by the time the study concluded. Sparrows were absent from burned plots for the rest of the breeding season after the fire occurred and for two breeding seasons thereafter. In the third breeding season after fire, sparrows returned at densities indistinguishable from the adjacent non-burned areas and nest survival probabilities for the birds that recolonized were indistinguishable from pre-fire levels. Sparrow habitat that was burned remained unsuitable for sparrow breeding for two years after the fire mainly because of loss of vegetation structure, but after two years they recovered to similar species composition and vegetation structure as nearby unburned areas. The unburned habitat directly adjacent to burned areas supported sparrow densities, territories, and nest success indistinguishable from unburned habitat. While the sparrows tolerated fire in this study, sparrow density was not enhanced by fire in this particular study (La Puma et al. 2007). 

# Ecology

## of the Cape Sable Seaside Sparrow



### Habitat Preferences



Prefers species rich sites dominated by **muhly grass**, often found in intermediate zones between permanently flooded sloughs and drier pine dominated habitat



Typically found in areas with relatively short hydroperiods

### Rising Waters



45 days of nearly dry conditions without fire are needed for successful nesting, and 90 days for a second clutch to fledge



Nests are built close to the ground in clumps of grass, which is why rising waters end the breeding season



Rising waters can also increase predation rates (from raccoons, rice rats, and snakes)

When hydroperiods are shorter, the area may contain woody species and may be more prone to frequent or severe fires.



When hydroperiods are longer, the area may convert to taller grasses and sedges, such as sawgrass, and lack the required structure for nesting.



These habitat changes can occur over short time periods, such as 3 to 4 years.



Burned areas are typically avoided for at least two years. The effects of the fire depend on:

type, timing, size, patchiness, severity, and frequency of fire

whether the fire consumes an entire patch of suitable habitat

post-fire conditions such as rapid flooding



Photo by Stephanie Romanach/USGS



## SURVEY DATA

### Overview

Bird count surveys were initiated in 1974 in an effort to describe the geographic range and abundance of the CSSS in the southern Everglades (Kushlan and Bass 1983, Pimm et al. 2002). The first range-wide point count survey was performed in 1981 (Kushlan and Bass 1983). Range-wide surveys were reinitiated beginning in 1992 and were performed every year thereafter through 2021; however, no range-wide surveys were conducted in 2020 because of the COVID-19 pandemic.

Range-wide surveys are conducted via point counts by helicopter visits to sites located on a 1 km grid that was set up to encompass all potential sparrow habitat (Kushlan and Bass 1983). The point count surveys are conducted during the peak of the breeding season, from mid-March through May or June (Kushlan and Bass 1983, Walters et al. 2000). For each survey,

observers wait three to five minutes after helicopter set-down and record the number of sparrows detected over a seven minute interval (Curnutt et al. 1998, Pimm et al. 2002); original survey time per site was recorded as 10 minutes (Bass and Kushlan 1982). The effective count for each survey location corresponds to the number of singing males within an estimated 200 m radius (Kushlan and Bass 1983). Survey flights start at sunrise, continue for three to four hours (approximately 6:30–9:30 am; Kushlan and Bass 1983, Curnutt et al. 1998, Pimm et al. 2002), and are not conducted in inclement weather such as rain or high winds (Pimm et al. 2002). Using the helicopter method permits access to areas that are otherwise difficult to access (Cassey et al. 2007). The full range-wide survey takes approximately two months to complete (Curnutt et al. 1998).

### Population Estimation

Kushlan and Bass (1983) developed a method to estimate CSSS population size using bird point count data from the range-wide surveys by multiplying the estimated mean density of birds by the total occupied area and doubling that number to include uncounted females. This method results in a calculation that is equivalent to multiplying the number of recorded singing males by 15.87, which was rounded to 16 (i.e., multiply by eight to account for sparrows in the area between census locations and multiply by two to

account for females; Kushlan and Bass 1983, Walters et al. 2000). The 16x multiplier is still applied today in population estimates (USFWS 2020). Kushlan and Bass (1983) based their calculation on the following assumptions: 1) the ability to hear sparrows within 200 m of each survey location, which is approximately 12.6 ha around the survey point, and 2) the density of singing males in the 12.6 ha area represents the density of singing males in the corresponding 1 km<sup>2</sup> block. Kushlan and Bass (1983) considered their population



Photo by Ecostudies Institute

estimate conservative, because they deemed that every singing male was not detected at each survey location.

To assess the validity of the 16x multiplier for population estimation, Curnutt et al. (1998) compared CSSS bird count numbers recorded during helicopter surveys with CSSS numbers observed on the ground from in-depth field studies in the same locations; this was

done using five in-depth study plots where CSSS territories were mapped over four separate years. The ratio of the mean CSSS density per plot divided by the mean number of birds in the point count surveys that represented the same area, corrected for the fact that the in-depth field sites were known to contain sparrows, resulted in an estimated correction factor of 18 instead of 16 (Curnutt et al. 1998).

## Data Challenges

There are a series of assumptions in the CSSS range-wide surveys and population estimation method. Indeed, at the inception of the range-wide surveys, it was recommended to implement an intensive breeding bird census program (Kushlan et al. 1982). Post and Greenlaw (2000) identify assumptions in the range-wide survey methods and 16x multiplier method to obtain a population estimate, including: 1) all of the males in an area are recorded, 2) males are distributed uniformly, 3) sparrow activity areas during breeding do not overlap, 4) each male has one corresponding mate, 5) all areas occupied by sparrows are surveyed (i.e., sparrows do not occupy habitat outside of the surveyed areas).

general trends, the spatial distribution of the breeding population, and potential patterns of change in breeding numbers.

Fletcher et al. (2000) compared the methodologies of rope-dragging transects, another common bird survey method, to fixed-radius point counts (point counts are used in the range-wide surveys) for surveying breeding bird abundance in CSSS habitat (from November 1997 to January 1998). The fixed-radius point counts took less effort, covered a larger area in a given time period, and controlled for time spent surveying, but they can have errors in distance estimates for singing birds and may have biased detection rates based on habitat

or behavior. The rope-dragging transects required more effort per area sampled but had observations from two observers (double observer) by design (because two people are needed to drag the rope). They also can detect greater abundance numbers, especially for species found close to the

ground such as seaside sparrows, but they risk the possibility of causing birds to move away as observers advance the transect. The results of Fletcher et al. (2000) provide support that no single method of bird surveying is capable of sufficiently detecting all birds.

There are numerous proposed recommendations to improve the range-wide surveys. Post and Greenlaw (2000) provide a series of suggested improvements to the sampling technique for the range-wide bird count surveys, including: random arrangement of point counts among years within each region, replicated bird counts within a year, distance sampling, and the calibration of detectability by observer and habitat. Walters et al. (2000) have a series of recommendations to improve the range-wide survey methods, including: estimating the proportion of males singing compared to the total males present at a given time as a cor-

**Sampling limitations due to remoteness of sparrow habitat, spatial coverage, and sampling method pose challenges for assessing population stability and detecting population increases.**

The methods used for the CSSS survey data do not allow for the estimation of detection probability and the adjustment for imperfect detection (Walters et al. 2000, Cade and Dong 2008), which results in population estimates that do not have associated variability estimates (Post and Greenlaw 1994, Walters et al. 2000). Although dramatic declines in sparrows are still likely detected using current methods (Cade and Dong 2007), the limitations of the sampling method make it difficult to assess population stability and to detect population increases. These trends are typically less pronounced than a sharp decline and can occur over a long time frame (Cassey et al. 2007), especially for changes that occur at small spatial or temporal scales. Walters et al. (2000) recognize that although the CSSS range-wide surveys only represent the number of territorial males during the breeding season that are detected, this information can still aid in understanding

rection to the population estimate, determining the proportion of males that are mated, conducting the survey over a larger area, estimating detection probability, and determining the actual area sampled in each survey (*detailed suggestions to implement improvements are described more in the next two paragraphs*; Walters et al. 2000). Suggestions for future surveys also include modifying the sample design to a stratified random sampling approach if abundance estimates are the primary goal (Walters et al. 2000). Cassey et al. (2007) recommend incorporating direct estimates of detection probability for each survey year and subpopulation.

Walters et al. (2000) provide detailed suggestions to estimate the proportion of area sampled and the detection probability associated with each survey. The application of fixed-radius points counts can be used to estimate the actual proportion of area sampled at each survey location. Estimation of detection probability can be achieved by comparing point counts with known numbers in the same area identified by territory mapping. The comparison of point counts with identified numbers of individuals in the in-depth study areas was already implemented in Curnutt et al. (1998). Distance sampling using variable circular plots or double-observer sampling methods can be implemented to estimate detection probability (Walters et al. 2000). Virzi et al. (2016) point out that variation in territory sizes and CSSS density among subpopulations can affect encounter rates and detection probabilities and therefore can affect the accuracy of population estimates. Virzi et al. (2016) state that using the standard multiplier based on large subpopulations with smaller territories and greater sparrow densities would overestimate population sizes for small populations. Territory sizes are generally larger for unmated males in small subpopulations with low densities of sparrows (Virzi et al. 2016). Additionally, highly skewed sex ratios with a greater proportion of males may also compound errors from the population estimation method currently used (Virzi et al. 2016). Walters et al. (2000) recommend the annual calculation and incorporation of proportion of



area sampled and detection probability, at minimum calculated over a range of both bird densities and environmental conditions. Using these methods, counts could be adjusted by proportion of area sampled and detection probability by dividing the number of counted birds by the product of these two factors. This method would permit population estimates to have an associated variance, comprised of the variance in actual point counts (including the fraction of area sampled) and the variance associated with detection probability (Walters et al. 2000).

Walters et al. (2000) also provide suggestions to account for the fact that not all males with territories may be singing during a single survey, not all males in the population may be breeding in a given year, and the proportion of the number of males singing to the number of females may vary by site or year. Limited historical data from one study area indicated that near Taylor Slough, 12% of males were unmated through the breeding season in 1974 and 11% of males were unmated in 1975 (Werner and Woolfenden 1983). Walters et al. (2000) recommend color marking of individuals in the in-depth study areas, flush-netting, radio-tracking, and counting the number of nesting females (at nests) to aid in determining the relationship between the number of males singing during a given survey and the total bird count at a site. Walters et al. (2000) also recommend the implementation of vocalization and playback surveys in the in-depth study areas to determine all males with territories.

In addition to the difficulty in modeling occupancy or abundance patterns over time because of limited capabilities to account for detection probability, the data are highly zero-inflated (many surveyed sites contain zero birds). Approximately 82% of observations are zeros (Cade and Dong 2005, Benschoter et al. 2019), which can make detecting patterns and relationships in the data challenging.

Some efforts have been implemented to address concerns about the CSSS range-wide surveys. In the

original range-wide surveys, a non-random sampling pattern was employed where sites were sampled from east to west through the breeding season, therefore confounding the effect of time and space. Thus, a random stratified sampling method was implemented in 1995, where eastern and western subpopulations were sampled alternately from survey to survey (Curnutt et al. 1998). Other survey improvements were employed in 1981 and 1992–1996 at a subsample of sites, such as having two observers record observations at a single site and comparing their observations; no difference between observers was detected using this approach (Curnutt et al. 1998). Other calibration methods were developed, including comparing survey results at adjacent survey sites that were taken at the same time and day but by different observers; no significant differences in observations between observers at adjacent sites were detected (Curnutt et al. 1998). From 2016–2020, survey methods were added to evaluate efficacy of improved methods to estimate sparrow density and abundance, including adding more point count survey sites, visiting point count survey sites more than once per year, and recording distance data and time-of-detection data; details regarding the results of these added survey methods are not yet published (Virzi and Tafoya 2021). It is also reported that periodically, sites not known to support sparrows have been checked for the presence of singing birds (Walters et al. 2000).

In an effort to account for detection probability, Curnutt et al. (1998) compared estimates from the survey point counts to the number of birds recorded in the in-depth census plots in the same areas and report that the point counts underestimate the number of sparrows by 36% (Curnutt et al. 1998). Additionally, some survey sites were surveyed twice in 1999 and 2000 (Pimm et al. 2002), which permitted the estimation of a mean detection probability of 0.58 for the CSSS for a subset of sites. Detection probability was not influenced by standing water depth but varied with month and declined with time of day, higher wind speeds, and higher CSSS density (Pimm et al. 2002). Virzi and Tafoya (2021) estimated detection probability via

point count surveys at approximately 0.50, indicating that replicated surveys in a given year are needed for improved abundance and density estimates. However, detection probabilities for each sample site for each survey do not exist for this data set (Cade and Dong 2008), which make the incorporation of detection probability limited.

Using statistical modeling on 15 years of data, La Puma (2010) determined that the range-wide survey methods are insufficient to detect large declines in occupancy or abundance over short time periods (three

years) but improve when longer time periods are analyzed (e.g., 5–10 years). Walters et al. (2000) conclude that because the accuracy of population estimation is unknown, it is difficult to know whether apparent extinction in some subpopulations is real or if it reflects sampling error. They recommended, at minimum, the estimation of detection probability across the range of environmental conditions and bird densities for the CSSS throughout its range (Walters et al. 2000).

Photo by Everglades National Park

## Risk

Pimm and Bass (2002) conducted a risk analysis to examine the causes of CSSS population fluctuations and to aid in understanding declines in population and spatial range and the ability to recover from them. The risk analysis combined a simple demographic model (using best estimates or inferred estimates of parameters based on empirical data) with calculations of the variability of suitable breeding habitat. The model evaluated relative rates of population increases for three demographic scenarios based on availability of suitable habitat. Their results suggest that the CSSS can persist with at least three viable subpopulations, and they recommend reaching target nesting conditions (i.e., no flooding during the breeding season) west of Shark River Slough. 



# Collecting Survey Data

on the Cape Sable Seaside Sparrow

Photo by Yvette Cano/NPS



## Data sources:

### Range-wide surveys

📍 Survey 1 km grid across CSSS range during peak breeding season

### Use point counts to record observed singing males over 7 minute interval



Multiply by 16 (x8 to account for area between census locations and x2 to account for females)

assumptions of the 16 multiplier  
All occupied areas are surveyed and all males are recorded  
Males have one mate and are distributed uniformly

## Demographic surveys

complement our understanding of:

- Nest timing importance
- Factors in nest survival
- Impact of multiple broods

Impact of habitat variability to population viability

## Challenges

Accessing remote locations and surveying the amount of area needed during the appropriate time period

Detecting population increases or declines over short time frames and small scales

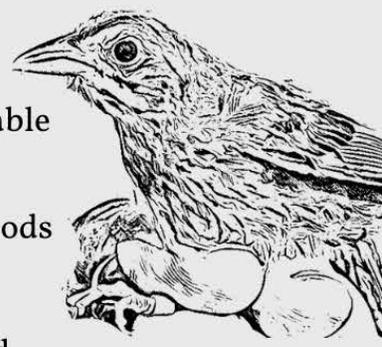
Knowing whether apparent extinction is real or reflection of sampling method

\* No surveying method can sufficiently detect all birds

## Best practices

- ✓ visit sites more than once a year
- ✓ extend surveys to suitable but unoccupied areas
- ✓ use double-observer methods to calibrate detectability
- ✓ determine proportion of males that are mated

✓ use male-bias measurements in population estimates





## POTENTIAL SUITABLE CONDITIONS

### Land Cover

Two vegetation data sets cover the CSSS subpopulations and ENP. The Florida Fish and Wildlife Conservation Commission and Florida Natural Areas Inventory developed the Cooperative Land Cover data set (2019) that contains land cover types for the state of Florida (Fig. 3). Although the marl prairie vegetation type is specified, it covers only a small portion of the

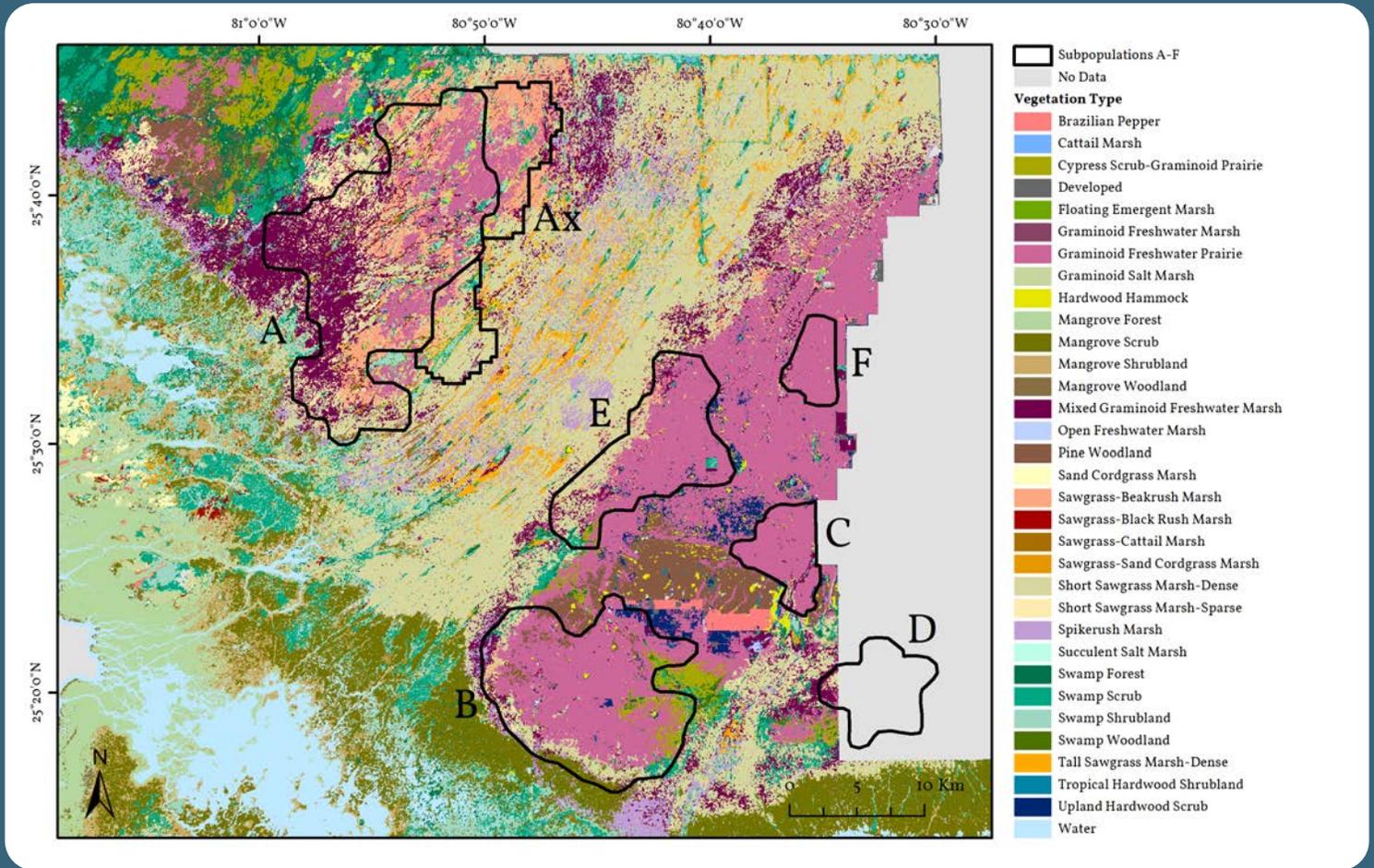
CSSS subpopulations while “sawgrass” and the broader vegetation type “glades marsh” cover the majority. The National Park Service has also developed a high-resolution vegetation map for ENP and BCNP using aerial imagery from 2009 (Ruiz et al. 2019, Ruiz et al. 2021) that covers the majority of the subpopulations except for D where only a small portion is covered (Fig. 4).

Photo by Ecostudies Institute



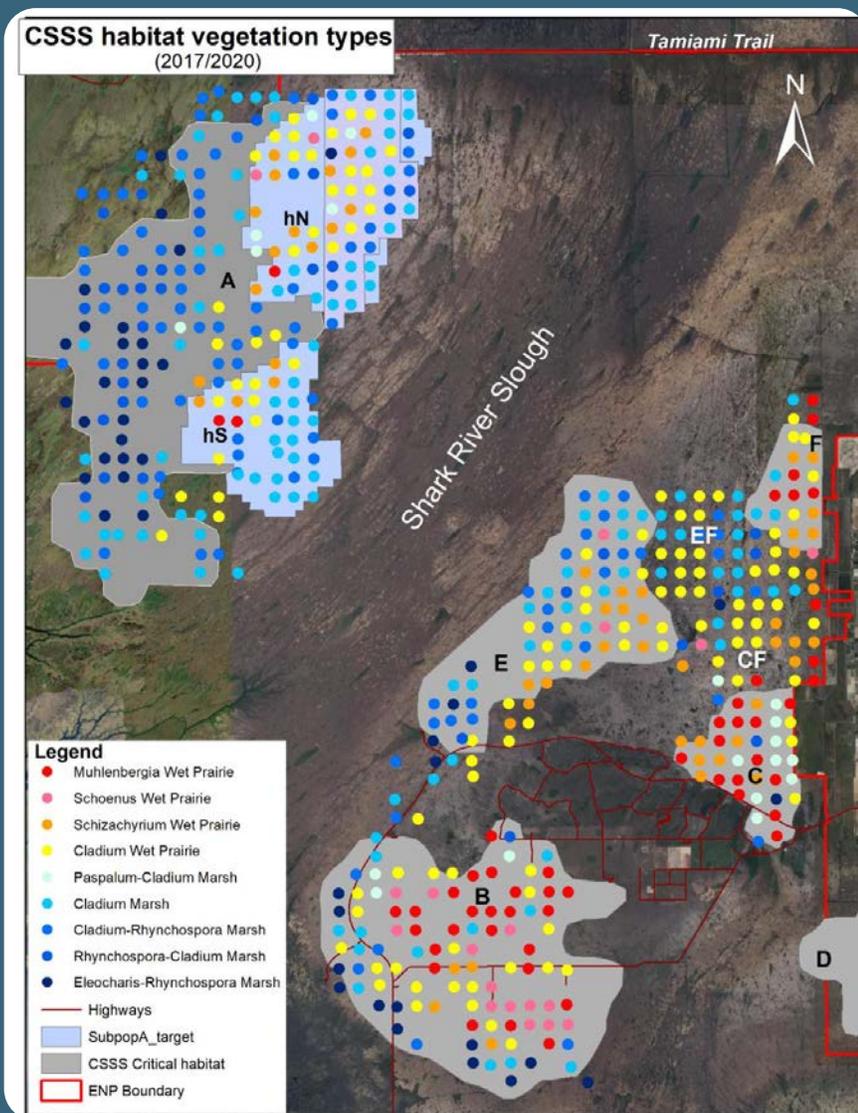


### National Park Service Vegetation Map



**Figure 4.** The National Park Service high-resolution vegetation map (Ruiz et al. 2019, Ruiz et al. 2021) overlaid with the boundaries of the Cape Sable Seaside Sparrow (*Ammospiza maritima mirabilis*) subpopulations (delineations for subpopulations derived from USFWS [2020]). Vegetation types that are dominant within the sparrow subpopulations are graminoid freshwater prairie, mixed graminoid freshwater marsh, and short sawgrass marsh-dense. Graminoid freshwater prairie is defined as a “short hydroperiod marsh characterized by a mix of graminoids that includes low-stature sawgrass (*Cladium jamaicense*), muhly grass (*Muhlenbergia capillaris* var. *filipes*), little bluestem (*Schizachyrium rhizomatum*), and black sedge (*Schoenus nigricans*), among others” (Ruiz et al. 2021). Mixed graminoid freshwater marsh is defined as a “co-dominant mix of two or more graminoid marsh species like spikerush (*Eleocharis* spp.), panicgrass (*Panicum* spp.), sawgrass, and/or beakrush (*Rhynchospora* spp.). This community is characterized by having no graminoid species in dominance, meaning there is 15% or less difference in percent cover between two or more graminoid species” (Ruiz et al. 2021). Short sawgrass marsh-dense is defined as a “short statured (<1 m tall) sawgrass marsh with 15-49% sawgrass cover. Spikerush, beakrush, and/or panicgrass, among others, maybe be present. However, their combined cover does not exceed that of sawgrass. This class differs from mixed graminoid freshwater marsh in that sawgrass is dominant” (Ruiz et al. 2021).

Although the NPS vegetation map (Fig. 4) displays a large area of the CSSS subpopulations as containing graminoid freshwater marsh, this vegetation classification is a broader definition that includes but is not exclusive to suitable CSSS habitat. Vegetation monitoring within the CSSS subpopulations produces a higher specificity of vegetation type (Sah et al. 2021; Fig. 5), and CSSS presence varies based on these more specific types (Lockwood et al. 2003, Ross et al. 2006).



**Figure 5.** Vegetation types within and around the boundaries of the Cape Sable Seaside Sparrow (*Ammospiza maritima mirabilis*; CSSS) subpopulations (A, B, C, E, and F) at survey sites in 2017 and 2020. Community types are shown in a spectrum from dry (red) to wet (blue). Two areas identified as potential future suitable habitat for the CSSS (subpopulation A target [SubpopA\_target]) are labeled hN and hS. Abbreviations: ENP = Everglades National Park. The figure is from Sah et al. (2021).

## Decision Making Tools

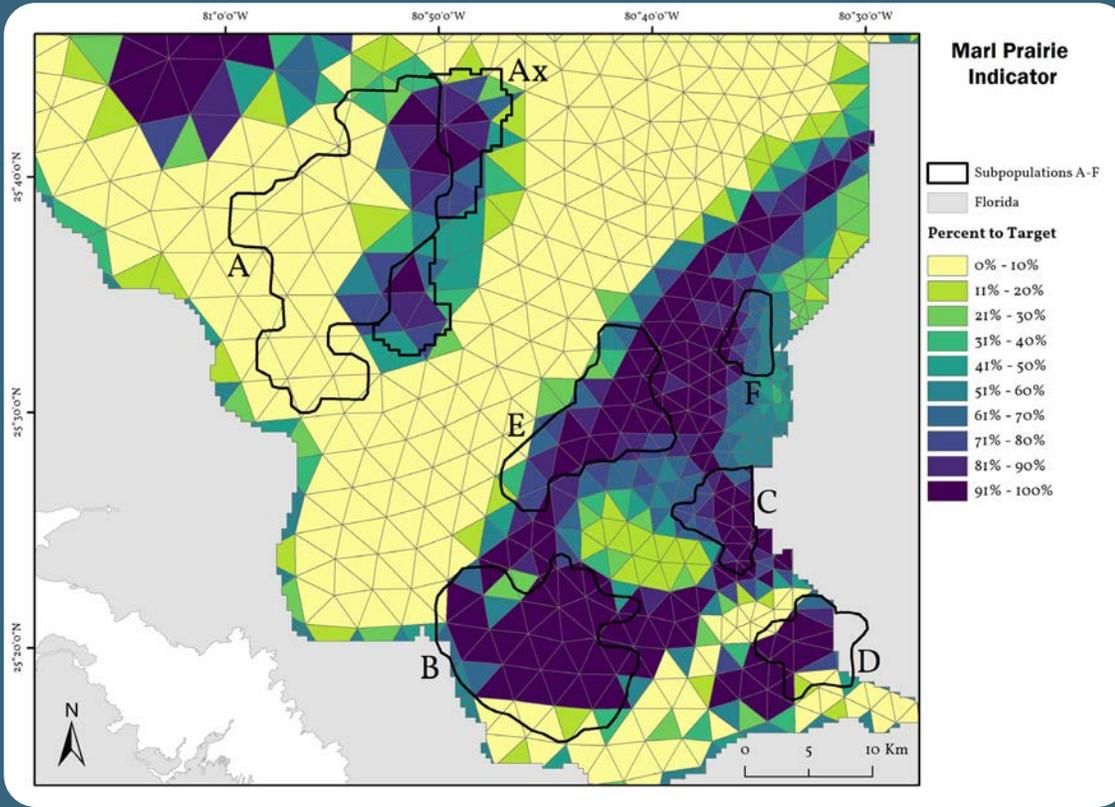
Predictive models have been developed as decision support tools for the CSSS since the 1990s (e.g., Nott and DeAngelis 1999). Four decision support tools have been developed in the last decade for restoration planning relative to the sparrow: 1) CSSS Marl Prairie Indicator, 2) EverSparrow, 3) Sparrow Helper, and 4) CSSS Viewer. Each of these tools is accessible on the Joint Ecosystem Modeling (JEM) website at [www.jem.gov](http://www.jem.gov). These four tools are frequently used to compare alternative water management scenarios for restoration planning as well as for the multi-agency ecosystem-based management recommendations. Two additional landscape-level tools can support decision-making through hydrologic forecasting and vulnerability assessment. The Everglades Vulnerability Analysis (EVA) provides a measurement of relative vulnerability to changing conditions, including vegetation type within the CSSS subpopulations. The JEM tool EverForecast provides a framework to examine tradeoffs and predicted outcomes for other species when water management focuses on the CSSS.

The CSSS Marl Prairie Indicator is a temporally and spatially explicit model that uses hydrologic frequency return periods to simulate marl prairie habitat suitability (Pearlstine et al. 2016). Based on historical locations of high CSSS presence, it calculates target return intervals over a period of record (optimally a minimum of 40 years, but it can use shorter time spans). The Marl Prairie Indicator provides spatial output showing percent-to-target scores for each Regional Simulation Model (RSM) polygon that indicate marl prairie habitat suitability for CSSS over the entire period of record. The CSSS Marl Prairie Indicator can be accessed on the JEM website at [jem.gov/Modeling/MarlPrairie](http://jem.gov/Modeling/MarlPrairie).

EverSparrow is a spatially explicit Bayesian model and the only existing model that quantifies the interdependent relationships between a range of environmental factors and CSSS presence (Haider et al. *in press*). Using hydrologic conditions such as mean 4-year hydroperiod and maximum depth, fire occurrence history, and vegetation structure throughout the range of the CSSS, EverSparrow provides weekly probabilities of CSSS presence at a 400 m resolution. The hydrologic response curves produced by EverSparrow illustrate changing CSSS habitat suitability with increasing maximum water depth and hydroperiod and can be used by managers to estimate ideal hydrologic conditions. EverSparrow has been integrated into the JEM EverForecast application and can be accessed at [jem.gov/Modeling/EverSparrow](http://jem.gov/Modeling/EverSparrow).

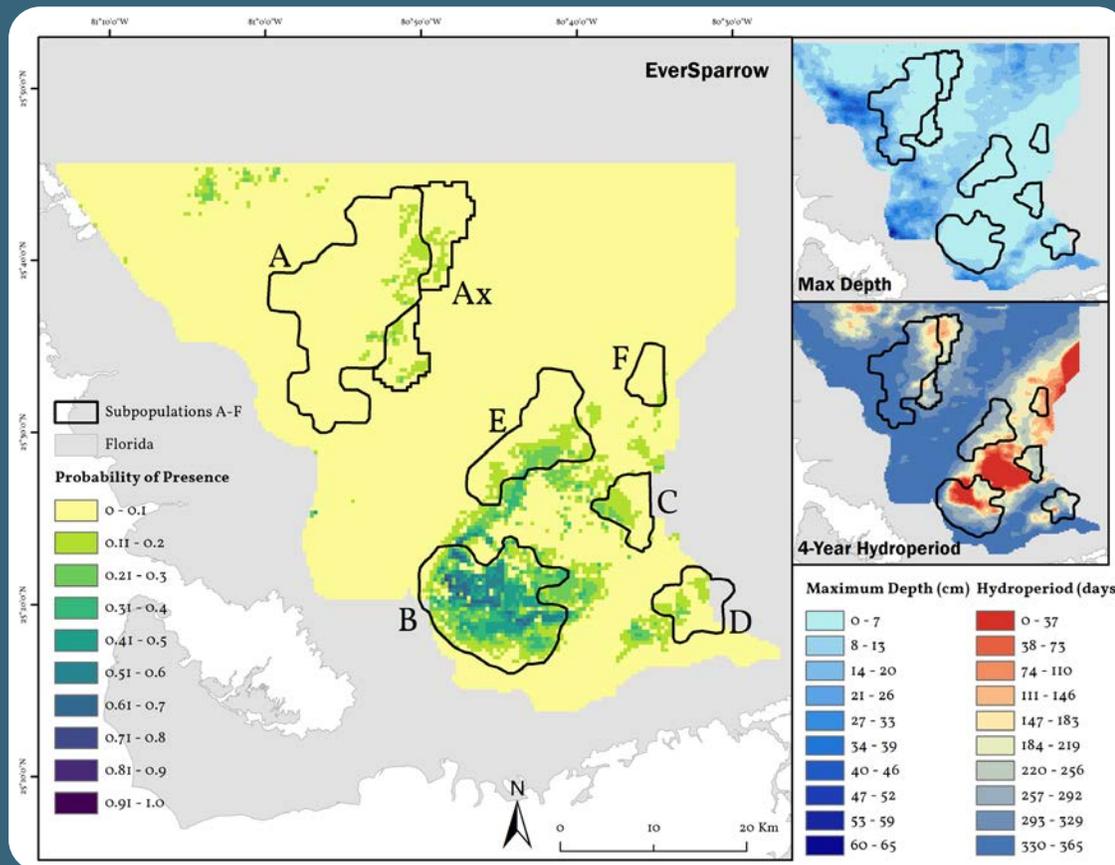
The CSSS Marl Prairie Indicator and EverSparrow models can also provide estimates of potential suitable CSSS habitat. We ran the CSSS Marl Prairie Indicator (Pearlstine et al. 2016) using the full Everglades Depth Estimation Network (EDEN) data set from 1991 to present (July 8, 2021) to produce percent-to-target scores for each RSM polygon (Fig. 6). We ran EverSparrow for 2020 and took the mean value across the primary breeding season months, March–June, to show average probability of presence for the year (Fig. 7). We also show two of the input hydrometrics for EverSparrow, maximum depth and 4-year hydroperiod.

### CSSS Marl Prairie Indicator Output



**Figure 6.** Model output for the Marl Prairie Indicator using water depths from the Everglades Depth Estimation Network from 1991 through present (July 2021) and overlaid with the boundaries of the Cape Sable Seaside Sparrow (*Ammospiza maritima mirabilis*) subpopulations (subpopulation delineations derived from USFWS [2020]).

### EverSparrow Output



**Figure 7.** Model output from the EverSparrow model averaged over the primary breeding season months, March–June, for 2020. Also shown are two hydrologic model inputs, maximum water depth and 4-year hydroperiod, along with the boundaries of the Cape Sable Seaside Sparrow (*Ammospiza maritima mirabilis*) subpopulations (subpopulation delineations derived from USFWS [2020]).

There are additional tools available to aid in decision making. The Sparrow Helper was developed using CSSS observations to determine the hydrologic variables that are highly correlated with CSSS presence (Beerens et al. 2016). Per subpopulation, mean water depth and mean 4-year hydroperiod were found as the most discriminating of the daily and annual metrics, respectively. Sparrow Helper outputs include the proportion of area within each subpopulation with the target 4-year discontinuous hydroperiod of 90–210 days. The tool provides annual tables and figures for the above hydrometrics as well as 20 others for each subpopulation. Sparrow Helper can be accessed on the JEM website at [jem.gov/Modeling/SparrowHelper](http://jem.gov/Modeling/SparrowHelper).

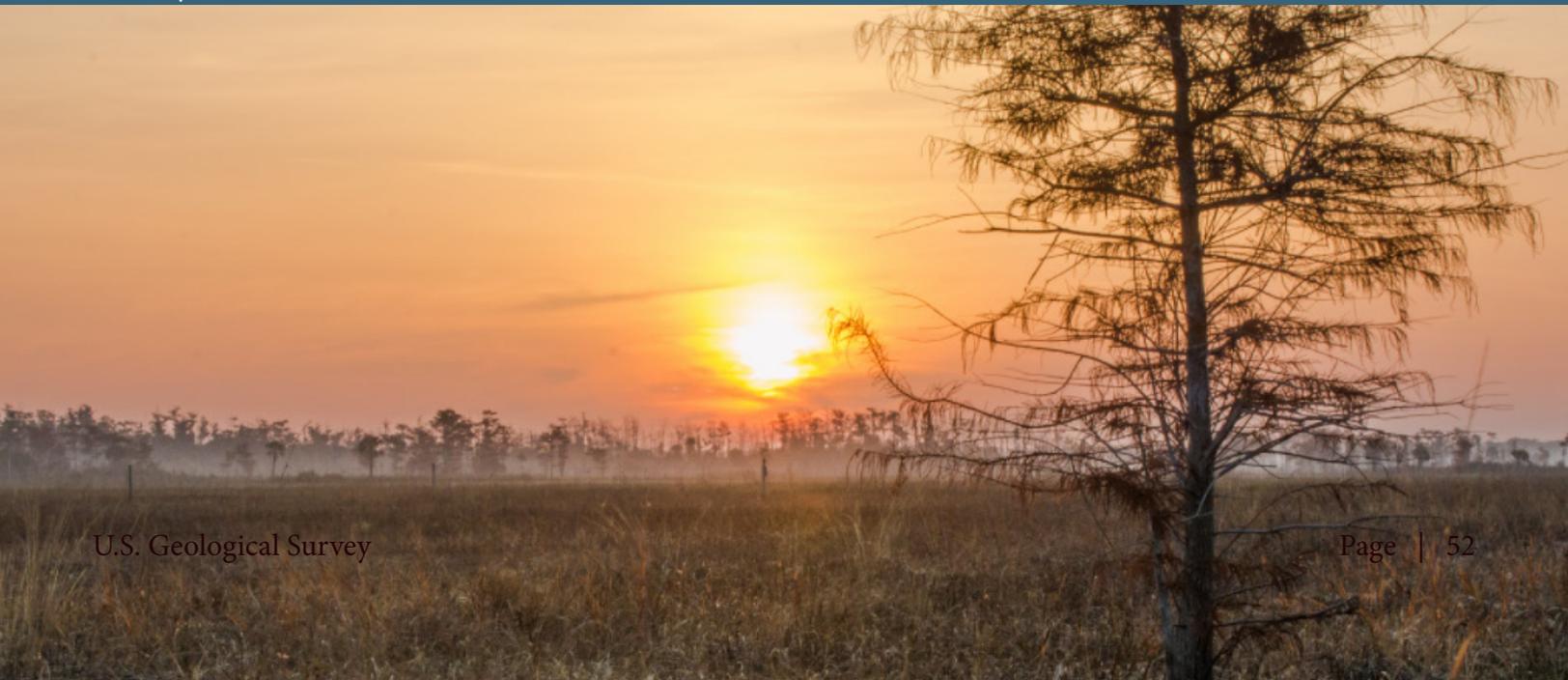
The CSSS Viewer uses the U.S. Geological Survey (USGS) EDEN daily water depth surfaces to estimate real-time water depths in sparrow habitat and annual summary statistics for each subpopulation (USGS 2014). Water depths, dry days, and hydroperiod are critical metrics, and the CSSS Viewer creates real-time spatial outputs for these metrics. The CSSS Viewer application provides daily interactive maps and animations at a 400 m resolution as well as downloadable tables and figures of annual hydrometrics for each subpopulation. The CSSS Viewer was developed by the USGS EDEN team and is accessible at [sofia.usgs.gov/eden/csss](http://sofia.usgs.gov/eden/csss).

The EVA uses Bayesian networks to estimate relative

vulnerability of ecosystem health on an annual scale across the Everglades landscape (D'Acunto et al. 2021). Two of the EVA submodules relate to marl prairie habitat suitability: peat dynamics (subsidence and accretion) and vegetation pattern. The EVA can be integrated with restoration scenarios and climate forecasts of sea level rise to measure vulnerability as compared to a user-defined ideal state. The tool provides a way to measure potential vulnerability of suitable CSSS habitat to changing future conditions. The EVA provides annual outputs at a 400 m resolution. Information about EVA can be found on the JEM website at [jem.gov/Modeling/eva](http://jem.gov/Modeling/eva).

Additionally, EverForecast is a spatially explicit, hydrologic, and ecological operational forecast tool that can help decision makers balance the needs of multiple species across the Everglades landscape (Pearlstine et al. 2020, Haider et al. 2021). EverForecast provides: 1) a range of near-term hydrologic conditions, 2) predicted species responses to forecasted hydrologic conditions, and 3) tradeoffs among species when hydrologic conditions are targeted to meet the needs of a particular species. The CSSS is one of the species considered in the model, in addition to others such as the Wood Stork (*Mycteria americana*), endangered Everglades Snail Kite (*Rostrhamus sociabilis plumbeus*), and American Alligator (*Alligator mississippiensis*). EverForecast is updated monthly and is accessible on the JEM website at [jem.gov/everforecast](http://jem.gov/everforecast). 

Photo by Bente Torvund



# Decision Support Tools

Use these tools to enhance your understanding of the Cape Sable Seaside Sparrow (CSSS) and for:

-  comparing alternative water management scenarios for restoration planning
-  multi-agency ecosystem-based management recommendations

Provides scores of marl prairie habitat suitability over decadal time scales

Marl Prairie Indicator



Calculates hydrologic metrics highly correlated with CSSS presence

Sparrow Helper



CSSS Viewer



Estimates real-time water depths in sparrow habitat and annual summary statistics for subpopulations

EverSparrow



Quantifies the interdependent relationships between a range of environmental factors and calculates probability of CSSS presence

Access these tools by visiting the Joint Ecosystem Modeling (JEM) website at

[jem.gov](http://jem.gov)





## CLIMATE CHANGE

The effects of climate change are of concern for isolated species with limited dispersal abilities, such as the CSSS (SFNRC 2009). The CSSS is already affected by habitat fragmentation (Pimm et al. 2002, Davis et al. 2005), which may compound the effects of climate change because the CSSS has limited habitat availability and dispersal capabilities. The CSSS is also susceptible to changing water levels, and therefore it is likely that the CSSS will be affected by an increase in storm frequency and intensity, increased drought, and sea level rise predicted for Florida (SFNRC 2009). The primary habitat of the CSSS, the marl prairie, is expected to be affected by sea level rise (SLR; SFNRC 2009); SLR can affect both the spatial extent (e.g., via inundation) and species composition (e.g., via degree of inundation and salinity) of habitats where the CSSS currently resides. The effects of SLR may lead to higher water levels in CSSS habitats, which can result in breeding habitat degradation (Elder and Nott 2008). Using modeled scenarios of one, two, and three feet above mean higher high water (MHHW) marks, the United States Army Corps of Engineers showed that CSSS habitat is vulnerable to sea level rise, particularly within subpopulations A, B, and D. Even under the baseline model scenario, there is evidence that the negative effects of sea level rise on CSSS habitat may already be underway in subpopulations A, B, and D. Scenarios of only one or two feet increases in MHHW may cause significant habitat loss and degradation to the eastern side of subpopulation A and to most or all of subpopulations B and D. Under higher sea level rise conditions over the longer term, all subpopulations may be affected (USFWS 2020). Furthermore, Sah et al. (2020) show that red mangrove cover and mean frequency increased from 2008–2018 along vegetation transects in the southwestern portion of subpopulation B (and areas southwest of subpopulation B), representing the transition from freshwater marsh to brackish water vegetation compo-

sition, as determined via detailed vegetation surveys along transects in ENP. Similarly, the southern portion of subpopulations B and D are reported to have already converted to mangrove (Tylan Dean, personal communication, July 21, 2021).

Given that the duration of the CSSS breeding season coincides with the length of the dry season in the Everglades (Lockwood et al. 1997, Elder and Nott 2008) and that sparrows nest close to the ground, changes in hydrology could affect CSSS reproductive success (Lockwood et al. 2001, Pimm and Bass 2002, Baiser et al. 2008, Boulton et al. 2011; *see Hydrology and Breeding Impacts section for more details*). Catano et al. (2015) report that the CSSS will likely be more affected by changes in precipitation compared to temperature, given the timing of the nesting season with specific water conditions. In 2020, Infanti et al. report that south Florida is likely to experience an increase in precipitation during the dry season (November–January) and a decrease during the wet season (June–August). How these changes impact sparrow reproduction will depend on the degree of the increases and decreases and the spatial and temporal dynamics of these rainfall patterns, as well how altered rainfall interacts with other factors such as SLR and water management. 

# Climate Change

## and the Cape Sable Seaside Sparrow



### Existing Challenges



Habitat Fragmentation



Water Management

### Increased Pressure

Precipitation more likely to affect habitat compared to temperature, given the timing of nesting with water conditions

### Predicted Changes



Increased storm frequency and intensity



Increased temperature



Change in fire frequency and intensity



Sea level rise

### Impact to Species



Subpopulations that experience more flooding and less suitable habitat support fewer birds relative to those that have less flooding and provide suitable habitat

### Impact to Breeding Habitat

Habitat suitability may decline due to:



Saltwater intrusion, freshwater flows, and their interaction



Mangrove encroachment



Peat soil collapse

## CAUTION

The effects of climate change are of

### particular concern

for isolated species with limited dispersal abilities, such as the Cape Sable Seaside Sparrow.



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