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By
Brian Reichert, Christopher Cattau, Wiley Kitchens, Robert Fletcher,
Jean Olbert, Kyle Pias, and Christa Zweig

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U. S. Geological Survey
Florida Cooperative Fish and Wildlife Research Unit
University of Florida
PO Box 110485
Gainesville, Florida 32611

PREFACE

This document is intended as an addendum to the 2008 report (entitled: *Snail Kite Demography Annual Report 2008*, Cattau et al. 2008). This document should not be considered a definitive stand-alone and completed piece of work. We also strongly recommend the 1997 report (entitled: *Movement and Demography of the Snail Kite in Florida*, Bennetts & Kitchens 1997a) for more complete explanations about the background and protocols of this study.

This document is an annual progress report regarding the status of our Snail Kite monitoring study and is intended to inform and update our funding agencies and other interested parties. This monitoring effort is long term. It should be noted that our field personnel are monitoring the Snail Kite throughout its range on a year-round basis. This progress report allows investigators to highlight significant progress and findings made both analytically and anecdotally. All information in this document is protected by the UF intellectual property policy. Data provided in this document may not be used for publication in any manner without the express prior written consent of the authors.

ABSTRACT

This report concentrates on demographic data collected during 2011, but also incorporates data collected since 1992. Recent demographic results reveal alarming trends concerning the Snail Kite population in Florida. Snail Kite abundance has drastically declined since 1999, with the population essentially halving from 2000 to 2002 and again from 2006 to 2008. Each of these two periods of population decline coincided, in part, with a severe regional drought throughout the southern portion of the kites' range. The 2001 drought significantly, yet temporarily, affected adult survival, especially for kites within the Everglades region, and the nesting patterns and lack of recruitment that have been observed since that time give us special concern about the recovery of the Snail Kite population. A life table response experiment (LTRE) has shown that 80% of the reduction in the stochastic population growth rate is attributable to adult fertility (i.e., the product of (1) young fledged per adult and (2) juvenile survival). Preliminary results from a population viability analysis (PVA) conducted in 2010 predict a 95% probability of population extinction within 40 years. These results are especially concerning, as the risk of extinction has increased as compared to results from a previous PVA conducted in 2006 (Martin, 2007). As juvenile recruitment has been lacking since 1999, recent analyses provide indications of an aging population with problems inherent to older individuals, including increased adult mortality rates and decreased probabilities of attempting to breed which have been shown to be exacerbated during times of harsh environmental conditions.

Multiple factors may be limiting the reproductive ability of the kites and reducing the carrying capacity of several of the wetland units throughout the state, and the reasons for this severe decline in population viability are probably tied to both short-term natural disturbances (e.g., drought) and long-term habitat degradations (e.g., the conversion of wet prairies to sloughs in WCA3A). There has been a notable decline in Snail Kite production from two critical Snail Kite habitats, Lake Okeechobee and WCA3A. Okeechobee, which from 1985 to 1995 was a productive breeding site, has been only a minor contributing unit (in terms of reproduction) since 1996. In 2010 and 2011, nesting was observed on Okeechobee for the first time since 2006, which may represent an increase in habitat quality.

Reproduction within WCA3A has been suppressed in recent years, as no young were fledged there in 2005, 2007, 2008 or 2010; however, in 2011, there were 11 successful nests in WCA3A. Nonetheless, the low level of nest productivity in WCA3A observed this year suggests

that habitat conditions may remain poor.

The decline of Lake Okeechobee and WCA3A as productive kite habitats has left the population heavily concentrated in and dependent upon the Kissimmee River Valley (KRV), particularly Lake Tohopekaliga (Toho), which accounted for 41% of all successful nests and 57% of all fledged young that were documented on a range-wide basis from 2005-2010. Toho remained productive in 2011, accounting for 33% and 36% of all successful nests and fledged young, respectively. During the 2011 breeding season, an unprecedented amount of breeding activity occurred on East Lake Toho (60 active nests, 39 successful nests), likely due in large part to the foraging opportunities at the adjacent Lake Runnymeade.

While the estimated population size for 2011 (i.e., 925 individuals, up from 826 in 2010) along with the increased number of fledglings counted during the 2011 breeding season are encouraging trends, it remains unclear whether such trends may potentially signify the beginning of a recovery phase. In this report we detail new findings related to Snail Kite demography, movement, and foraging. We also make specific recommendations that may help guide management decisions aimed at increasing Snail Kite population growth.

INTRODUCTION

The Snail Kite (*Rostrhamus sociabilis plumbeus*) is a wetland-dependent raptor that displays an extreme form of dietary specialization, feeding almost exclusively on a single species of freshwater apple snail (*Pomacea paludosa*) (Howell, 1932; Stieglitz & Thompson, 1967), which is the only species of this genus native to Florida (Rawlings et al., 2007). The Snail Kite inhabits flooded freshwater wetlands and shallow lakes in peninsular Florida. These wetlands exhibit considerable variation in their physiographic and vegetative characteristics, which include graminoid marshes (e.g., wet prairies, sloughs), cypress prairies, lake littoral shorelines, and even some highly disturbed areas such as agricultural ditches and retention ponds (Sykes, 1983b, 1987a; Bennetts & Kitchens, 1997a). Three features that remain constant within the selected habitats are the presence of apple snails, sparsely distributed emergent vegetation, and suitable nesting substrates – all of which are critical to the foraging and nesting success of the Snail Kite.

Snail Kite survival depends on maintaining hydrologic conditions that support these specific vegetative communities and subsequent apple snail availability in at least a subset of critically-sized wetlands across the region each year (Bennetts et al., 2002; Martin et al., 2006). The historical range of the Snail Kite once covered over 4000 km^2 (2480 mi^2) in Florida, including the panhandle region (Davis & Ogden, 1994; Sykes et al., 1995), but since the mid-1900s it has been restricted mainly to the watersheds of the Everglades, Lake Okeechobee, Loxahatchee Slough, the Kissimmee River Valley (KRV), and the Upper St. Johns River of the central and southern peninsula (Fig. 1). After several decades of landscape fragmentation and hydroscape alteration, the kite population is now confined to a fragmented network of freshwater wetlands that remain within its historical range, and the viability of the population rests entirely on the conditions and dynamics of these wetland fragments (Bennetts & Kitchens, 1997; Martin, 2007). The Snail Kite is unique in that it is the only avian species that occurs throughout the central and south Florida ecosystem and whose population in the U.S. is restricted to freshwater wetlands in this region. The dependence of the Snail Kite on these habitats makes it an excellent barometer of the success of the restoration efforts currently underway (Kitchens et al., 2002) (e.g. USFWS *Multi-Species Transition Strategy for Water Conservation Area 3A*, 2010).

Wetland habitats throughout central and southern Florida are constantly fluctuating in response to climatic or managerial influences, resulting in a mosaic of hydrologic regimes and vegetative communities. Snail Kites respond to these fluctuations demographically and through movements within the network of wetlands in central and southern Florida (Bennetts & Kitchens, 1997; Kitchens et al., 2002; Martin et al., 2006, 2007a, 2007b). In order to optimize conservation strategies for the complex system inhabited by the Snail Kite in Florida, it is essential to have a thorough understanding of the kite's ability to move among wetlands, their resistance and resilience to disturbance events (e.g., droughts), and the demographic effects that specific management actions and other habitat changes have on the kite population.

This report presents demographic data on the Snail Kite in Florida. It concentrates on data collected in 2011, but also synthesizes data collected since 1992.

METHODS

Study Area

The Florida population of Snail Kites is best viewed as a single spatially-structured population, distributed among a network of heterogeneous wetland units in central and southern Florida (Bennetts & Kitchens, 1997a, 1997b; Martin et al., 2007a, 2007b). Kites utilize the entire spatial extent of their range, exhibiting interchange among wetland units (Bennetts & Kitchens, 1997a, 1997b; Martin et al., 2006). The study area encompasses the "core" wetland units used for nesting by Snail Kites throughout peninsular Florida, and includes all wetlands in which breeding activity has been observed (Fig. 1).

Monitoring Protocol

Survey method

Multiple consecutive surveys were conducted throughout the designated wetland units (Fig. 1) from March 1st to June 30th at 2-3 week intervals of each year since 1992. This time period coincides with the occurrence of peak nesting (Bennetts & Kitchens, 1997a). The surveys followed a format similar to the quasi-systematic transects conducted by airboat for the annual count (Sykes, 1979, 1982; Bennetts et al., 1994).

During each survey we inspected every sighted kite using both binoculars and spotting scopes. We categorized each observed individual as follows: "marked" if the kite carried a band that could be uniquely identified; "unmarked" if the sighted kite did not carry an identifiable band; or "unknown" whenever the banding status of the kite could not be determined (Martin et al., 2007a).

Nest monitoring

We searched for Snail Kite nests anytime we were in the field and kite(s) displayed behavior indicative of reproductive activity (e.g., flushing from a nest, calling to a mate, defending territory). Nest searches were not limited to population surveys or even to the population survey period, as we conducted searches (1) during preliminary "pre-season" (November-February) surveys of wetlands known to support breeding kites, (2) during population surveys (March-June) of all wetlands in Figure 1, and (3) during other year-round field excursions (e.g., conducting time energy budgets, banding and/or radio-tagging individuals, radio tracking individuals). If a kite displayed breeding behavior but no nest was located, we marked the GPS location as a potential nest site and returned at a later date (approximately 1-2 weeks later) to search again.

Nests were checked with a telescoping mirror pole to determine their status. Water depths at certain nests were determined by placing a meter stick vertically into the water column until it rested on the sediment. GPS locations of the nests, nesting substrates and heights of the nests, as well as, nest statuses and fates were also recorded. Nests were considered successful if they fledged at least one young. We categorized the nests that failed using the following four categories:

1. Predation or post scavenging event with no nest collapse: any nest that included scattered remains of young or adults kites (e.g., wing; conglomerate of feathers etc.), broken egg shells, or no eggs where a full clutch was present before.
2. Predation or post scavenging event associated with nest collapse: any nest built on robust substrate (e.g. shrubs), whose collapse was likely to have been caused by a land predator (e.g., raccoons). This category only included nests with a reasonable access to land predator (i.e., water depth < 50 cm and/or relatively close to land < 50m)
3. Nest collapse: any nest failure associated to the falling of the nest out of its original location.
4. Unable to determine reason of nest failure: any nest that contained an incomplete egg clutch on subsequent monitoring visits (possibly due to abandonment or adult mortality) or any nest that that could not be relocated.

Banding

The average fledging age of kites is 28.7 days (Sykes, 1987b) but may range from 24 to 35 days (Sykes et al., 1995; Synder, 1989). At approximately 24 days of age, Snail Kite nestlings were banded with unique alpha-numeric bands.

Data Reported and Statistical Analysis

Nest success and productivity

We defined nest productivity as the number of young fledged per successful nest and calculated average nest productivity on a wetland specific basis. We calculated nest success using the following estimator: $\hat{S} = x/n$, where $\hat{S}$ is the maximum likelihood estimate of the probability of nest survival (i.e., apparent nest success), x is the number of nests that produced at least one fledgling, and n is the number of nests initially observed with at least one egg (i.e., active nests) (Williams et al. 2002). We calculated the standard error (SE) of nest success as follows: $SE = \sqrt{\hat{S} * (1 - \hat{S})/n}$ and we used the normal approximation to compute 95% confidence intervals (95% CI) (Williams et al. 2002). Nests that were not initiated (i.e., never reached the egg laying stage) and nests found after young had already reached the minimum fledging age (i.e., 24 days old) were not included in our analyses of nest success or productivity.

Range-wide annual survival

Extensions of the Cormack-Jolly-Seber model (CJS) (Cormack, 1964; Jolly, 1965), implemented in program MARK 6.1 (White & Burnham, 1999) were used to estimate survival probability and detection probability (i.e., the probability of detecting a Snail Kite given that it is present in the study area during the period of sampling). Hypothesized models were ranked using the Akaike Information Criterion (AIC) (Burnham & Anderson, 1998). The protocol has been published elsewhere (Bennetts & Kitchens, 1997a; Bennetts et al., 2002; Martin et al., 2006, 2007a) and further explanation can be found in Appendix A of Cattau et al. (2009).

A set of biologically relevant models was developed that allowed survival (Phi) and detection (p) probabilities to vary by time (t), differ by age class (Juveniles: 30 days – 1 year; Adults: >1 year), and to be dependent upon a suite of biologically relevant variables, including hydrological covariates and habitat indices.

Historically, range-wide survival estimates have been highly influenced by the hydrological conditions present in WCA3A (Martin, 2007; Martin et al., 2007c). Previous analyses identified WCA3A, in terms of its influence on demography, as the most critical component of Snail Kite habitat in Florida (Mooij et al., 2002; Martin, 2007, Martin et al., 2007c). Therefore, we included hydrological covariates representative of conditions in WCA3A (model 9, Table 2). This model constrained adult survival by years that were categorized by a clustering analysis using the 3-gauge average in WCA3A (*see* Cattau et al. 2008).

As the observed decrease in population growth rate in years after 1998 has been attributed in part to the long-term effects of habitat degradation in WCA3A (Martin et al., 2008), we included models that assumed different probabilities of survival for the periods before (pre-degradation) and after (post-degradation) 1999. Also included were models which tested the hypothesis that adult survival declined steadily beginning in 1999 in response to habitat degradation. Models which tested this hypothesis forced adult survival to decrease linearly (models 3 and 4, Table 2) or on the cumulative logit scale (model 10, Table 2).

We used the information theoretic approach of model averaging across the most parsimonious models to produce model averaged estimates of range-wide survival and detection (Anderson 2008).

Goodness of fit

To assess goodness of fit we utilized the median c-hat approach (Cooch and White 2008) using the most general model (*Phi (Adult(t) Juvenile(t)), p (Adult(t) Juvenile(t))*, where "Phi" represents apparent survival, "t " for time variation (in years) and "p" represents detection probability). The approach yielded a *c-hat* > 1. Therefore, parameter variance components were adjusted to account for over-dispersion and we used quasi-AIC (QAICc) for model selection instead of AICc.

Population size

We used the super-population estimator (described in detail by Dreitz et al. (2002) and Martin (2007a)) to derive the population size of Snail Kites between 1996 and 2011. The estimator incorporates intra-breeding season mark-recapture data to estimate the probabilities of detection and survival for both marked and unmarked individuals. As an open-population type model, it allows for the migration of individuals in and out of the study area. Therefore, the 'super-population', is an estimate of abundance for the all of the Snail Kites that used the study area during the duration of the breeding season.

RESULTS

Reproduction

Number of nests counted

In 2011, a total of 293 nests were located range-wide during the survey season. Of this total, 235 nests were observed in an active state (i.e., containing eggs or nestlings) (Table 1). Sixty-nine percent of the active nests occurred in the KRV, with the majority occurring on Toho, which alone accounted for 33% of the range-wide nesting effort. An unprecedented number of active nests were located in East Toho this year (Table 1), which accounted for 26% of the range-wide nesting effort. In 2011, nesting was documented for the first time in Lake Runnymeade, which is where many of the kites nesting in East Toho were observed foraging. Also of note, WCA3A and Okeechobee accounted for 10% and 17% of the range-wide nesting effort in 2011, respectively (Table 1). As has largely been the case since 2005, the majority of the nesting effort continues to be concentrated in the KRV (Fig. 3a).

Number of juveniles banded and number of young fledged

We were able to confirm the nest fate for 215 of the 235 active nests. From these active nests of known fate, 201 young were confirmed to have fledged (Table 1). We banded 241 nestlings during the pre-fledging stage. However, some nests were depredated after nestling(s) were banded but before fledging; therefore, not all of the nestlings that we banded actually fledged. Also note that we were not able to band all of the young that were confirmed to have fledged; therefore, the total number of young fledged includes banded and non-banded individuals that were known to reach fledging age.

The total number of young fledged throughout the entire state dropped substantially after 1998; however, it was higher in 2011 than in any year since 1998 (Fig. 2). In 2011, Toho and East Toho each produced 36% of the young that were fledged statewide, while Okeechobee contributed 13% and other wetlands made minor contributions (Table 1). As has largely been the case since 2005, the KRV is still responsible for the majority of all kite production (Fig. 3c).

Nest success and productivity

As noted in the methods, nests that were found after young had reached the minimum fledging age (i.e., 24 days old) were not included in our analyses of nest success or productivity.

Thus, 11 nests (one each from Okeechobee, Runnymeade and Kissimmee, and four each from East Toho and WCA3A) containing 22 fledglings (one from Kissimmee, two from Runnymeade, three from Okeechobee, four from WCA3A, and 12 from East Toho) were excluded when estimating nest success and nest productivity. These censored nests and their respective fledged young are, however, included in the respective tallies in Table 1.

In 2011, range-wide nest success averaged 0.50 (SE = 0.04). Thirty-three percent of all successful nests in 2011 occurred in Toho, where nest success was 0.51 (SE = 0.06). And 35% of all successful nests occurred in East Toho, where nest success was 0.67 (SE = 0.07). Nest success in WCA3A in 2011 was 0.50 (SE = 0.13), which represents the highest nest success achieved in this wetland since 2003 (Fig. 4a). All wetland-specific nest success estimates for 2011 appear in Table 1. Estimates of annual range-wide nest success from 1995 to 2011 are presented in Figure 4b.

In 2011, range-wide nest productivity (i.e., mean number of young fledged per successful nest) was 1.77 (SE = 0.08). Nest productivity was much higher in Toho and East Toho than in WCA3A, which had the lowest nest productivity of any wetland in 2011 (Table 1, Fig. 5). Nest productivity in WCA3A has declined significantly since the late 1990s (Fig. 6).

Survival

Models that allowed juvenile survival to vary by time (i.e., separate parameters were assigned to juvenile survival for each year) were well supported by the data (w = 1; Table 2). Hence, all of the top models allowed juvenile survival to vary through time. This is likely due to the spatiotemporal variation in both kite production and environmental conditions. Adult survival does not show such drastic variation from year to year, and thus was best modeled in terms of quantifiable hydrological and environmental conditions. Widespread drought conditions occurred during 1992, the second half of 2000, and in 2001, 2007, and 2009. Because the annual minimum daily mean stage level in 2009 was similar to that of 2001 and 2007, we also included models that assumed drought conditions for 2009. Long-term habitat degradation as defined by Martin et al. (2008) affected post-1998 years (as denoted by *preDeg, postDeg, and Deg Trend*; Table 2).

The most parsimonious model (Juv(t), Adult(DE(92=01, 00, 07=09)) p(Juv(t) Adult(t)); Table 2) allowed juvenile survival to vary by time, while it constrained the variation in adult survival to that between drought and non-drought years. The three parameter drought effect on survival assumed similar survival rates for 1992 and 2001, 2007 and 2009, and 2000 separately. Detection probability was allowed to vary between adults and juveniles and to vary by year (Table 2).

The most parsimonious model (QAICc Weight = 0.408) was only slightly more supported by the data than models 2-5 (Table 2). The only difference in the second best model was that one (instead of three) unique drought effect parameter was included. Model selection provides evidence that the low water conditions which occurred in 2009 did have an effect on range-wide adult survival similar to the conditions which occurred in 2007. The relative ranking and support for models assuming drought effects provide evidence that adult survival was dependent on the hydrological conditions in WCA3A (model 3), which is consistent with results from past survival analyses (Martin et al. 2007 and Cattau et al. 2008). Although nesting has been minimal in WCA3A, adults continue to utilize this habitat, especially during the non-breeding season. Therefore, adult survival is still critically influenced by the hydrological conditions in the unit.

The five most parsimonious models were all within four delta QAICc units from each other; therefore, we considered all of them to be well supported by the data. This provided justification for utilizing the information theoretic approach of model averaging to obtain estimates of apparent survival and detection based on multiple competing hypotheses instead of using only the top ranked model from the set.

In contrast with the results from Reichert et al. (2010), models which incorporated hypotheses about long-term degradation of habitat in 3A were not well supported by the data. Six of the models, with only 0.2851 of the total QAICc weight, included hypotheses that adult survival differed between pre-degradation (1992-1998) and post-degradation (1999-2010) eras.

Adult survival declined significantly from 2000 through 2002, and again from 2006 through 2008 (Fig. 7a). These historically low survival estimates correspond temporally to significant declines in the population (Fig.8) and to region-wide droughts. Adult survival decreased by 16% from 2000 to 2002 (Martin et al. 2006), and by approximately 35% from 2006 to 2008. Juvenile survival has varied widely over time and reached a record low in 2000-2001 (Fig. 7a). Evidence shows that juvenile survival significantly decreased in the years 2004 to 2006 and rebounded in 2007 (Fig. 7a). Although Florida also experienced severe drought conditions in

2007 and 2009, there was less of an effect on juvenile survival. This disjunct is likely due to the fact that the majority of young fledged in 2007 and 2009 came from the KRV. Lake levels in the KRV have historically been less affected by adverse drought conditions (Bennetts & Kitchens 1997). Again, note that confidence limits for the adult and juvenile survival estimates for 2010 and point estimates for 2011 will not be available until the conclusion of the analysis using data from the 2012 breeding season.

Detection

Detection probability (p) has increased for both juveniles and adults (Fig. 7b) since 1992. Models in our analysis that allowed detection probabilities to differ by age group and vary through time received all of the support from the data (w = 1; Table 2). This assumption was made based off of field observations which suggested that juvenile Snail Kites tend to occupy the same wetlands and stay within close proximity of each other. Also, it has been suggested that during the breeding season (i.e., the survey period March - June) juvenile Snail Kites may occupy less optimal habitat than adults, which could result in juveniles spending less time in traditional breeding areas and lower their detection probability.

As evidence from the data suggests (observed through the QAICc model ranking) detection rates vary from year to year. Therefore, survival probabilities for the last time interval (i.e. 2009 to 2010) should be interpreted with caution as final estimates will not be available until the conclusion of the analysis using data from the 2011 breeding season.

In addition to differences in detection through time and between age groups, preliminary evidence suggests that encounter rates (a component of detection probabilities) differs between wetland sites (see Appendix A in Reichert et al. 2010). Although the estimation procedures presented here do not account for this source of variation, we are currently working to test this hypothesis as a part of the iterative process to continually improve procedures to parameter estimation.

Population Size

The Snail Kite population in Florida progressively and dramatically decreased between 1999 and 2002 from approximately 3400 to 1700 adult birds. Population size estimates of abundance between 2002 and 2006 suggest a possible period of stabilization at approximately

1500-1600 birds. However, the 2007 estimate was significantly lower than that in 2006, and the estimates from 2008, 2009 and 2010 were all significantly lower than that in 2007, suggesting that the Snail Kite population halved again between 2006 and 2008, leveling off around 700-800 individuals (Fig. 8). The 2011 population size estimate, 925 (SE ± 52), is significantly greater than the 2008 through 2010 estimates (Fig. 8); however, more data is necessary to determine whether the population is entering a biologically meaningful recovery phase.

DISCUSSION

Our recent demographic studies reveal disconcerting trends in the Snail Kite population in Florida. Kite numbers have drastically declined since 1999, with the population essentially halving from 2000 to 2002 and again from 2006 to 2008 (Fig. 8) (see also Chapter 3 Martin et al. 2007c) While the population estimate for 2011 is greater than population estimates for the previous three years, it remains unclear whether this trend might signify the beginning of a population rebound. Concurrent with the periods of population decline was a decline in the number of nesting attempts and the number of young fledged (Fig. 2), as well as a period of generally lower and more variable nest success (Fig. 4b) (see also Chapter 4, Martin et al. 2007c). A number of factors likely contributed to these observed declines, including short-term natural disturbances (e.g., drought) and long-term habitat degradations (e.g., the conversion of wet prairies to sloughs in WCA3A).

First of all, one of the major historic components of the kite's habitat network in Florida, Lake Okeechobee, remained relatively unproductive for over a decade. Okeechobee, which from 1985 to 1995 was a productive breeding site, was only a minor contributing unit (in terms of reproduction) from1996 to 2010, and this has been attributed, in part, to a shift in the water management regime (Bennetts & Kitchens, 1997). Hurricanes in 2004 further degraded kite nesting and foraging habitat in Okeechobee (*personal observation*). Since 1996 most kite nesting attempts on Okeechobee have occurred in the Clewiston Flats southwest of Moonshine Bay; however, this habitat becomes unsuitable for kite nesting and foraging at lake stages below 15 feet NGVD, which has been the case in many recent years. In 2010 kites nested on Okeechobee for the first time since 2006, but no nesting was documented in Clewiston Flats. Kite nesting in Okeechobee in 2010 occurred around Observation Island, in Eagle Bay, and along the north shore to the east and west of the Kissimmee River inflow. Kites returned to similar locations (and in greater numbers) to nest in 2011, making Okeechobee the third most productive habitat, in terms of kite reproduction, this year (Table 1). Okeechobee water levels fell drastically in the month of May, which caused at least some nest failure (Appendix B) and may have potentially suppressed juvenile survival and deterred additional nesting attempts. While the jump in production from Okeechobee in 2011 is promising, close coordination with water managers will be necessary to maximize success in the future. Furthermore, the geographic extent and temporal availability of suitable snail kite habitat on the lake is uncertain, which may affect the utility of

Okeechobee as a critical stopover point for kites that are moving among wetlands, as Okeechobee lies nearly in the center of the kite's range. The loss of such a refugia and stopover habitat may have significant demographic consequences (especially during drought), which network analyses and further spatially explicit population modeling could help elucidate. Lake Okeechobee will be critical to the kite's long-term population persistence, especially given the susceptibility of juvenile kites in the KRV to an increase frequency of local disturbance events due to cold weather and the treatment of hydrilla (the effects of which are currently being investigated). Although snail shells were only available for collection at seven nests in Lake Okeechobee, 100% of the snails collected were exotic. Although native egg clusters were observed in several locations, the recolonization of Lake Okeechobee seems to be, at least in part, dependent upon the recent presence of exotic apple snails.

WCA3A is another critical habitat that has been largely unproductive in recent years. Snail Kite reproduction in WCA3A decreased sharply after 1998 (Fig. 2) (see also Martin et al. 2008), and alarmingly, no kites were fledged in 2001, 2005, 2007, 2008 or 2010, and in 2009, only two kites (both from the same nest) were observed to have fledged from WCA3A. The lack of reproduction in WCA3A during this period likely stemmed, at least in part, from a shift in water management regimes in which (1) rapid recession rates and low water stages often shortened the window during which favorable breeding and foraging conditions were present, likely having proximal effects on nesting effort, success, and juvenile survival and (2) prolonged high water events led to long-term habitat degradation that negatively affected Snail Kite nesting and foraging habitat (Mooij et al. 2002, Martin, 2007, Zweig & Kitchens, 2008) and suppressed apple snail populations (Darby et al. 2009). While 11 successful nests were observed in WCA3A in 2011, the low level of nest productivity observed this year (Figs. 5, 6) suggests that habitat conditions may remain poor.

The relatively low reproductive output from Lake Okeechobee and WCA3A has left the kite population heavily concentrated in and dependent upon the KRV (Fig. 3), particularly Lake Toho, which accounted for 41% of all successful nests and 57% of all fledged young that were documented on a range-wide basis from 2005-2010. Toho remained productive in 2011, accounting for 33% and 36% of all successful nests and fledged young, respectively. During the 2011 breeding season, an unprecedented amount of breeding activity occurred on East Toho (60 active nests, 39 successful nests, 73 fledged young), likely due in large part to the foraging opportunities at the adjacent Lake Runnymeade. Shells collected at nests in 2011 reveal that a large proportion of snails utilized by kites in the KRV are exotic (Fig. 9). A recent study (Cattau 2008, Cattau et al. 2010) demonstrated that difficulties experienced by kites handling exotic snails may have negative repercussions to juvenile energy balances and raised potential concerns about the effects of exotic snails on kite recruitment. Field work for this study was conducted from 2003 through early-2007; however, due to the low numbers of kites using Toho in 2003-2004, the bulk of the data from Toho was collected in 2005-2006. Subsequent work has revealed that the median size of exotic snails consumed by kites in Toho decreased significantly after 2006 (Fig. 10). Recent observations indicate that kites are successfully foraging in hydrilla dominated habitats (Appendix D, in Cattau et al. 2009) which appear to be utilized by smaller size exotic apple snails (Appendix E, in Cattau et al. 2009). So this decrease may have arisen from a shift in the size frequency distribution of exotics snails that are available to foraging kites or from a shift in the foraging behavior of kites to select for smaller snails, or both. Regardless, the consistent utilization of smaller sized exotic snails in 2007-2011 relative to 2005-2006 seem to buffer the potential deleterious consequences of exotic snails on the kite population that were hypothesized in Cattau et al. (2010). Additional work related to the nesting and foraging ecology of kites on Toho is also being conducted (see Appendices B, C in Reichert et al. 2010), which should improve our understanding of the conditions necessary to facilitate successful reproduction and survival.

In addition to the potential impacts on survival and reproduction, the recent distributional shift of the primary Snail Kite nesting range has increased the heterogeneity associated with detection rates. For example, we hypothesize that detection rates have recently increased at least in part due to the increased percentage of the kite population utilizing lacustrine wetlands within the KCOL and Lake Istokpoga, leading to increased encounter rates for survey methodologies inherent to these areas. We are currently investigating possible improvements for modeling procedures of survival and super population size (see Appendix A in Reichert et al. 2010) which will address this paradigm shift.

Recent mark-recapture analyses have elucidated age-dependent variation within adult Snail Kite demographic parameters. Specifically, results provide evidence of senescent declines in both Snail Kite survival and breeding probabilities, an effect that is exacerbated during harsh environmental conditions. Based on a declining proportion of Snail Kites attempting to breed since 1996 and minimal juvenile recruitment since 1999, the Snail Kite population may be becoming comprised of predominantly older individuals (see Appendix A in Reichert et al. 2010). Therefore, individual-based patterns of declining survival and fertility with age may also increase the probability of extinction.

Given that the Snail Kite population in Florida is critically endangered, it is important to note the conservation implications of recent genetic findings from Haas et al. (2009). While *R. s. sociabilis* is genetically isolated from *R. s. plumbeus* and *R. s. major*, there may have been recent genetic exchange between the latter two geographically isolated subspecies. Furthermore, there is no evidence of genetic diversion between *R. s. plumbeus* populations in Florida and Cuba. If population declines continue to drive the Snail Kite toward extinction in Florida or if deleterious genetic effects (e.g., inbreeding depression) arise due to low population numbers, it may be feasible to consider captive breeding of and supplemental translocation from the *R. s. plumbeus* population in Cuba and or the *R. s. major* population in Central America. However, further studies of genetic and ecological exchangeability are necessary before such actions could be taken.

RECOMMENDATIONS

Since the Snail Kite population is at risk of extinction and because adult fertility plays such an overwhelming role in the population growth rate, it is critical to identify and attempt to remedy all factors that negatively affect Snail Kite reproduction and juvenile survival. In recent years, kites have been highly concentrated in the KRV during the breeding season. It is critical for the persistence of the Snail Kite population that we actively manage for kites in the KRV while restoration efforts are being made to bring other historic nesting areas (e.g., WCA3A, Okeechobee) back online. All proposed water and vegetation management actions should undergo critical evaluation processes in order to eliminate, or at least minimize, any potential negative impacts on Snail Kites. Further studies related to kite nest success and foraging efficiency on Toho are also being conducted (see Appendices B and C in Reichert et al. 2010), which will aid future management decisions.

The water regulation schedule in WCA3A does not mimic the seasonal patterns driven by the natural hydrological cycle; therefore, water management in WCA3A may be in conflict with the life history strategy of the Snail Kite. In recent years water levels in WCA3A have been maintained at unusually high levels (in part due to recent hurricanes) for the period September to January. At times, these high water stages during the pre-breeding season have been coupled with fast recession rates through the breeding season and dry conditions during and after the breeding season (i.e., when juveniles would be fledging and dispersing). Such a scenario shortens the window of opportunity for Snail Kite reproduction and may decrease nest success and juvenile survival. See Cattau et al. (2008) for a discussion on the proximate effects of water management on kite reproduction and survival. Several researchers (e.g., Mooij et al., 2002; Kitchens et al., 2002; Darby et al., 2005; Zweig & Kitchens, 2008) have raised concerns about potentially adverse effects of prolonged high water stages in WCA3A. See Appendices B and C in Cattau et al. (2008) for discussions of long term effects of prolonged high water on vegetative communities and kite habitat. We collaborated extensively on a recent set of guidelines that addresses these water management concerns (i.e., USFWS *Multi-Species Transition Strategy for Water Conservation Area 3A*, 2010) and suggest improving and refining the recommendations through monitoring and employing an adaptive-management framework. While targets were met for the wet season high and the recession rate of WCA3A in 2011, conditions later in the breeding season became much drier than those dictated by the target minimum stage window.

Thus, although many successful nests in WCA3A fledged young earlier in the breeding season (median fledge date 4/4/2011), the survival probability of those juveniles may be suppressed due to low water conditions later in the season.

Management of the kite population will also require landscape scale considerations as long distance movements are more restricted than previously understood (see Appendix A). Restoration projects that involve wholesale dry downs of an entire region (e.g., restoration of Lake Tohopekaliga) (Welch, 2004) may want to consider the option of conserving water in at least some local patches within the region to be affected, to serve as refuge for Snail Kites Draw downs of local patches should occur sequentially, allowing a sufficient recovery period for previously dried areas to return to a productive level.

Since natural resource managers currently lack a fully integrative approach to managing hydrology and vegetative communities with respect to apple snail and Snail Kite populations, we also strongly recommend the implementation/continuation of a monitoring program of Snail Kite foraging success/efficiency of individual kites, nest success and recruitment of offspring as related to foraging habitat characteristics (i.e., hydrologic conditions, vegetative community compositions, snail densities) in WCA3A. Elucidation of (1) the relationships among these habitat characteristics and (2) the effects these habitat characteristics have on Snail Kite reproduction and recruitment will fill the information gaps that currently preclude our ability to holistically manage hydrology, vegetation, snails, and kites, and by doing so, it will be possible to apply hydrologic prescriptions with predictable responses to real-time or user defined scenarios.

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Table 1. Wetland-specific nesting summary statistics for the 2011 Snail Kite breeding season.

	ENP	WCA2B	WCA3A	WCA3B	STA3	OKEE	GW	SJM	TOHO	ETOHO	KISS	IST	LH	LJ	LR	**Overall**
Number of…																
Active Nests	2	1	23	2	1	39	4	2	78	60	11	4	4	2	2	235
Known-fate Nests	2	1	18	0	1	35	4	2	73	56	11	4	4	2	2	215
Successful Nests	0	0	11	NA	0	16	0	0	37	39	4	3	0	0	2	112
Failed Nests	2	1	7	NA	1	19	4	2	36	17	7	1	4	2	0	103
Young Fledged	0	0	11	NA	0	26	0	0	73	73	8	5	0	0	5	201
Proportion of…																
Active Nests	0.01	0.00	0.10	0.01	0.00	0.17	0.02	0.01	0.33	0.26	0.05	0.02	0.02	0.01	0.01	
Successful Nests	0.00	0.00	0.10	NA	0.00	0.14	0.00	0.00	0.33	0.35	0.04	0.03	0.00	0.00	0.02	
Young Fledged	0.00	0.00	0.05	NA	0.00	0.13	0.00	0.00	0.36	0.36	0.04	0.02	0.00	0.00	0.02	
Nest success	0.00	0.00	0.50*	NA	0.00	0.44*	0.00	0.00	0.51	0.67*	0.30*	0.75	0.00	0.00	1.00*	0.50*
Nest productivity	NA	NA	1.00*	NA	NA	1.53*	NA	NA	1.97	1.74*	2.33*	1.67	NA	NA	3.00*	1.77*

* Eleven nests (one each from Okeechobee, Runnymeade and Kissimmee, and four each from East Toho and WCA3A) containing a total of 22 fledglings (one from Kissimmee, two from Runnymeade, three from Okeechobee, four from WCA3A, and 12 from East Toho) were excluded when estimating nest success and nest productivity because they were found after young had reached the minimum fledging age (i.e., the nests were already successful). These censored nests and their respective young are, however, included in the tallies of the observed number of nests and fledged young presented in this table, and they were also included when calculating relative contributions of these counts to range-wide totals (Figs. 2, 3).

Table 2. Selected single-state models (receiving >0.0 QAICc Weight) of apparent survival and detection of Snail Kites (“Juv”: juveniles (0-1 years); “Adult”: (>1 years)) in Florida from 1992-2010. Factors affecting survival (Phi) and detection (p) probabilities that were incorporated into the models included age, year, hydrological conditions and long-term habitat degradation in WC3A (“DegTrend” & “PreDEG/PostDEG”).

#	Survival (Phi)	Detection (p)	QAICc	QDelta AICc	QAICc Weight	Num. Par	QDeviance
1	Juv(t) Adult(DE(92=01, 00, 07=09)) *Adult apparent survival was negatively affected by drought). Drought effect (DE) was similar for years 1992 and 2001, 2007 and 2009, but different than 2000.*	Juv(t), Adult(t)	8895.35	0.000	0.408	61	2635.88
2	Juv(t) Adult(DE(92=00=01=07=09)) *Adult apparent survival was negatively affected by drought). Drought effect (DE) was similar for years 1992, 2000, 2001, 2007, and 2009.*	Juv(t), Adult(t)	8897.25	1.908	0.157	59	2641.89
3	Juv(t) Adult (CA3AVG) *Adult apparent survival dependent on years categorized by hydrological conditions at gauge CA3AVG in WCA3A, see Cattau et al. (2008).*	Juv(t), Adult(t)	8897.36	2.019	0.149	62	2635.84
4	Juv(t), Adult(DE(92=01, 00, 07=09) (PreDEG, PostDEG)) *Adult apparent survival was negatively affected by drought and was lower for years 1999-2009 (PostDEG) compared to years 1992-1998 (PreDEG). Drought effect (DE) was similar for years 1992 and 2001, 2007 and 2009, but different than 2000.*	Juv(t), Adult(t)	8897.37	2.026	0.148	62	2635.85
5	Juv(t) Adult(DE(92=00=01=07=09) (PreDEG, PostDEG)) *Adult apparent survival was negatively affected by drought and was lower for years 1999-2009 (PostDEG) compared to years 1992-1998 (PreDEG). Drought effect (DE) was similar for years 1992, 2000, 2001, 2007, and 2009*	Juv(t), Adult(t)	8899.31	3.960	0.056	60	2641.89
6	Juv(t) Adult(DE(92=00=01=07=09) (Linear) DegTrend *Adult apparent survival was negatively affected by drought and began declining in 1999 (linear trend). Drought effect (DE) was similar for years 1992, 2000, 2001, 2007, and 2009*	Juv(t), Adult(t)	8899.81	4.469	0.044	61	2640.35
7	Juv(t) Adult(DE(92=00=01=07,09) (PreDEG, PostDEG)) *Adult apparent survival was negatively affected by drought and was lower for years 1999-2009 (PostDEG) compared to years 1992-1998 (PreDEG). Drought effect (DE) was similar for years 1992, 2000, 2001, 2007, and different for 2009.*	Juv(t), Adult(t)	8901.23	5.881	0.022	61	2641.76

8	Juv(t) Adult(DE(92=01=07) (Linear) DegTrend *Adult apparent survival was negatively affected by drought and began declining in 1999 (linear trend). Drought effect (DE) was similar for years 1992, 2001, and 2007.*	Juv(t), Adult(t)	8901.92	6.578	0.015	61	2642.46
9	Juv(t) Adult(t) *Adult apparent survival varied through all years.*	Juv(t), Adult(t)	8908.01	12.662	0.001	76	2617.64
10	Juv(t) Adult(DE(92=00=01=07=09) DegTrend (CLogit) *Adult apparent survival was negatively affected by drought and began declining in 1999 (cumulative logit). Adult apparent survival was negatively affected by drought). Drought effect (DE) was similar for years 1992, 2000, 2001, 2007, and 2009.*	Juv(t), Adult(t)	8912.10	16.752	0.000	61	2652.63
11	Juv(t) Adult(DE(92=01, 00, 07=09)) *Adult apparent survival was negatively affected by drought). Drought effect (DE) was similar for years 1992 and 2001, 2007 and 2009, but different than 2000.*	Juv=Adult(t)	8913.33	17.981	0.000	42	2692.72

Table 3. Model averaged estimates (see *AICc Weights*, Table 2) of adult and juvenile survival (*Phi*), 1992-2010. Note that standard error terms for 2010 and point estimates for 2011 will be available after the 2012 survey season is completed.

Year	Adult survival (± SE)			Juvenile survival (± SE)		
1992	0.663	±	0.0639	0.632	±	0.090
1993	0.873	±	0.0126	0.425	±	0.058
1994	0.873	±	0.0126	0.674	±	0.087
1995	0.873	±	0.0120	0.839	±	0.068
1996	0.873	±	0.0122	0.606	±	0.073
1997	0.873	±	0.0123	0.574	±	0.047
1998	0.873	±	0.0120	0.354	±	0.044
1999	0.873	±	0.0133	0.239	±	0.098
2000	0.836	±	0.0828	0.123	±	0.050
2001	0.663	±	0.0629	0.112	±	0.073
2002	0.872	±	0.0159	0.424	±	0.080
2003	0.873	±	0.0124	0.575	±	0.083
2004	0.871	±	0.0158	0.172	±	0.059
2005	0.872	±	0.0124	0.261	±	0.103
2006	0.871	±	0.0162	0.231	±	0.064
2007	0.722	±	0.0352	0.604	±	0.079
2008	0.872	±	0.0135	0.462	±	0.066
2009	0.724	±	0.0381	0.325	±	0.070
2010	0.871			0.520		

Figure 1. Study area, with the numbers indicating the five core wetland regions sampled during annual population monitoring surveys. Kissimmee River Valley (KRV) includes Lake Istokpoga and the entire Kissimmee-Chain-of-Lakes (KCOL), most notably Lakes Tohopekaliga (Toho), East Toho, Kissimmee, Runnymeade (LR), Jackson (LJ) and Hatchineha (LH). St. John's Marsh (SJM) includes East and West portions of the Blue Cypress Marsh Complex and surrounding wetlands. Loxahatchee Slough includes Grassy Waters (GW), Hungryland Wildlife Management Area (WMA), Corbett WMA, and surrounding wetlands. Everglades includes Everglades National Park (ENP), Big Cypress National Preserve (BICY), Water Conservation Area (WCA) 1, WCA2A, WCA2B, WCA3A, WCA3B, and the Stormwater Treatment Areas (STAs).

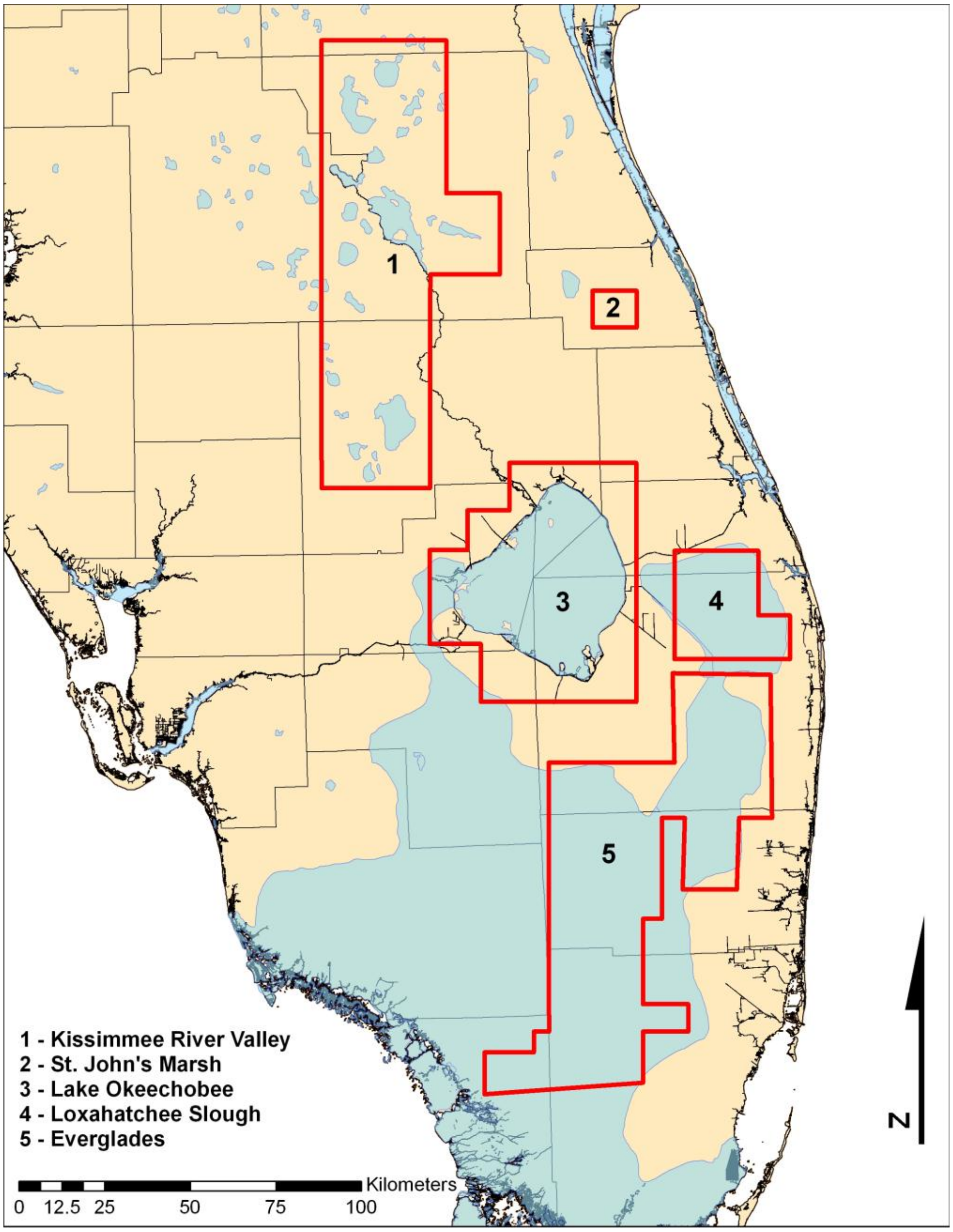


Figure 2. Number of young fledged, 1992-2011. Note that these values represent raw counts (uncorrected for detection) of young that reached the minimum fledging age (i.e., 24 days old) in monitored nests. KRV = Kissimmee River Valley; SJM = St. John's Marsh; Everglades includes Everglades National Park, Big Cypress National Preserve, and all WCAs and STAs.

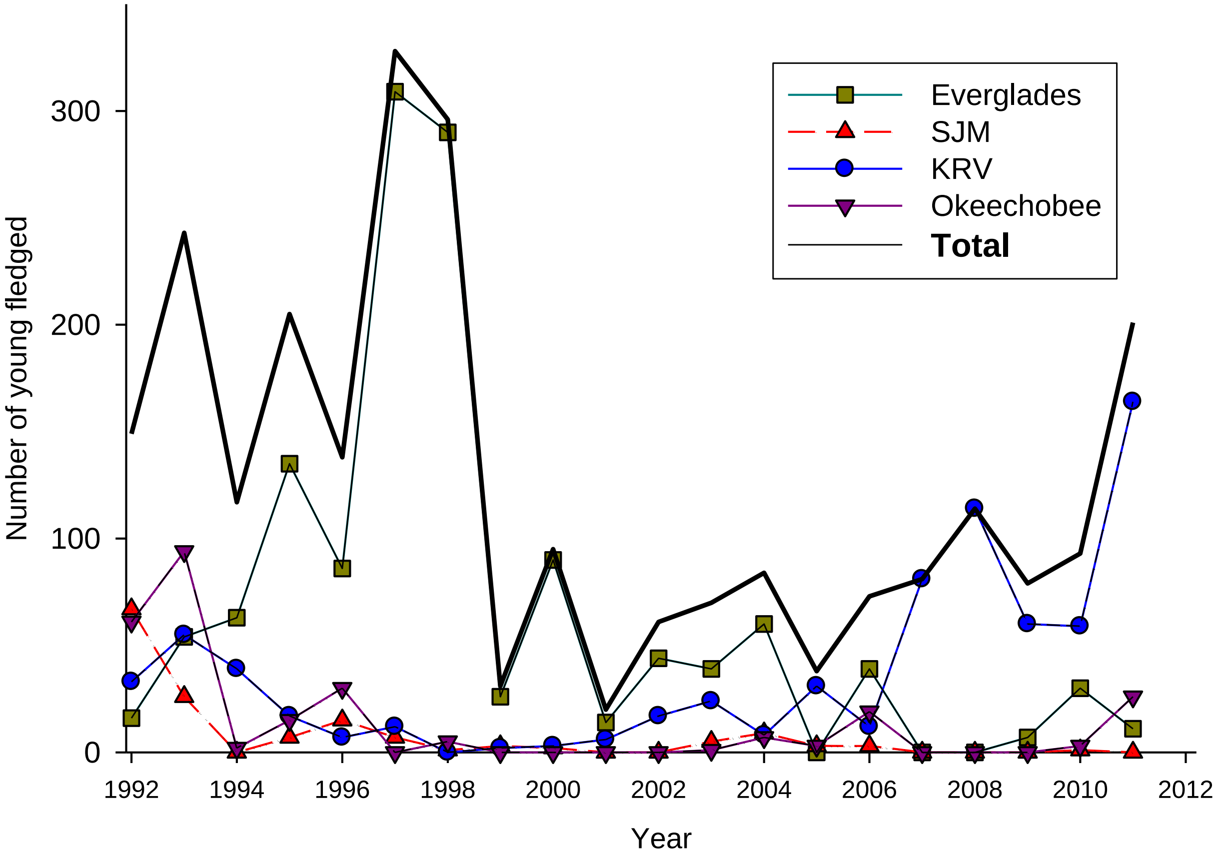


Figure 3. Relative annual contribution of each wetland region to the total number of (a) active nests, (b) successful nests, and (c) young fledged, on a range-wide basis from 1995-2011. KRV = Kissimmee River Valley; SJM = St. John's Marsh; LOX = Loxahatchee Slough; OKEE = Lake Okeechobee; EVER = Everglades (including STAs).

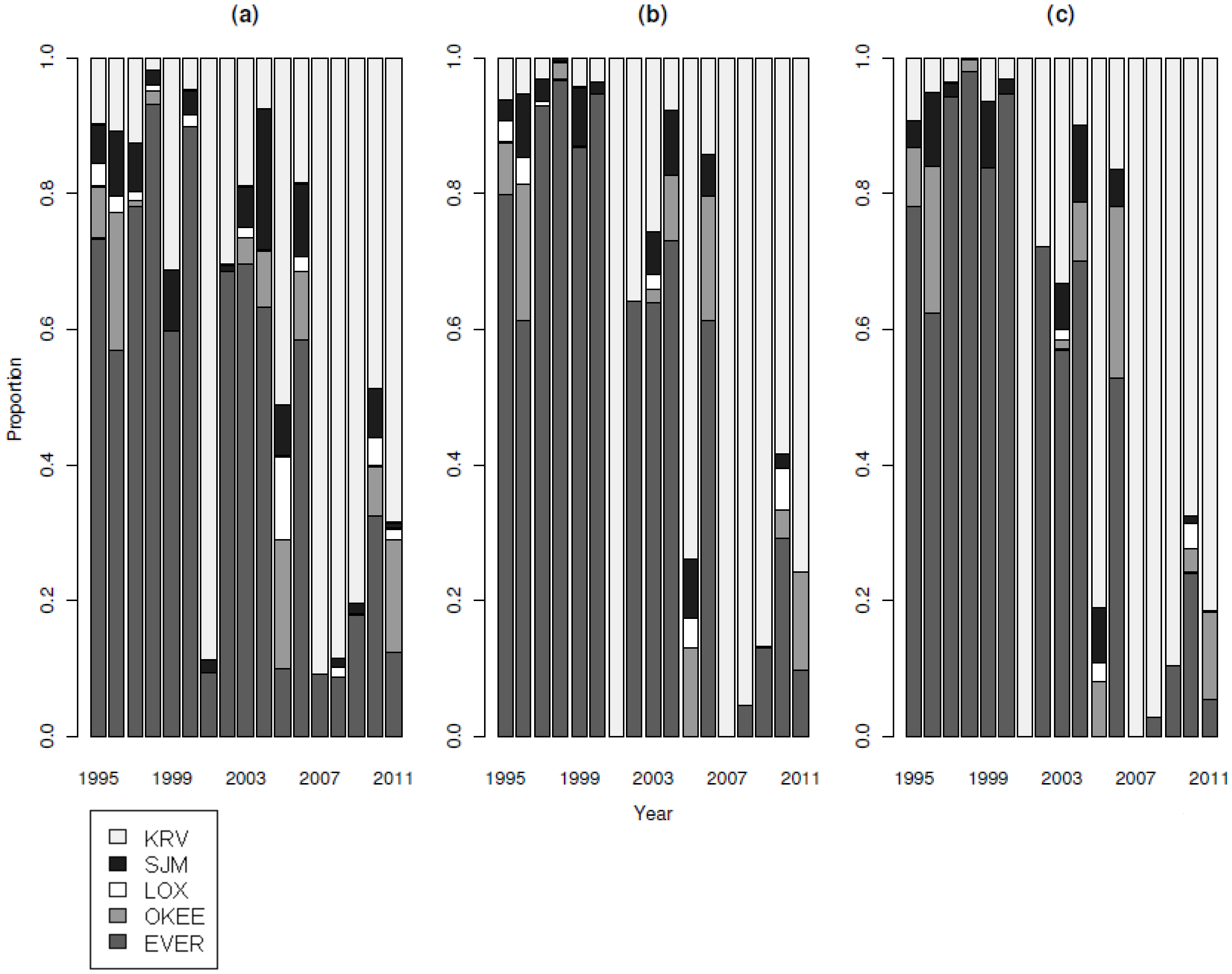


Figure 4. Nest success (± 95% CI), 1995-2011 in (a) WCA3A, and (b) range-wide. Note that in 2005, 2007 and 2010, all nests in WCA3A failed, while in 2001 and 2008, no active nests were observed in WCA3A.

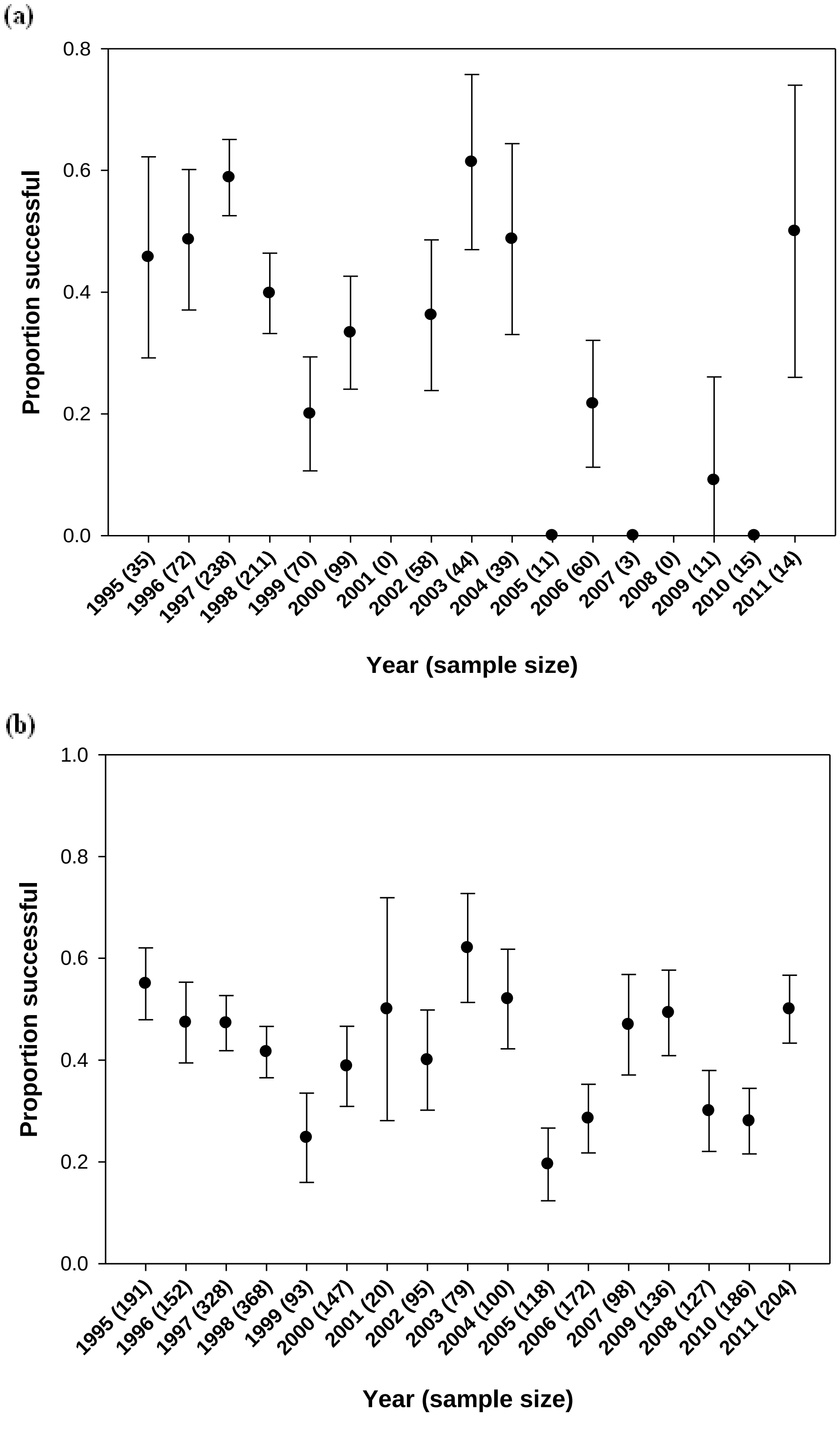


Figure 5. Mean (± 95% CI) number of young fledged per successful nest in select wetlands during 2011. Note small sample sizes for Lakes Kissimmee (KISS) and Istokpoga (IST).

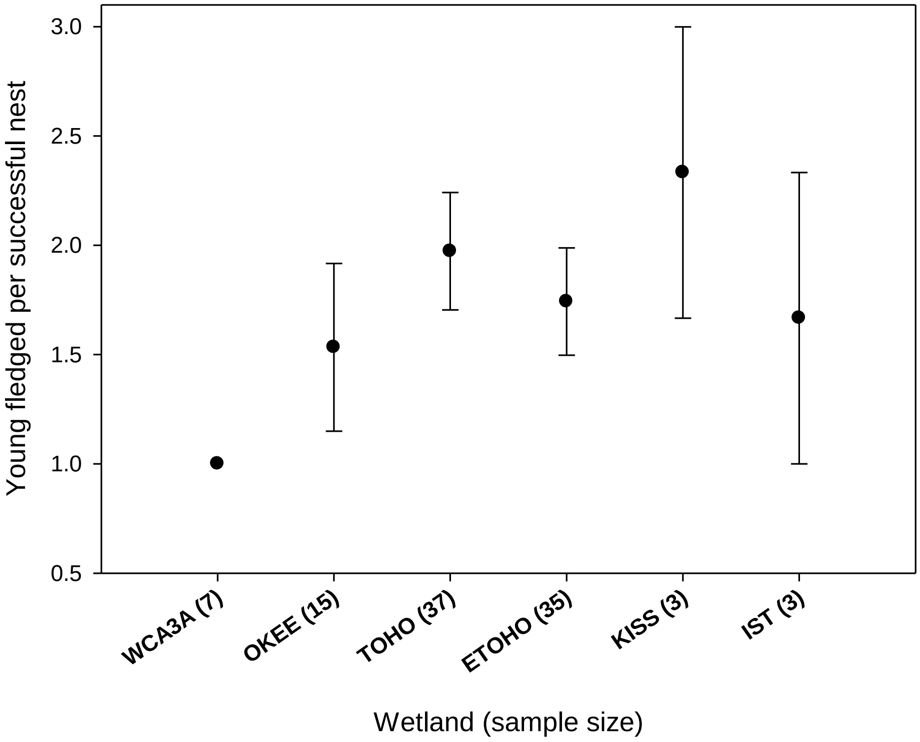


Figure 6. Mean (± SE) number of young fledged per successful nest in WCA3A, 1995-2011. No successful nests were observed in WCA3A in 2005, 2007, 2008 or 2010. Only one successful nest was observed in 2009 (white circle). All other years (black circles) had sample sizes greater than six. Lines (solid = model predicted, dashed = standard error of the model fit) represent results from a simple linear regression of mean nest productivity as a function of time.

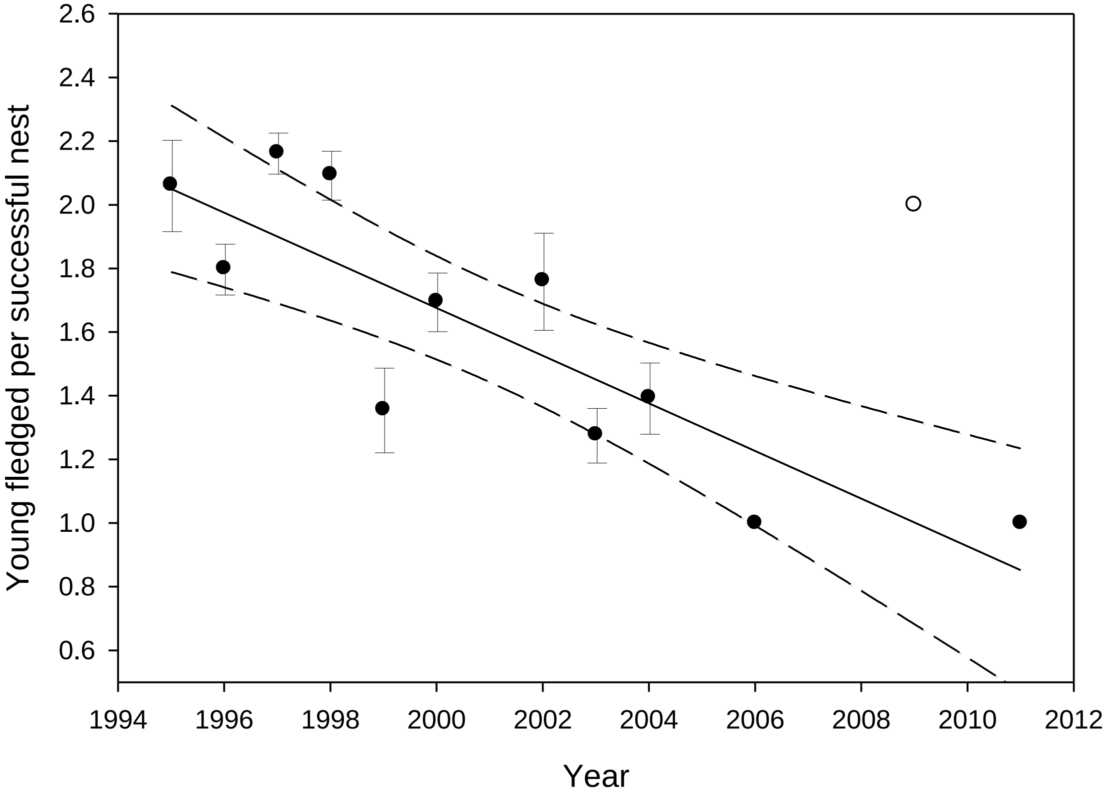
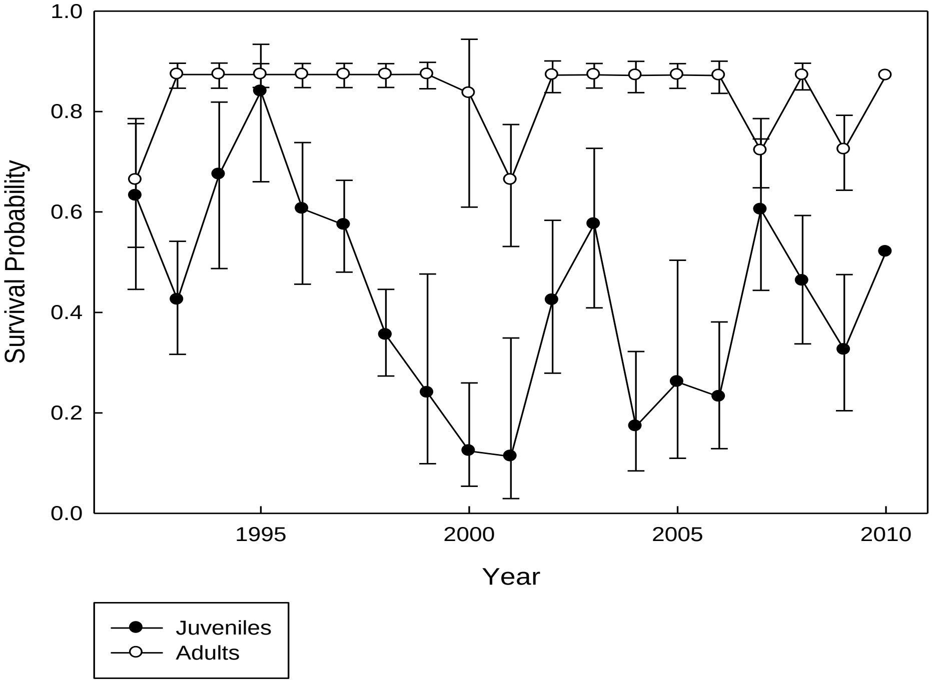


Figure 7. (a) Model averaged **e**stimates of adult (white circles) and juvenile (black circles) survival, 1992-2010; (b) Detection probability, 1992-2010. Error bars correspond to 95% confidence intervals.

(a).



(b).

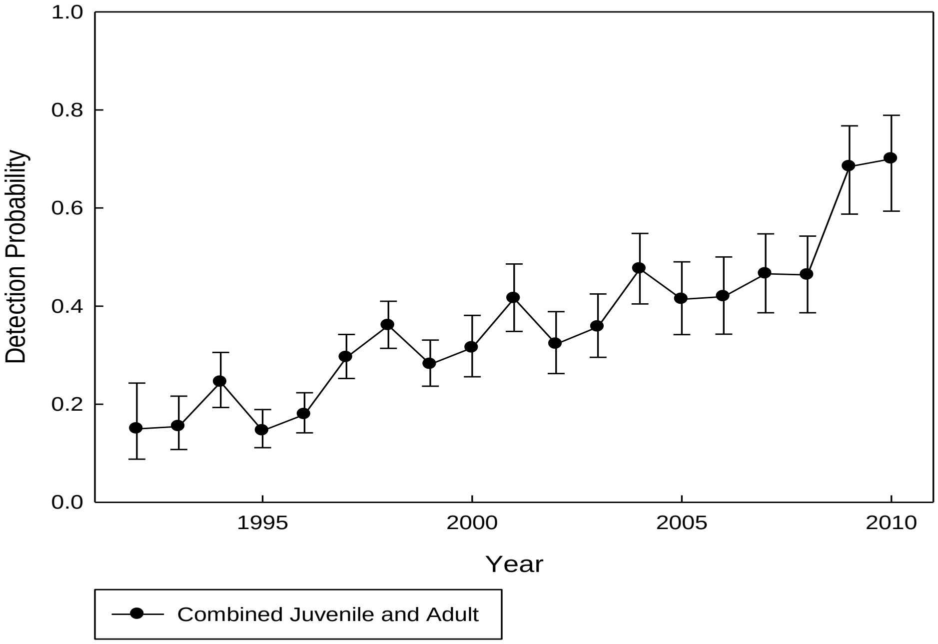


Figure 8. Population size of Snail Kites, 1997-2011, estimated using the super-population approach (Dreitz et al. 2002; Martin et al. 2007c).

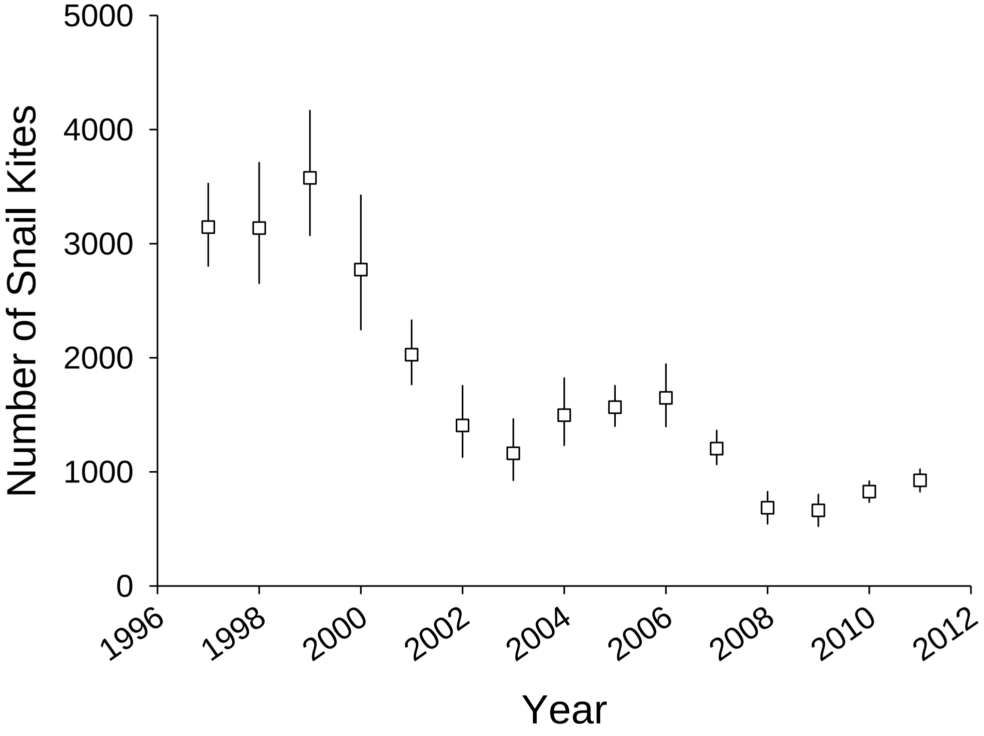


Figure 9. Proportion (± SE) of snails collected at kite nests in 2011 that were exotic. Sample size for each wetland is equal to the number of nests from which shells were collected.

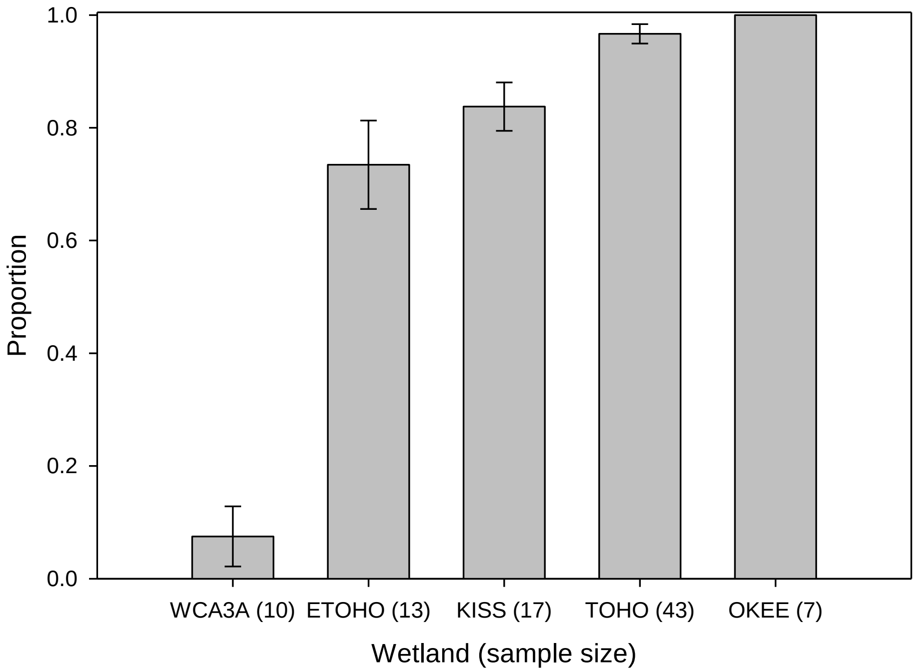
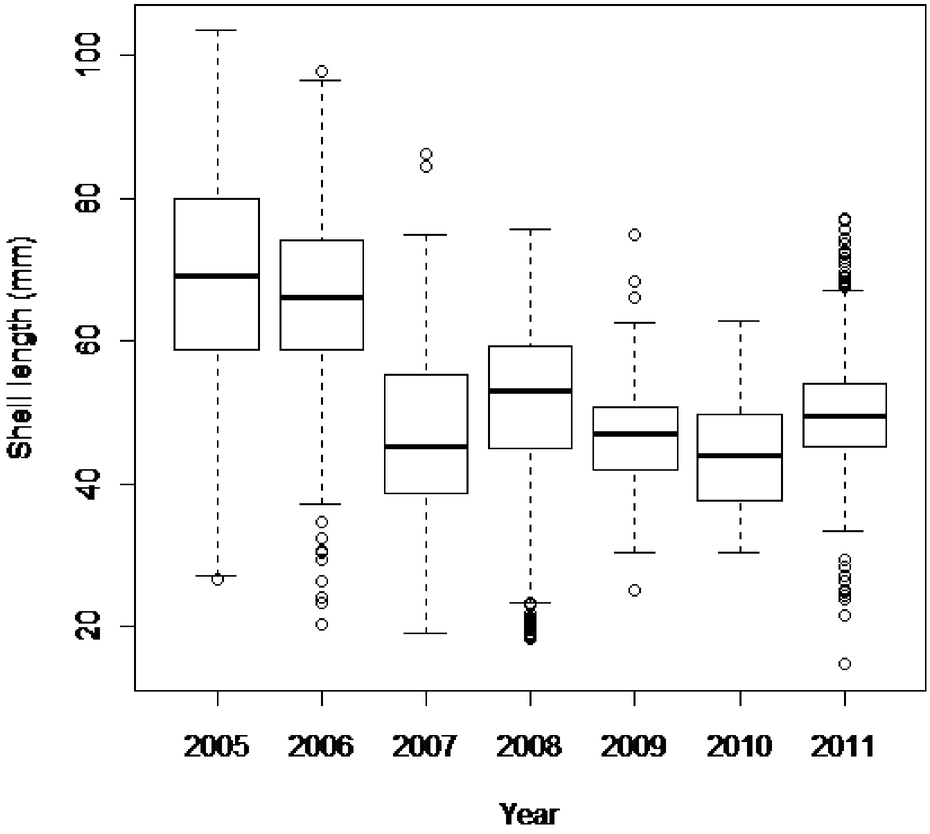


Figure 10. Median shell size of exotic snails consumed by snail kites on Toho, 2005-2011. Shells were collected under feeding perches and at nests. Note the decrease in median size starting in 2007.



APPENDIX A. MOVEMENT OF SNAIL KITES AND LANDSCAPE CONNECTIVITY

Landscape connectivity, or the degree to which a landscape can influence movement of organisms (Taylor et al. 1993), can greatly influence the viability of populations. Several models show that lower connectivity reduces population viability for a variety of organisms. Snail kites have often been described as a panmictic population; i.e., individuals can move freely throughout their range and, as a consequence, the entire range can be viewed as one ‘well-mixed’ population and connectivity is thought to be less important. This view stems from the fact that kites have been observed to move very long distances across Florida. Nonetheless, movements of kites, particularly during the breeding season, are more commonly localized and decline exponentially with distance (Fletcher et al. 2011; Fig. A1), such that short-term movement probabilities among wetlands are more heterogeneous and within-breeding season movements appear more limited than previously assumed. Because habitat suitability and prey availability are closely linked to local hydrology and can change dramatically within breeding seasons, such limited movement may influence population viability. Low water levels can cause kites to move in search for suitable habitat; these movements are associated with survival costs for younger, inexperienced individuals. By understanding these more localized movements, we may be able to better interpret changes in resource availability for kites across their geographic range and we may be able to identify functional populations clusters (e.g., the Kissimmee Chain of Lakes) that should be considered effective management units for the recovery of this endangered species.

Quantifying short-term, site-specific movements requires new tools for identifying relevant patterns for management. Previously, we have used multi-state mark-recapture models to estimate movement probabilities (e.g., Martin et al. 2006); these approaches are very useful but require estimating movements at relatively coarse scales (e.g., one must pool several sites to simply estimate movement among regions). We have recently developed an alternative approach that we have validated and found that it can accurately predict site-specific movement rates (Fletcher et al. 2011). This approach uses social network analysis to formulate the problem of movement across landscapes and can readily be applied to kite movement data. This approach is effective for: 1) identifying factors influencing site-specific movement rates; and 2) predicting movement in new wetlands or in years where we did not collect data on movement. We have found that the most effective social network model for kite movement incorporates heterogeneous movement rates to and from wetlands (a mixed-effects, ‘sender-receiver’ model in

Fletcher et al. 2011).

This new approach shows that: 1) within-breeding season movements are better described by assuming movement declines exponentially with distance; and 2) site-specific movement rates are highly variable beyond distance alone, suggesting other site-level factors likely influence movement rates (Table A1). This heterogeneity cannot be described by distance between wetlands or wetland size alone and suggests that other, possibly unmeasured factors are likely contributing to movement rates (Table S2 in Fletcher et al. 2011). We are currently expanding this approach to assess: 1) other site-specific (e.g., wetland type) and time-specific (e.g., drought versus non-drought years) factors influencing movement rates; 2) wetland (site) importance for connectivity across the geographic range and if (and how) it has changed over time; and 3) if changes in movement rates influence population viability assessments.

Table A1. Model selection, based on Akaike's Information Criterion (AIC), for each model predicting the probability of movement between wetlands (2005-2009).

	P	-2LL	AIC	ΔAIC
Constant movement rate among wetlands	1	289.1	291.1	144.6
Movement declines exponentially with distance	1	180.0	182.0	35.5
Site-specific movement + distance	4	138.5	146.5	0.0

Notes: *P* = number of estimated parameters; -2LL = -2 times the log-likelihood (the deviance); AIC = -2LL +2P; ΔAIC_i = AIC for model *i* minus the minimum AIC in the candidate models we compared. Smaller AIC values reflect more parsimonious (better fitting) models, given the movement data. See Fletcher et al. (2011) for more details.

MOVEMENT OF SNAIL KITES AND EFFECTIVE MANAGEMENT UNITS

We are further using these network approaches to identify effective management units for kites, based on the patterns of movements among wetlands. To do so, we are applying the concept of 'modularity' to kite movements (Fig. A2). Modularity describes the tendency for movement to be aggregated among wetlands, such that clusters of wetlands occur where movements are common within modules but are infrequent among modules. This approach is unique for identifying effective management units because: 1) no a priori knowledge of module location, number or strength is needed (e.g., if no modules occur for kites, this approach will say so); 2) modules can be of different sizes and configurations; and 3) the identification of modules ignores distance, such that other important factors driving aggregations of movement will be included. This concept also suggests an intermediate scale—above the site but below the entire geographic range—for population dynamics of species.

If snail kites show modular structure in movements, there are several implications. First, we may need to consider management implications beyond specific sites that will influence kites within modules. For example, management regimes on Lake Kissimmee may be highly influential to kites on Lake Toho, if these two lakes are located within the same module (we expect they are). Second, understanding wetland importance for the recovery of kites may change, because wetlands critical to maintaining cohesion between modules may be critically important for long-term viability (as opposed to those that are critical within modules). Third, modular structure in movements has been shown to be highly influential to population viability in comparison to non-modularity structure, suggesting that by understanding this scale of population dynamics, we will be able to better interpret the viability of the kite and how management regimes will influence the recovery of this species.

We are currently developing algorithms to appropriately identify modules and determine if these effective units for population management are consistent over time as the population size has declined and under drought and non-drought conditions. We are refining algorithms from engineering and physics that attempt to find the best partitions (modules) for a network (Guimera and Amaral 2005). Initial analyses based on the movement data of Fletcher et al. (2011) suggest strong modularity of kite movements ($P<0.01$), where two modules have been identified: 1) the Kissimmee Chain of Lakes; and 2) the remaining wetlands where kites breed in central-south Florida. These effective management units most closely coincide with the type of wetland (lacustrine versus plaustrine) and suggest that these two types of wetlands should be considered separately for kite management.

Figure A1. Observed within-breeding season movement distributions (gray bars) for snail kites (2005-2009) and a movement kernel taken from the literature. Movement kernel is based on a negative exponential function, where the probability of movement between two patches is exp(-αd_{ij}). We calculated α as the inverse of the average movement distance observed in the literature.

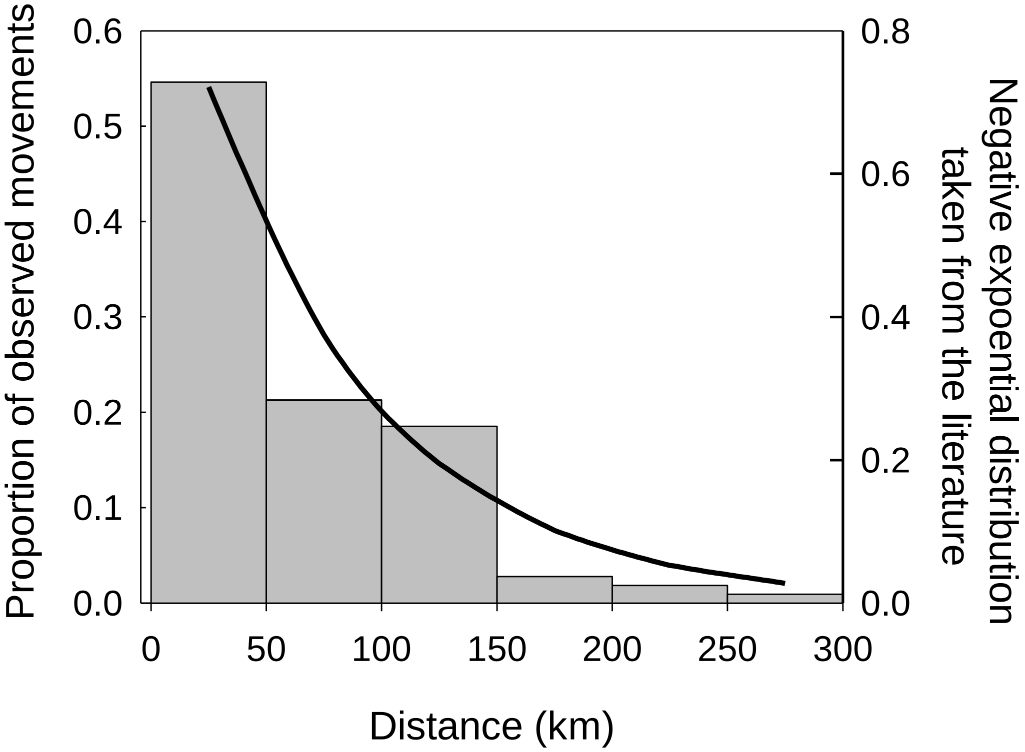
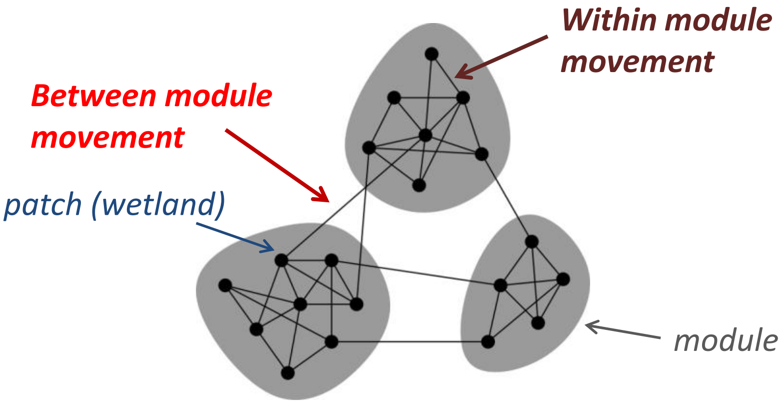


Figure A2. The concept of modularity in populations. In this hypothetical situation, three modules occur, based on the movement of individuals among patches (wetlands). Modules represent 'effective population units' that are identified by movement.



APPENDIX B. LAKE OKEECHOBEE SNAIL KITE NESTING

Water levels on Lake Okeechobee were more than two feet (NGVD) below optimal target levels at the start of 2011 and remained low throughout the Snail Kite breeding season. Temporary forward pumps were activated on May 31st (at which time the lake was still approximately two feet below optimal target levels) and ran through June 24th, removing an additional six inches of water. Active Snail Kite nests were observed in Okeechobee from February 10th through June 15th, with nest fates and juvenile survival potentially being affected by low water levels.

While our ability to robustly assess the survival of juveniles fledged in Okeechobee in 2011 will require additional years of band resight data, it should be noted that when the the forward pumps were turned on, the majority of young were less than 90 days old (Fig. 1), which is the period of highest mortality for fledglings (Bennetts et al. 1998).

We test for an effect of daily water stage (obtained from DBHYDRO; station = LOKEE) on daily survival rate (DSR) of snail kite nests in Okeechobee using Program Mark, and we found a significant positive effect ($\beta = 1.54$, SE = 0.40; Fig. 2).

Figure B1. Age distribution of snail kite nestlings/fledglings produced in Lake Okeechobee at the time forward pumps were activated (on May 31st 2011).

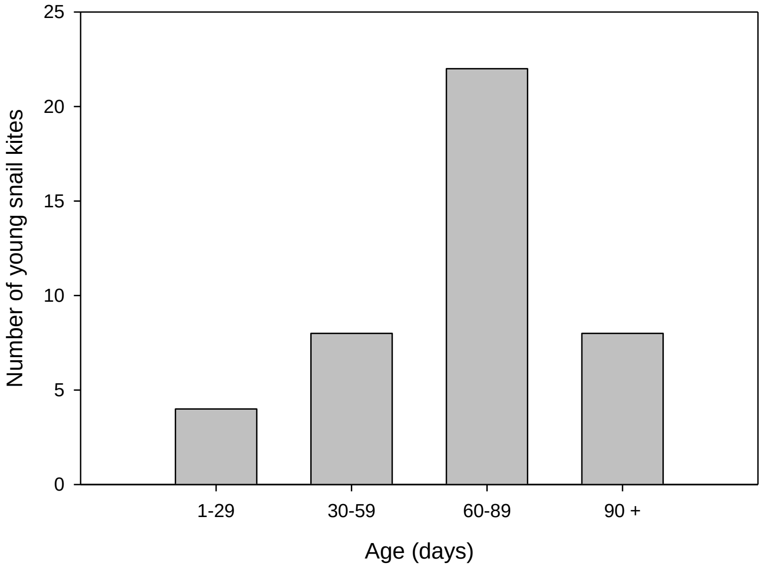
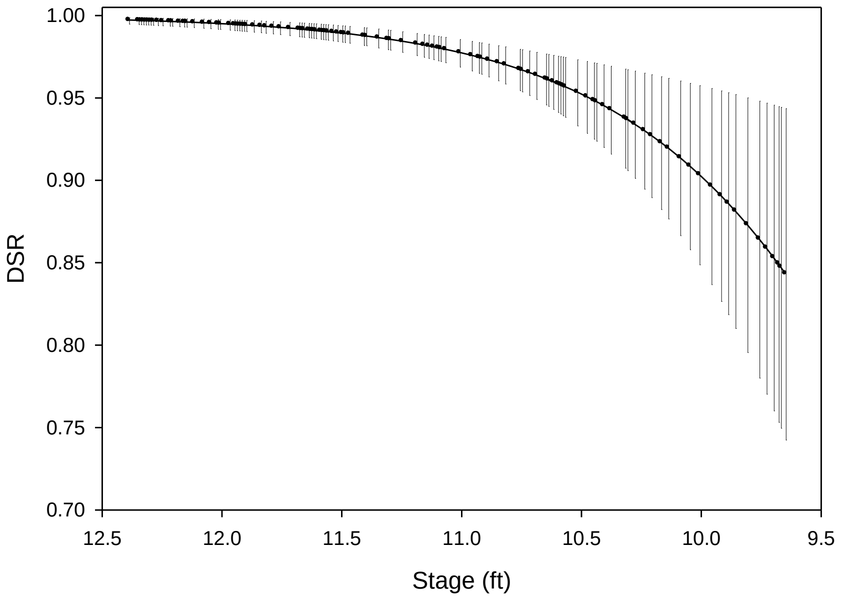


Figure B2. Modeled daily survival rate of snail kite nests on Okeechobee during 2011 as a function of daily water stage.



APPENDIX C. RECONSTRUCTING HISTORICAL HABITAT WITH MULTISTATE MODELS

Abstract

Historical vegetation data is important to ecological studies, as many structuring processes operate at long time scales, from decades to centuries. Capturing the pattern of variability within a system (enough to declare a significant change from past to present) relies on correct assumptions about the temporal scale of the processes involved. Sufficient long-term data is often lacking and current techniques have their weaknesses. To address this concern, we constructed multistate models to provide community transition probabilities to hindcast vegetation communities considered critical foraging habitat for an endangered bird, the Florida snail kite. This is the first use of multistate models to hindcast vegetation data and the technique shows great promise. Results from our hindcast closely mirrored the population collapse of the snail kite population using only environmental data to inform the model. This gives us confidence in the hindcasting results and their use in future demographic models.

Keywords: Multistate models, Everglades, vegetation communities, hindcasting, snail kite

Introduction

The global interest in vegetation/land cover change has highlighted the importance of historical or pre-disturbance conditions for perturbed vegetative landscapes, particularly wetlands. Historical vegetation data is important to ecological (particularly restoration) studies since many structuring processes operate at long time scales, from decades to centuries (Lindborg and Eriksson 2004, Magnuson 1990). If data has not been collected at the appropriate temporal scale for the system, a critical ecological process could be missed, resulting in misleading analyses (Tillman 1989, Jackson et al. 2001). Capturing the pattern of variability within a system (enough to declare a significant change from past to present) relies on correct assumptions about the temporal scale of the processes involved (Collins et al. 2000). This is particularly important in succession studies (Collins et al. 2000), restoration ecology (Jackson et al. 2001), or when examining a system before/after unpredictable disturbance events such as fire, floods, or hurricanes (Michener et al. 1997).

Ecologists currently rely on techniques such as chronosequencing (Lebrija-Trejos et al. 2010) or historical ecology (Swetnam et al. 1999) to deal with missing temporal habitat data, each with their own strengths and limitations (Johnson and Miyanishi 2008, Szabo and Hedl 2011). Remote sensing can also be a useful tool, as the LANDSAT set of satellites has a 30+ year period of record, but its 30 meter resolution limits its usefulness for communities at a finer spatial scale. Other modeling techniques used for prediction (Guisan and Zimmerman 2000) might also be useful for hindcasting. This has been accomplished in an ecological context using a number of different methods, including linear regression, multiple regression, and other previously validated models such as biophysical models (Baron 2006, Tulp and Schekkerman 2008, Wethy and Woodin 2008, Greene et al. 2009).

At the habitat community level, we were particularly interested in one modeling technique, multistate models, that had been used in a predictive capacity within a Markovian framework (Breininger et al. 2010), but not for hindcasting. Multistate models were originally developed to deal with multiple states of wildlife (physiological or spatial states) in survival analyses (Hestbeck et al. 1991, Nichols et al. 1994), but have been applied in a limited way to vegetative habitats (Hotaling et al. 2009, Breininger et al. 2010). They use a likelihood-based approach to model transition probabilities between discrete states over time and can accommodate missing data (Lebreton and Pradel 2002). Multistate models can directly estimate environmental parameters, past and present, connected with state/community changes (Breininger et al. 2010), a main goal in vegetation ecology studies.

We explored the use of multistate models to extend the period of record for habitat data of the endangered Everglades snail kite. Demographic data has been collected from the mid 1990’s to the present (Reichert et al. 2010), and a population crash in 1999-2001 halved the population (Martin et al. 2007). One hypothesis for the crash was the degradation of the kite’s critical habitat, most importantly Water Conservation Area 3A South (WCA3AS) (Martin et al. 2008). This has been postulated, but never substantiated. A habitat study was initiated in the critical habitat area in 2002 to track changes in foraging habitat. Unfortunately, the snail kite population greatly reduced its use of this area in the last decade, and there was not enough concurrent data sufficient for robust estimation of snail kite demographics in a traditional manner. We used the habitat data to construct a multistate model to estimate transition probabilities for one vegetative community state, associated with foraging quality, to another for each year as a function of hydrology, as hydrology is the driving factor for habitat change within the snail kite critical habitat (Kitchens et al. 2002). We then explored the potential of multistate modeling to hindcast vegetation communities into the 1990's to provide demographic/habitat overlap. We also examined the proportion of quality foraging habitat to degraded foraging habitat before and after the demographic decline to determine if there was a change in habitats that might affect the snail kite population. The principal application of the modeled data will be in future demographic analyses to improve snail kite demographic models.

Methods

WCA3AS was the largest and most consistently used component of the habitat designated critical to the Snail Kite (Figure 1; Kitchens et al. 2002, Mooij et al. 2002). Its historic contribution to kite reproduction is significant (Kitchens et al. 2002). Shifts of Snail Kite nesting density up the slight, but significant, elevation gradient (Zweig et al. 2011) in WCA3AS have been documented over the past two decades (Bennetts and Kitchens 1997). This is presumably in response to degradation of nesting or foraging habitat as a result of sustained high water levels from impoundment and water management (Kitchens et al. 2002).

To monitor foraging habitat, we used slough and wet prairie samples in WCA3AS from the larger-scale vegetation study (Zweig and Kitchens 2008). Twenty 1-km^2 plots (Figure 1) were placed in a stratified random manner across the landscape gradients in WCA3AS. Plots were stratified by the landscape level gradients of peat depth, water depth and snail kite nesting activity. Five a priori physiognomic types were identified: slough, sawgrass, tree/shrub island, cattail, and wet prairie. Two or three transects in each plot were placed perpendicular to ecotones, beginning in one a priori type and terminating in another, e.g., slough to sawgrass. We collected 0.25 m^2 samples of all standing biomass along a belt transect, clipping the vegetation at peat level at 3 m intervals, and included any submerged aquatic plants within the sample. Samples were collected from each transect in every plot (~1200 samples) in the wet season (November/December) of each year from 2002-2005. A reduced sampling design (~400 samples, November only) was implemented in 2006-2007 and continues to the present. Plant samples were sorted by species, counted, dried to a constant weight, and weighed to the nearest 0.1 g. Approximately 14,000 samples were collected and processed between 2002 and 2010.

Hydrology

Hydrologic variables were calculated from two U.S. Geological Survey wells in WCA3AS (Sites 64 and 65) from 1991-2009. We created a list of variables (Table 1) that might influence vegetation communities, including factors that incorporate time lags or hydrologic legacy (Zweig and Kitchens 2008). There were two reasons for choosing the time period 1991-2009. First, a change in maximum and average water depths (whether management or climatic) occurred in WCA3AS around this time (Zweig et al. 2011), therefore using any data before 1991 would skew the range of hydrologic data. Second, this allowed us to hindcast vegetation data to 1994 and still assess the relevance of time-lagged variables up to 4 years previous to the sample (Zweig and Kitchens 2008).

Multivariate and Multistate Analysis

To account for high densities of low biomass species and high biomass of low density species, the data were relativized in an index, importance value (IV), calculated by:

$$\text{IV for species i} = ((Rd_i + Rb_i)/2)*100,$$

where Rd_i is the relative density of species i and Rb_i is the relative biomass of species i. Relative measures are the sum of biomass or density of species i divided by the sum of biomass or density of all species within each sample.

We combined the a priori slough/wet prairie 0.25 m^2 samples on each transect, as these are foraging habitat for the snail kite and the communities of interest, into one point ($n = 351$) and performed a hierarchical cluster analysis on the IV data with a Sorenson distance measure and flexible beta of -0.25 in PC-ORD (McCune and Mefford 1999). To choose how many clusters were present during the study period, we ran an indicator species analysis (ISA) (Dufrêne and Legendre 1997) to prune the cluster dendrogram.

We modeled annual transition probabilities from 2002 to 2009 for the 3 states from the cluster analysis (*Eleocharis elongata* slough (Elg), *Nymphaea odorata* slough (Nyo), and *Eleocharis cellulosa* prairie (Elc)) using a likelihood-based multistate analysis in Program MARK (version 6.1, White and Burnham 1999). All transitions were modeled to occur (Figure 2). No misclassification was assumed due to the nature of the plant sampling. The 'multistate recaptures only' model in MARK estimates survival (S), detection (p), and annual transition probabilities (ψ). We constrained both survival and detection to be equal to 1, as we assumed perfect detection (the plants were harvested and sorted to species) and that a site will always survive from one sample to the next. We added peat depth as a site-dependent covariate and hydrologic values (Table 1) based on previous analyses (Zweig and Kitchens 2008, Zweig and Kitchens 2009) were added as time-varying covariates to the model.

We constructed models to test for the effect of peat and hydrologic factors, or a combination thereof, on conditional transition probabilities (conditional on the state at t for t+1) between vegetation communities and tested them against null models and a peat-only model. The null model assumed that the transition probabilities (ψ^{ElgElg}, ψ^{ElgNyo}, ψ^{ElgElc}, etc.) were constant through time. The peat-only model assumes the transition probabilities are a linear function of peat. We were interested in the best fit model for predicting and hindcasting and thus the most accurate model, and we considered the most accurate to be the one that included the lowest value ΔAIC, highest AICc weight, and most significant beta parameters (Table 1).

Hindcasting, Predicting, and Comparing Pre- and Post-Population Decline

We considered the state dynamics within our study area to be Markovian, so that the state in time t+1 is dependent on the state at time t (Breininger et al. 2010).Using this and the conditional transition probabilities calculated from our best fit model, we can hindcast the number of sites within each state within our study area (Breininger et al. 2010). The number of sites (n) in each state at time t and t+1 can be defined as a vector and the transition probabilities (ψ) from time t to t+1 as a matrix:

$n_t^{Elg} @ n_t$

which is more easily modified in matrix notation:

$$n_{t+1} = \psi_t n_t .$$

For our analysis, we had n_{t+1} and could calculate ψ_t from our model using historic hydrologic values, but wanted to know n_t. Using matrix algebra, we solved for n_t by dividing each side of the equation by the inverse of ψ_t, using the solve function in R (version 2.1.3.0, CRAN 2011):

$$\psi_t^{-1} n_t = n_{t-1}$$

This is possible because our matrix is square and the determinant of any matrix is not zero. Past transition matrices were calculated from the model's parameters using linear equations:

$$[logit(\psi]^{ElgNyo}) = \beta_0 + \beta_1(hydroperiod_{t-1})$$

$$[logit(\psi]^{ElgElc}) = \beta_0 + \beta_1(moddry_{t-1})$$

$$[logit(\psi]^{NyoElg}) = \beta_2 + \beta_3(peat) + \beta_4(max1yrprev_{t-1})$$

$$[logit(\psi]^{NyoElc}) = \beta_5 + \beta_6(peat) + \beta_4(max1yrprev_{t-1})$$

$$[logit(\psi]^{ElcElg}) = \beta_7 + \beta_8(peat) + \beta_9(days > 50\%ile1yrprev_{t-1})$$

$$[logit(\psi]^{ElcNyo}) = \beta_{10} + \beta_9(max1yrprev_{t-1})$$

Transitions were solved for by taking the log of each equation. Transitions for ψ^{ElgElg}, ψ^{NyoNyo}, and ψ^{ElcElc} were calculated by subtraction ($\psi^{ElgElg} + \psi^{ElgNyo} + \psi^{ElgElc} = 1$). The actual total number of sites samples each year from the full samples was 58. The resulting vector for n_{t-1} always summed to 58, but some of the states had negative numbers associated with that time period. Each vector was scaled by adding the lowest negative number (k) to each member of the vector, dividing each by the sum of the vector (s), and then multiplying each member of the vector by 58, e.g. for *Nymphaea odorata* slough: $\frac{n^{Nyo} + k}{s} * 58$. Each new vector was used in the next hindcast until 1994.

A reduced sampling protocol was introduced in 1996. All samples were used to create the multistate model, but comparing pre- and post-snail kite population crash would require full sample numbers for all years. We used annual transition probabilities from the model and equation 2 to predict full sample numbers for 2006-2009. The resulting annual totals for each state (*E. elongata* slough, *N. odorata* slough, and *E. cellulosa* prairie) from 1994-2000 and 2001-2009 were compared with a Wilcoxon signed-rank test.

Results

Multivariate and Multistate Analysis

The cluster analysis and indicator species analysis suggested 3 states, an emergent slough dominated by *E.elongata* (Elg), a slough dominated by *N. odorata* (Nyo), and a wet prairie dominated by *E.cellulosa* (Elc). The best model for transitions between these three states was:

$S(1)p(1)\ \psi^{\dagger}ElgNyo\ (hydroperiod)\ \psi^{\dagger}ElgElc\ (moddry)\ \psi^{\dagger}NyoElg\ (peat + max1yrprev)\ \psi^{\dagger}NyoElc\ (pea$

,

where environmental variables are within parentheses (Table 1). Hydroperiod had a positive relationship with ψ^{ElgNyo} , and modddry was also positively related to ψ^{ElgElc} . Peat and max1yrprev were negatively related to ψ^{NyoElg} and ψ^{NyoElc} (Table 2). Peat and days>50th%ile were positively related to ψ^{ElcElg} and days>95th%ile had a positive relationship with ψ^{ElcNyo} . Several top models had a ΔAIC between 0-2 and similar weights, but this model was the only one with significant parameters (confidence interval did not include 0) for all hydrologic factors. We feel this is important considering its use as a predictive model. As a rule, the highest transition probabilities from 2002-2009 were that a community would stay the same from one year to the next (ψ^{ElgElg} , ψ^{NyoNyo} , ψ^{ElcElc}). The probabilities for ψ^{NyoElc} were very low (<0.01%) and were considered to be zero.

Hindcasting, Predicting, and Comparing Pre- and Post-Population Decline

The *E.elongata* and *E.cellulosa* communities showed the most dramatic changes from 1994-2009 (Figure 3). *E.elongata* had a range of 0-27 sites per year and *E.cellulosa* had a range from 5-37 sites per year. In the hindcasting results, *E.elongata* communities were not present in our study sites from 1994-1998. *E.cellulosa* was the dominant community until 2001, where it decreased by more than half and continued declining to the present. The *N. odorata* community showed a low in 1999 and 2000 of 5 and 9, respectively. According to a Wilcoxon ranked-sum test of the pre- and post-kite decline periods (1994-2000 and 2001-2009), there was a significant decrease ($p = 0.001$) in the number of *E.cellulosa* communities in our study area and a significant increase ($p = 0.04$ and 0.001, respectively) in both *N. odorata*, and *E.elongata* communities (Figure 4).

Discussion

The deeper emergent community, *E.elongata*, is absent from our study sites from 1994-1998 and then becomes a main community from 1999-2009. In reality, *E.elongata* would not be absent from the entire landscape, but according to the model, was not represented within our study sites. The more shallow emergent community, *E.cellulosa*, that is likely better foraging habitat for the snail kite (Kitchens et al. 2002), decreases by 32% in 2000 and another 40% in 2001. *E.cellulosa* and *E.elongata* abruptly switch dominance in 2000 and 2001. After the decrease in snail kite population, *E.cellulosa* numbers continue to decline until the last year of the analysis. *E.elongata* loses dominance to the deepest slough community, *N. odorata*, in 2005, and the snail kite's critical habitat continues trending towards deeper communities from 2005-2009. The trajectory towards *N. odorata* sloughs (Figure 3) is similar to other types of analyses performed in this area of the Everglades (Zweig and Kitchens 2008, Zweig et al. 2011). The associated hydrologic and peat variables in the final model also make sense when taken into ecological context: Extended hydroperiods would increase the transition probability for *E.elongata*, a deeper emergent community, to *N. odorata*, the deepest slough; and moderate droughts would increase the transition probabilities of *E.elongata* to *E.cellulosa*, the drier emergent community. The shallower the peat depth, the greater the transition probabilities are from *N. odorata* to another community. *N. odorata* prefers deeper peat where it survives droughts in a tuber-like form (Zaremba and Lamont 1993). The deeper the maximum water depths are in the previous year, the less likely communities are to transition away from *N. odorata*. Higher peat depths and more days of water depths above the 50th percentile would cause transitions probabilities to be higher from *E.cellulosa* to *N. odorata*.

The increase of deeper foraging habitat is important to snail kite population dynamics, as there are lower densities of snails in slough habitat and it is also more difficult, with the lack of emergent structure, for snail kites to capture snails in sloughs (Karunaratne et al. 2006). The low density of snails in *N. odorata*-dominated habitat may reflect the lack of emergent structure for oviposition (Karunaratne et al. 2006) and the lack of epiphytic periphyton on *N. odorata* (Browder et al. 1994), a preferred food of the native apple snail (Sharfstein and Steinman 2001, Kitchens et al. 2002).

Modeling community state changes within our study area yielded a model with significant parameters for each environmental variable. We were able to both hindcast all sites and predict missing sites in the present with this model to expand our period of record of complete habitat data to 1994-2009 instead of only 2002-2005. Habitat results from our hindcast closely mirrored the population collapse of the snail kite population in 2000 and 2001 using only environmental data, and not snail kite population data, to inform the model (Table 1). We also used different hydrologic variables than were in the snail kite population analyses (Martin et al. 2008). This gives us confidence in the hindcasting results and their use in future demographic models.

Historical reference information is critical in ecological studies to account for spatial and temporal variation of patterns and processes (White and Walker 1997), and for setting attainable and sustainable restoration goals (Swetnam et al. 1999). Very few researchers have the luxury of a long-term dataset (Jackson et al. 2001, Kreyling et al. 2011), even when it is important to understanding the system. Hindcasting is one of several methods (Lotze and Worm 2008) used to reconstruct this missing reference information, and has been successful in several areas of ecology (Baron 2006, Tulp and Schekkerman 2008, Wethy and Woodin 2008, Greene et al. 2009). We were interested in hindcasting and predicting habitat dynamics for an endangered bird to support the hypothesis that a recent population crash was due to degradation of critical habitat (Martin et al. 2008). The population crash closely follows an abrupt change dominant habitat states, indicating relative success of our model and the lack of a large time lag between hydrologic/habitat changes and snail kite population responses, perhaps contraindicating any further extinction debt (Tillman et al. 1989) in the future. This type of multistate hindcasting could be useful in other systems that are lacking references habitat data.

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Table 1: Definitions for hydrologic abbreviations used in slough/wet prairie multistate model for Water Conservation Area 3A South, Florida, USA.

Hydrologic Variable (abbreviation)	Definition
Severe Drought (sevdry)	# of days (annually) that water level is below the average minimum for 1992-2009 - 1 standard deviation
Moderate Drought (moddry)	# of days (annually) that the water depth is below the average minimum for 1992-2009
Average (avg*n*yrprev)	Mean water depth (annual). Listed for current year or *n* years previous to the sample year
Max (max*n*yrprev)	Maximum water depth (annual). Listed for current year or *n* years previous to the sample year
Min (min*n*yrprev)	Minimum water depth (annual). Listed for current year or *n* years previous to the sample year
Severe Flood (sevflood)	# of days (annually) that water level is above the average maximum for 1992-2009 + 1 standard deviation
Moderate Flood (modflood)	# of days (annually) that water level is above the average maximum for 1992-2009
Days > 45cm	# of days water depth is above 45cm (annual)
Days > 30cm	# of days water depth is above 30cm (annual)
Hydroperiod (hydroperiod)	# of days water depth is above 0cm (annual)
Days < *n*th%ile	# of days water depth is less than a percentile calculated from 1992-2009. Calculated 5, 10, 20, 25, 50, 75, 80, 90, 95th percentile and listed for current year or *n* years previous to the sample year
Days > *n*th%ile	# of days water depth is more than a percentile calculated from 1992-2009. Calculated 5, 10, 20, 25, 50, 75, 80, 90, 95th percentile and listed for current year or *n* years previous to the sample year

Table 2: Top 9 multistate models indicating hypotheses about the effect of environmental variables on vegetation community transitions, and three null models (dot, time, peat), in Water Conservation Area 3A South, Florida, USA. Hydrologic variables are calculated from 1992-2009. Variables are defined in Table 1.

Model	AICc	ΔAIC	AICc weights	No. parms
$S(1)p(1)\ \psi^{\dagger}ElgNyo\ (hydroperiod)\ \psi^{\dagger}ElgElc\ (moddry)\ \psi^{\dagger}NyoElg\ (peat + max1yrprev)\ \psi^{\dagger}NyoElc\ (peat + max1yrprev)\ \psi^{\dagger}ElcElg\ (peat + Da$	281.67	0.0	0.238	13
$S(1)p(1)\ \psi^{\dagger}ElgNyo\ (peat +\ hydroperiod)\ \psi^{\dagger}ElgElc\ (peat +\ \ eat\ l\ hypothesis"$	282.11	0.44	0.191	15
$S(1)p(1)\ \psi^{\dagger}ElgNyo\ (hydroperiod)\ \psi^{\dagger}ElgElc\ (peat +\ \ eat\ l\ hypothesis"$	282.93	1.26	0.127	14
$S(1)p(1)\ \psi^{\dagger}ElgNyo\ (hydroperiod)\ \psi^{\dagger}ElgElc\ (\ eat\ l\ hypothesis"$ $\psi^{NyoElg}(max1yrprev)\psi^{NyoElc}(max1yrprev)\psi^{ElcElg}(Days > 50\%ile1yrprev)\psi^{ElcNyo}(Days > 95\%ile1yrprev)$	283.64	1.97	0.09	16
$S(1)p(1)\psi(peat)$	283.98	2.32	0.075	13
$S(1)p(1)\psi(> 95\%ile1yrprev)$	285.09	3.42	0.043	16
$S(1)p(1)\psi(Max1yrprev)$	286.59	4.92	0.021	16
$S(1)p(1)\psi(> 90\%ile1yrprev)$	286.72	5.05	0.019	16
$S(1)p(1)\psi(> 75\%ile1yrprev)$	286.76	5.09	0.020	16
$S(1)p(1)\psi(> 50\%ile1yrprev)$	286.94	5.27	0.017	16
$S(1)p(1)\psi(.)$	301.01	19.34	0.00	7
$S(1)p(1)\psi(t)$	301.84	20.18	0.00	21

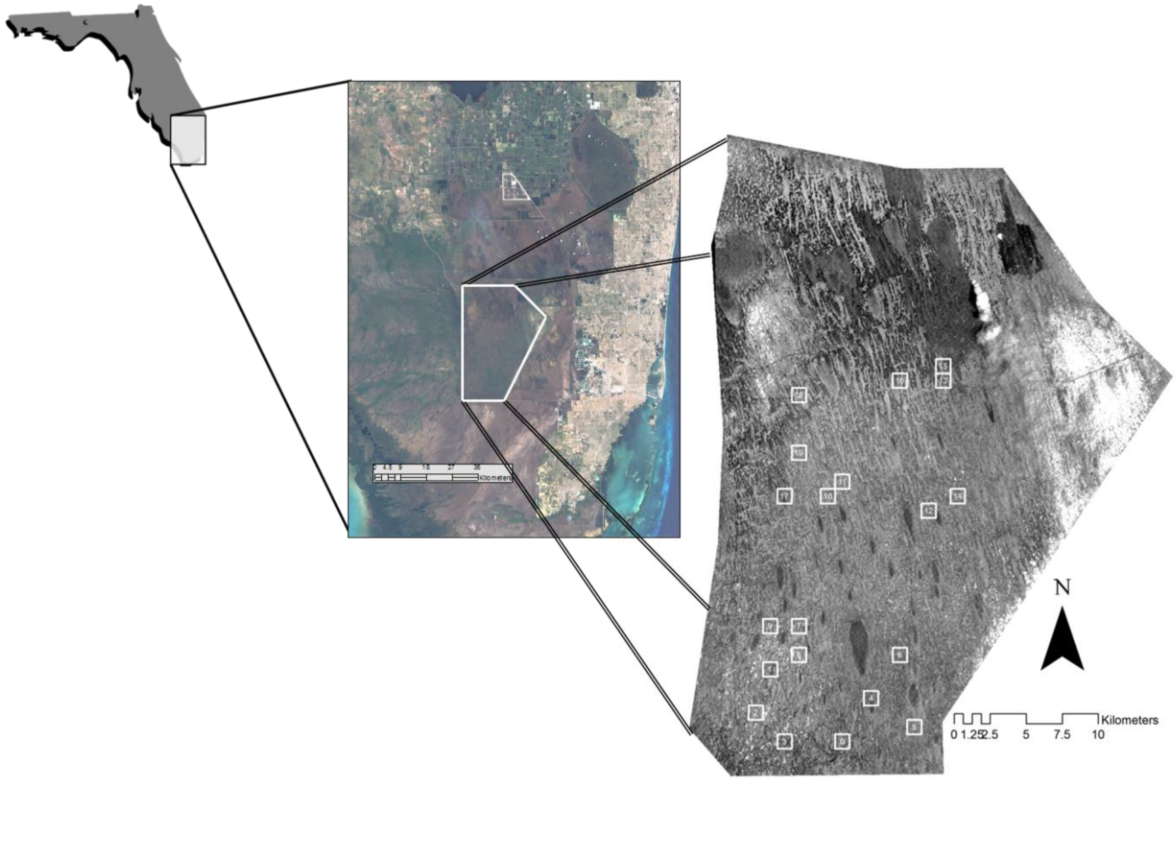

Figure 1: Habitat study plots in Water Conservation Area 3A South, Florida, USA.

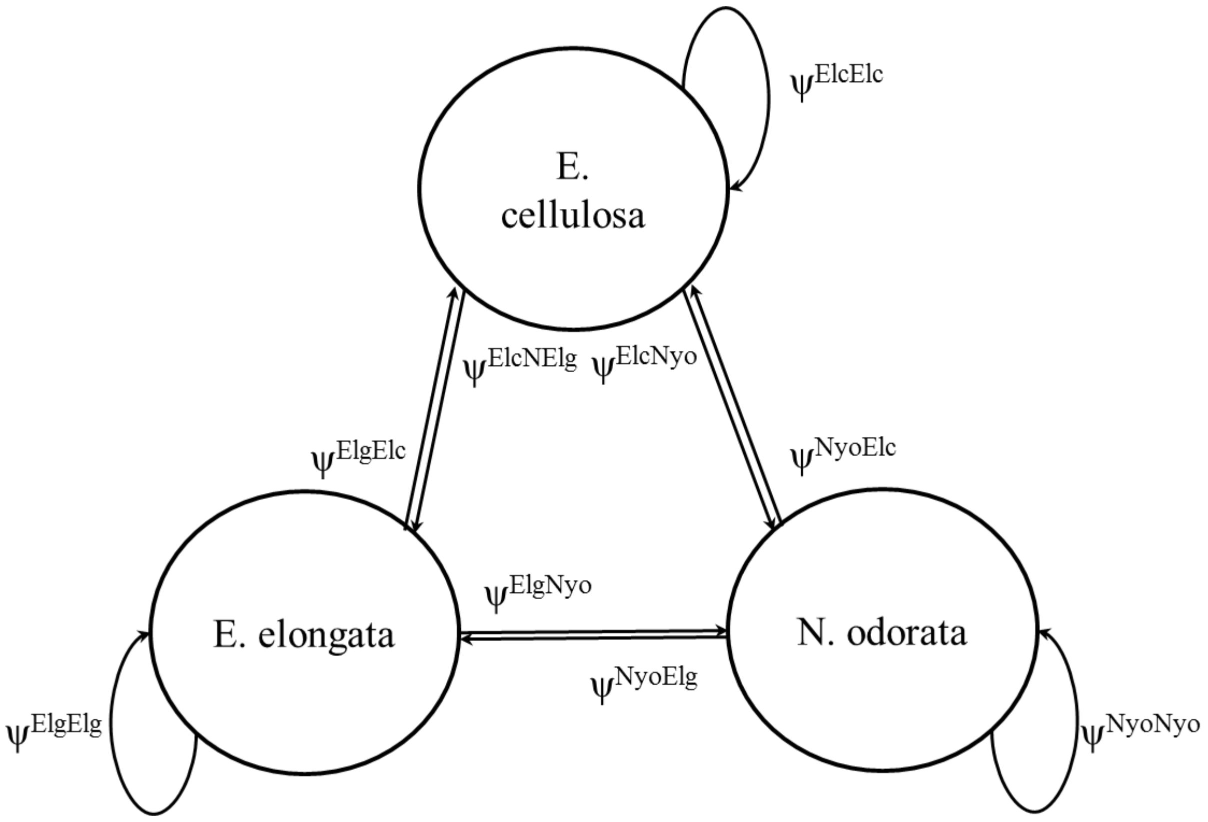

Figure 2: Multistate model for slough/wet prairie habitats in Water Conservation Area 3A South, Florida, USA. Ψ indicates transition probabilities from one state to the next. *Eleocharis elongata* = *E. elongata* = Elg, *Eleocharis cellulosa* = *E. cellulosa* = Elc, *Nymphaea odorata* = *N. odorata* = Nyo.

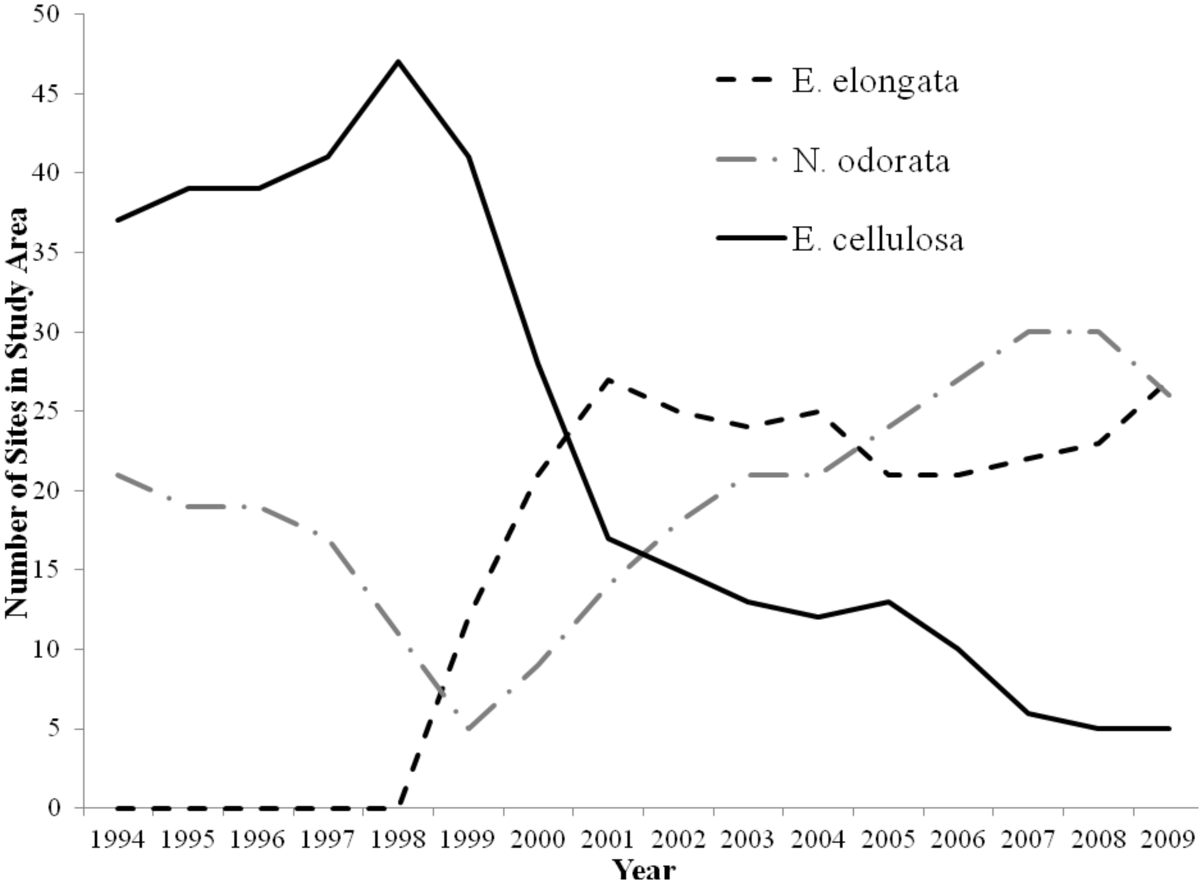

Figure 3: Number of sites in each community state in Water Conservation Area 3A South, Florida, USA from 1994-2009.

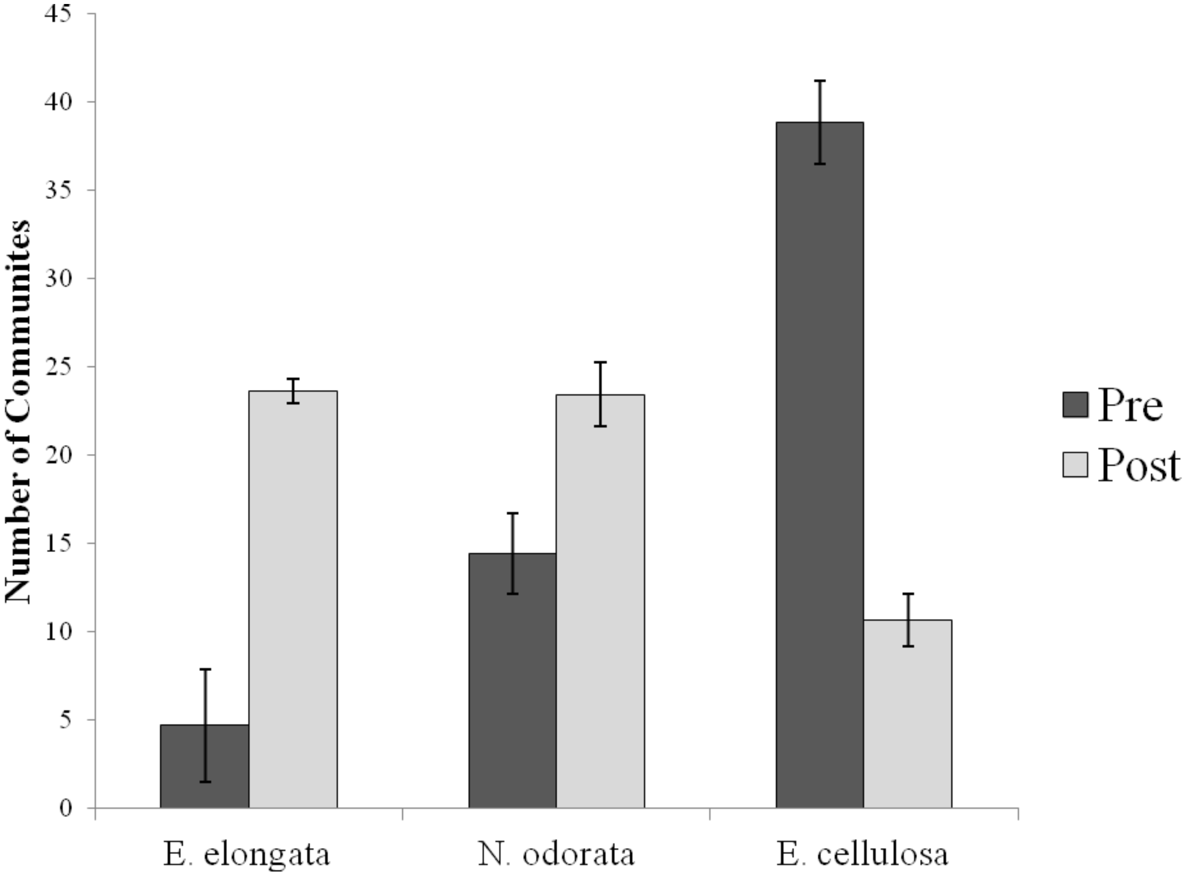

Figure 4: Comparison of number of sites in each community state in Water Conservation Area 3A South, Florida, USA before and after snail kite population decline in 1999-2001.

Appendix E. Habitat Monitoring for the Florida Snail Kite

Introduction

The Florida Snail Kite (Rostrhamus sociabilis) is a wetland-dependant endangered species adapted to a unique and extremely dynamic system, the Everglades. The Snail Kite's range encompasses the entire Everglades watershed, a mosaic of wetland habitat types that are highly impacted by anthropogenic activities (Davis et al 1994). Alterations in water depths, hydroperiods and habitat degradation have short and long-term impacts on Snail Kite demography, principally nest success and juvenile survival (Beissinger and Snyder 2002, Bennetts et al 2002, Kitchens et al 2002). Particularly in this time of Everglades restoration, understanding the effect that environmental processes can have on habitat, what changes will occur with alteration of those processes, and how it affects Snail Kite reproduction potential is essential to a sound conservation strategy (Bennetts et al 1998). This is especially important as the declining Snail Kite population is reaching critical lows (Martin et al 2007).

The Snail Kite is a dietary specialist and its primary prey is the apple snail (*Pomacea paludosa*), whose population levels and availability as prey are also controlled by hydrology and habitat (Darby et al. 2002). Apple snail availability has decreased (Darby, pers. comm.) and is a suspected contributor to Snail Kite decline. However, even with sufficient prey available, habitat structure is critical in enabling Snail Kites to find food resources (Bennets et al 2006). We believe that not only is the rate at which Snail Kites encounter sufficient apple snails important, but just as critical is the rate at which Snail Kites encounter apple snails that are available on emergent vegetation, particularly during the breeding season. Simply studying constraints on the apple snail would not explain changes in Snail Kite demography (Bennetts et al. 2006), but incorporating constraints on availability of foraging habitat, especially in breeding regions, would contribute significantly to the entire conservation perspective.

Water Conservation Area 3A was the largest and most consistently used component of the habitat designated critical to the Snail Kite (Kitchens et al. 2002, Mooij et al. 2002). Its historic contribution to kite reproduction is significant (Kitchens et al. 2002). The current negative population trends of the Snail Kite may reflect the degradation of foraging and nesting habitat quality in Water Conservation Area 3A South (WCA3A) alone (Martin 2007, Martin et al. 2007). Shifts of Snail Kite nesting density up the slight, but significant, elevation gradient

(Zweig et al 2011) in WCA3A have been documented over the past two decades (Bennetts and Kitchens 1997). This is presumably in response to degradation of nesting or foraging habitat as a result of sustained high water levels from impoundment and water management (Kitchens et al 2002). Nesting activity has shifted up the elevation gradient to the west, and has also moved south in response to recent increased drying rates, restricting current nesting to the southwest corner of WCA3A and reproduction in this critical breeding area has waned significantly.

In WCA3A, kites forage mainly in wet prairies and emergent sloughs where their primary prey, apple snails, are most visible and abundant (Bennetts et al. 2006, Karunaratne et al. 2006). Although apple snails are found in varied wetland habitats, abundances tend to be higher in sparse prairies and emergent sloughs and very low in *Nymphaea odorata*-dominated sloughs (Karunaratne et al. 2006). Previous studies in this region (Wood and Tanner 1990, David 1996) indirectly documented the conversion of wet prairies to aquatic sloughs, which constitutes a loss of quality Snail Kite foraging habitat (Kitchens et al. 2002). None of these studies were designed to provide inference beyond the isolated sites in which they were conducted, and unfortunately occurred largely outside kite foraging and nesting areas. There is concern that conversion of wet prairie/emergent slough habitats to deeper, less desirable sloughs will lower kite reproduction, primarily through lower prey base availability in those communities (Karunaratne et al 2006).

To address the issue of habitat degradation within breeding areas and its effect on snail kite reproduction success, a vegetation study was initiated in 2002 to monitor critical kite foraging habitat in WCA3A. It is now particularly vital to monitor kite habitat given their critical state and a continuing trend towards higher maximum water levels and a more extreme hydrologic range within WCA3AS. In this study, we hypothesize that vegetation communities are trending towards deeper species due to water management and that it affects the kite's use of WCA3AS.

Methods

To monitor foraging habitat, we used data across the landscape (Fig. 1) of WCA3AS from a continuing vegetation study (Zweig and Kitchens 2008). This has been the primary nesting area since the 1960's and has been consistently used over time. Twenty 1-km2 plots (Fig.1) were placed in a stratified random manner across the landscape gradients in WCA3A South. Plots were stratified by the landscape level gradients of peat depth, water depth and snail kite nesting activity. Five a priori physiognomic types were identified: slough, sawgrass, tree/shrub island, cattail, and wet prairie. Two or three transects in each plot were placed perpendicular to ecotones, beginning in one a priori type and terminating in another, e.g., slough to sawgrass. We collected 0.25 m^2 samples of all standing biomass along a belt transect, clipping the vegetation at peat level at 3 m intervals, and included any submerged aquatic plants within the sample. Shrubs were sampled in the same manner as the herbaceous vegetation; there were no trees in transects. Samples were collected from every transect in every plot during the dry (May/June) and wet season (November/December) of each year from 2002-2006 and only in the wet season from 2007-2011. These were sorted by species, counted, dried to a constant weight, and weighed to the nearest 0.1 g. Approximately 14,000 samples were collected and processed between 2002 and 2011. We used only the wet season data as there were fewer issues of sampling error due to small, new growth and matted prairie vegetation than in the dry season. We used only the a priori slough and prairie samples to focus on kite foraging habitat and its changes.

Multivariate Analysis

To account for high densities of low biomass species and high biomass of low density species, the data were relativized in an index, importance value (IV), calculated by:

IV for species i = ((Rdi + Rbi)/2)*100,

where Rdi is the relative density of species i and Rbi is the relative biomass of species i. Relative measures are the sum of biomass or density of species i divided by the sum of biomass or density of all species within each sample.

We combined the a priori physiognomic 0.25 m^2 samples in each plot into one point (n = 417) and performed a hierarchical cluster analysis on the IV data with a Sorenson distance measure and flexible beta of -0.25 in PC-ORD (McCune and Mefford 1999). To choose how many clusters were present during the study period, we ran an indicator species analysis (ISA) (Dufrêne and Legendre 1997) to prune the cluster dendrogram.

Univariate Analysis

Count data per 0.25 m^2 sample for only prairie/sloughs, by species, were modeled with a negative binomial generalized linear model in R (glm.nb in MASS): Density ~ 1-Year + Community (Prairie or Slough). Only emergent species (*P. geminatum, P. hemitomon, E. cellulosa, and E. elongata)* and *N. odorata* were modeled as they are important habitat species or indicators of habitat change.

Results

The cluster/ISA indicated 11 communities within the study area and they were named by the indicator species for that community: *Paspalidum geminatum/Panicum hemitomon*, Bacopa transitional, *Nymphaea odorata*, Sparse sawgrass prairie, *Cladium jamaicense*, *Eleocharis elongata*, *Blechnum serrulatum/Osmunda regalis*, Peltandra virginica, *Utricularia* spp, *Pontideria cordata*, and *Eleocharis cellulosa*.

Univariate Analysis

All modeled species densities were significantly affected by year (2002-2011) and community (prairie or slough) ($p < 0.05$). Stem density of all emergents decreased significantly over time (Figs. 5-8). Stem density of N. odorata increased significantly over time (Fig. 9).

Discussion

Previous multivariate analyses have suggested 10 communities (Zweig and Kitchens 2008), but increased amounts of data allow for more communities to be delineated. The 'new' community is a sparse sawgrass prairie that has no spatial pattern and occurs across the landscape. The rest of the 10 communities from the cluster/ISA analysis correspond to previous communities from Zweig and Kitchens (2008). The sparse sawgrass community occurred in every year, except 2007, which was a year of decreased sampling effort.

The univariate results suggest that the foraging habitat within WCA3AS is trending towards deeper communities, with emergent vegetation decreasing and floating aquatic vegetation increasing. The communities are separated by peat and hydrology, but specific hydrologic parameters are not currently available. See Appendix C for details on hydrology and slough communities.

All of the communities described in this study are similar in species composition to the study by Karunaratne et al. (2006). The densities of *E. cellulosa* would be categorized as very sparse Eleocharis, a characteristic of communities with higher apple snail abundance. The densities of *P. hemitomon* would be categorized by Karunaratne et al. (2006) as sparse and also higher apple snail abundance. The species are sparse and becoming more sparse over time.

Stem density of emergents in the sloughs within the critical kite breeding habitat is decreasing and density of the floating aquatic *N. odorata*—a less desirable kite foraging habitat—is increasing. Both the multivariate and univariate analyses indicate a decline in foraging habitat for the snail kite. Without the proper vegetative habitat structure, even sufficient apple snail density can not sustain kite populations. Prey availability is critical (Bennets et al 2006).

WCA3A has been the most critical habitat unit within the Snail Kites' range, providing both the largest extent of quality nesting and foraging habitats and the highest juvenile production (Kitchens et al 2006). Given the importance of WCA3A within the Snail Kites' habitat network (Kitchens et al. 2002, Martin 2007), the vegetation community transformations documented in this study are particularly pertinent and may help explain why WCA3A appears to be offline for reproduction and recruitment. Six out of seven transects in the study transitioned or remained in a deeper, less desirable Snail Kite foraging habitat, while one remained as Paspalidium. Many transects made abrupt changes in community composition in 2005 due to hurricane Wilma, but returned to more normal community compositions in 2006. We have demonstrated that, even in a relatively short period of four years, wet prairie/emergent sloughs are converting to deeper, less desirable Snail Kite habitats in response to hydrologic factors, with a strong temporal trend.

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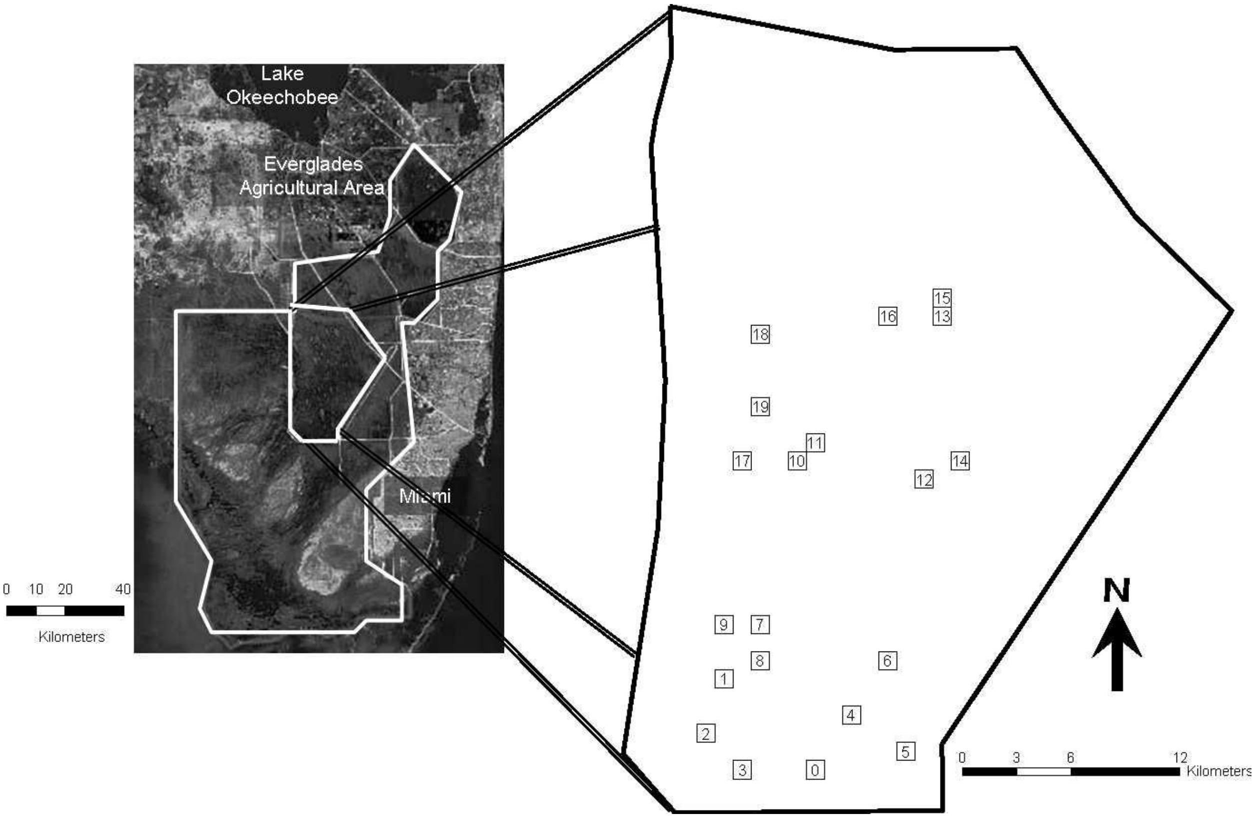

Figure 1: Study plots in Water Conservation Area 3A South. This analysis concentrated on plots 7, 8, and 9, but data is being collected landscape-wide.

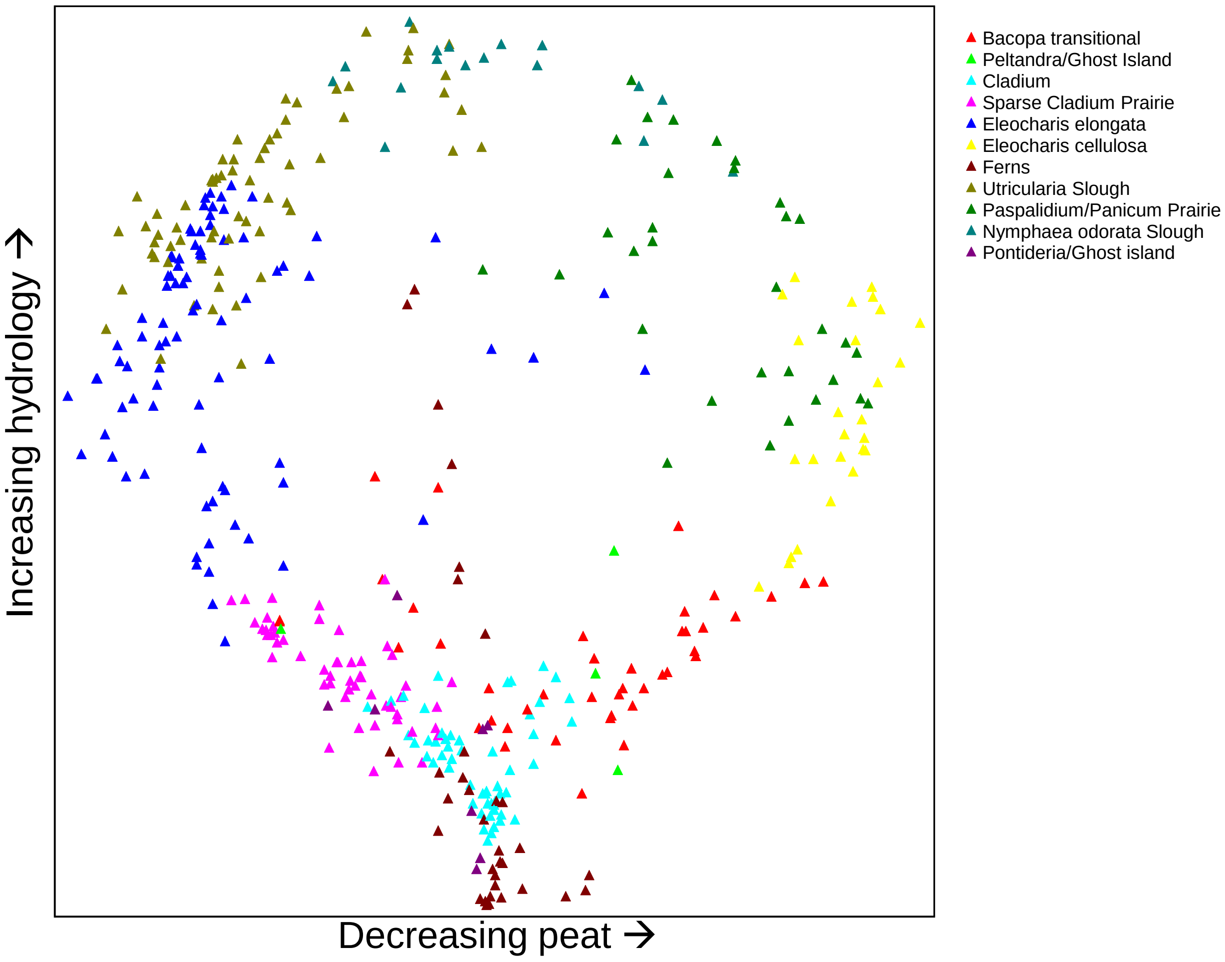

Figure 2: Non-metric multidimensional scaling ordination with vegetation data in Water Conservation Area 3A South. This graph demonstrates how vegetation communities are influenced by hydrologic and peat variables.

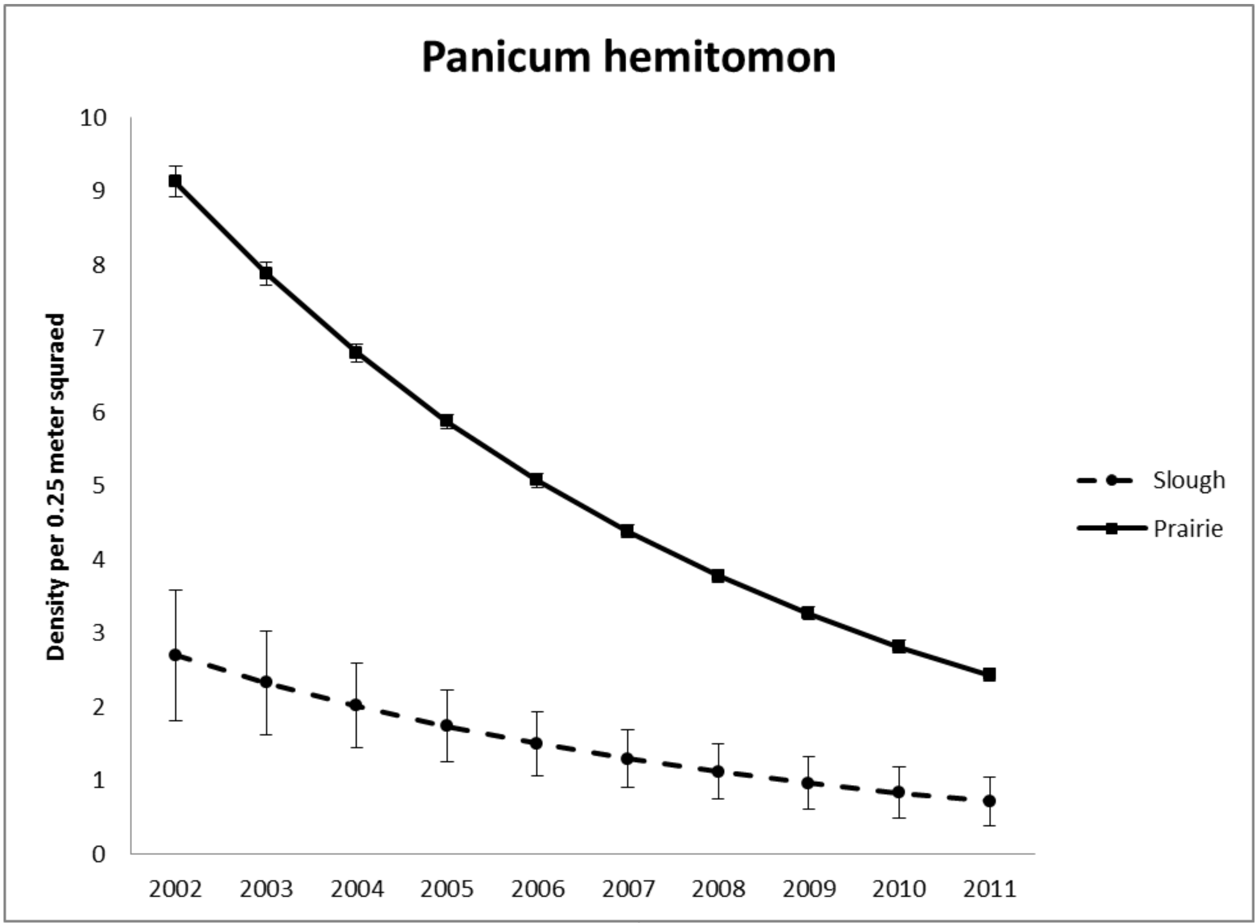

Figure 5: Modeled density estimates per 0.25-m^2, with standard errors, of *Panicum hemitomon* within our study area in Water Conservation Area 3A South.

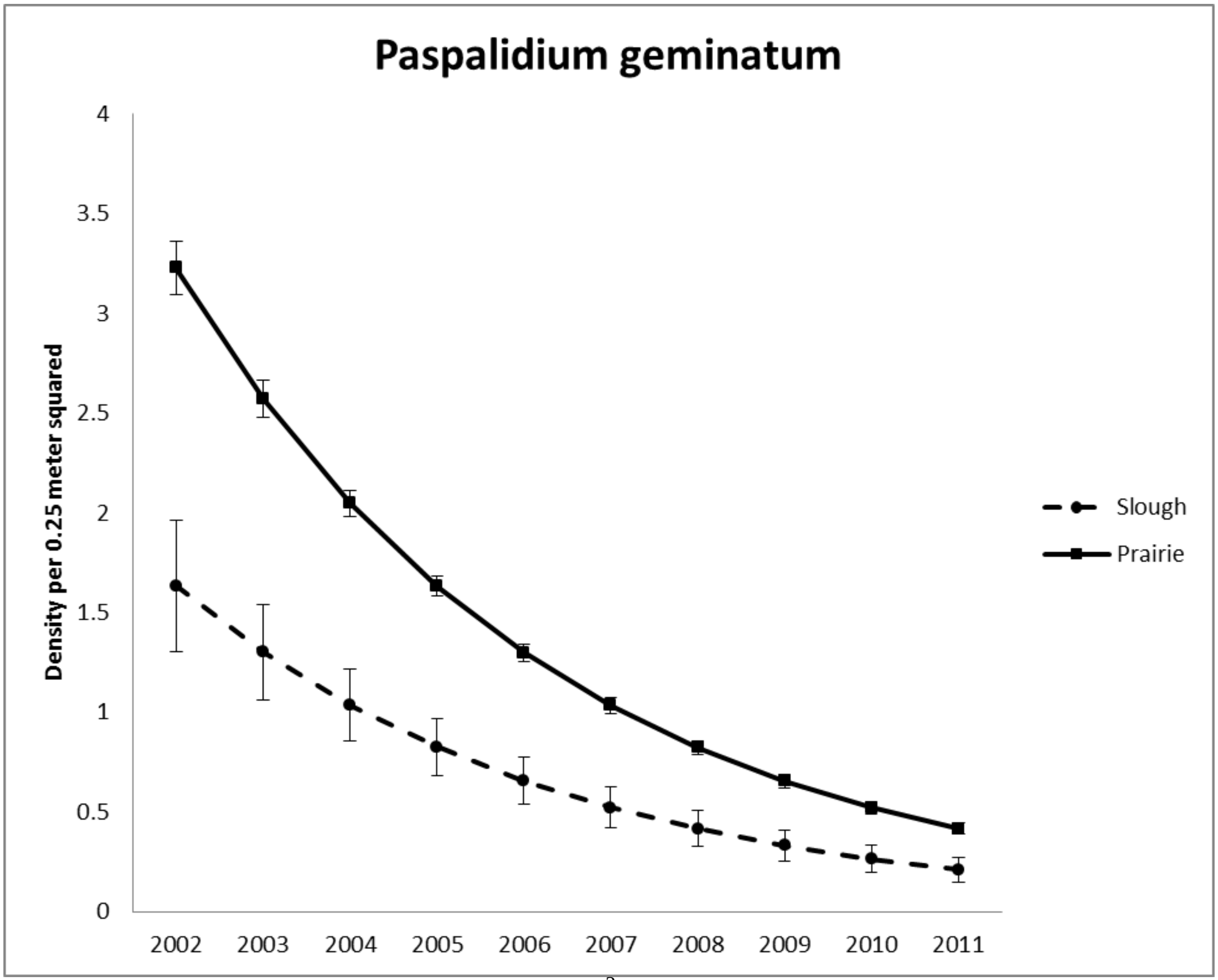

Figure 6: Modeled density estimates per 0.25-m^2, with standard errors, of *Paspalidium geminatum* within our study area in Water Conservation Area 3A South.

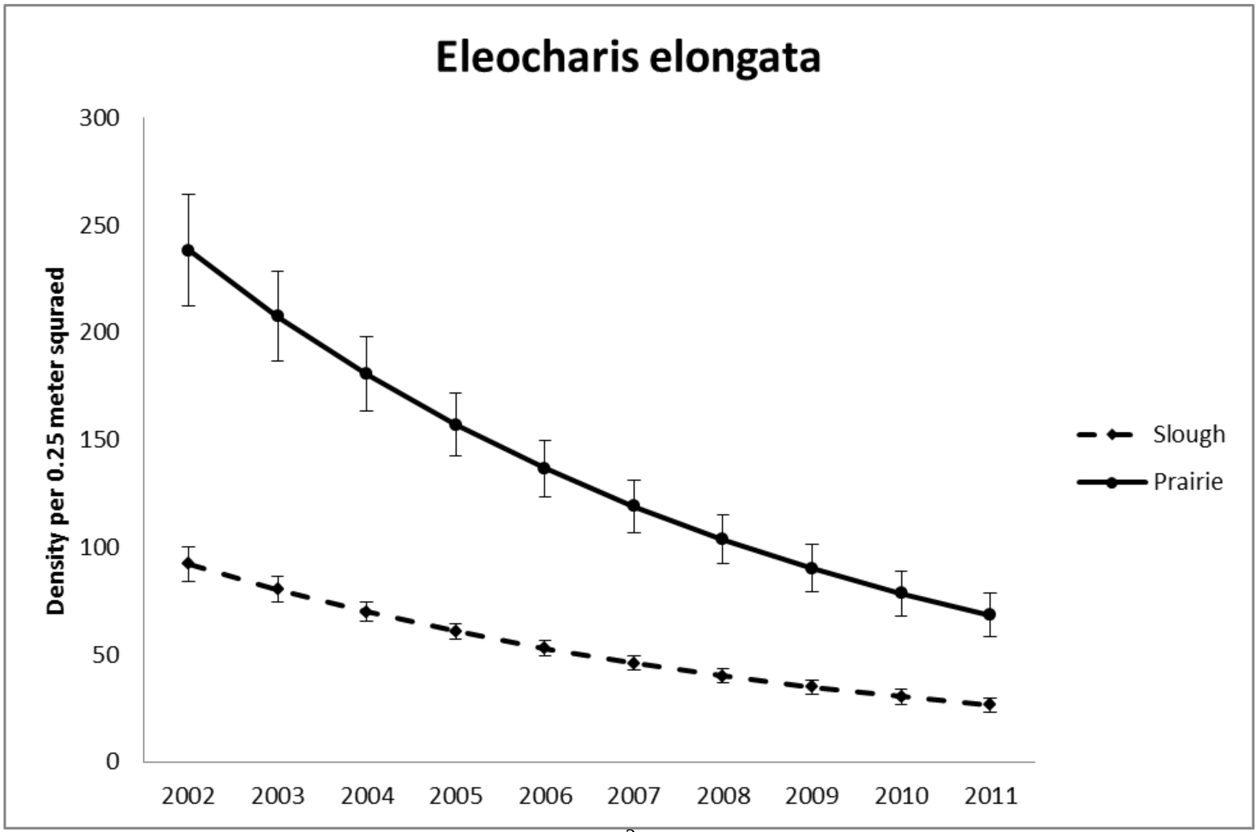

Figure 7: Modeled density estimates per 0.25-m^2, with standard errors, of *Eleocharis elongata* within our study area in Water Conservation Area 3A South.

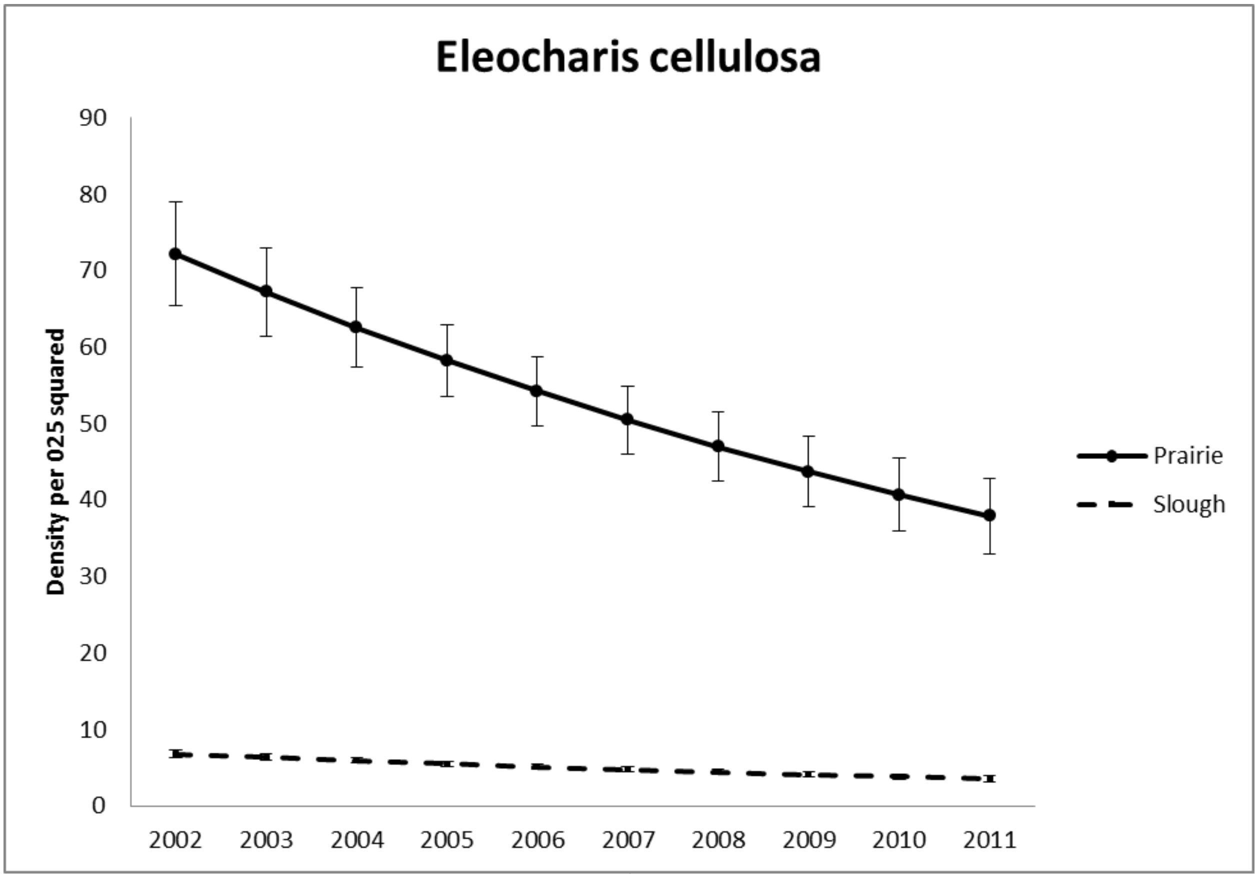

Figure 8: Modeled density estimates per 0.25-m^2, with standard errors, of *Eleocharis cellulosa* within our study area in Water Conservation Area 3A South.

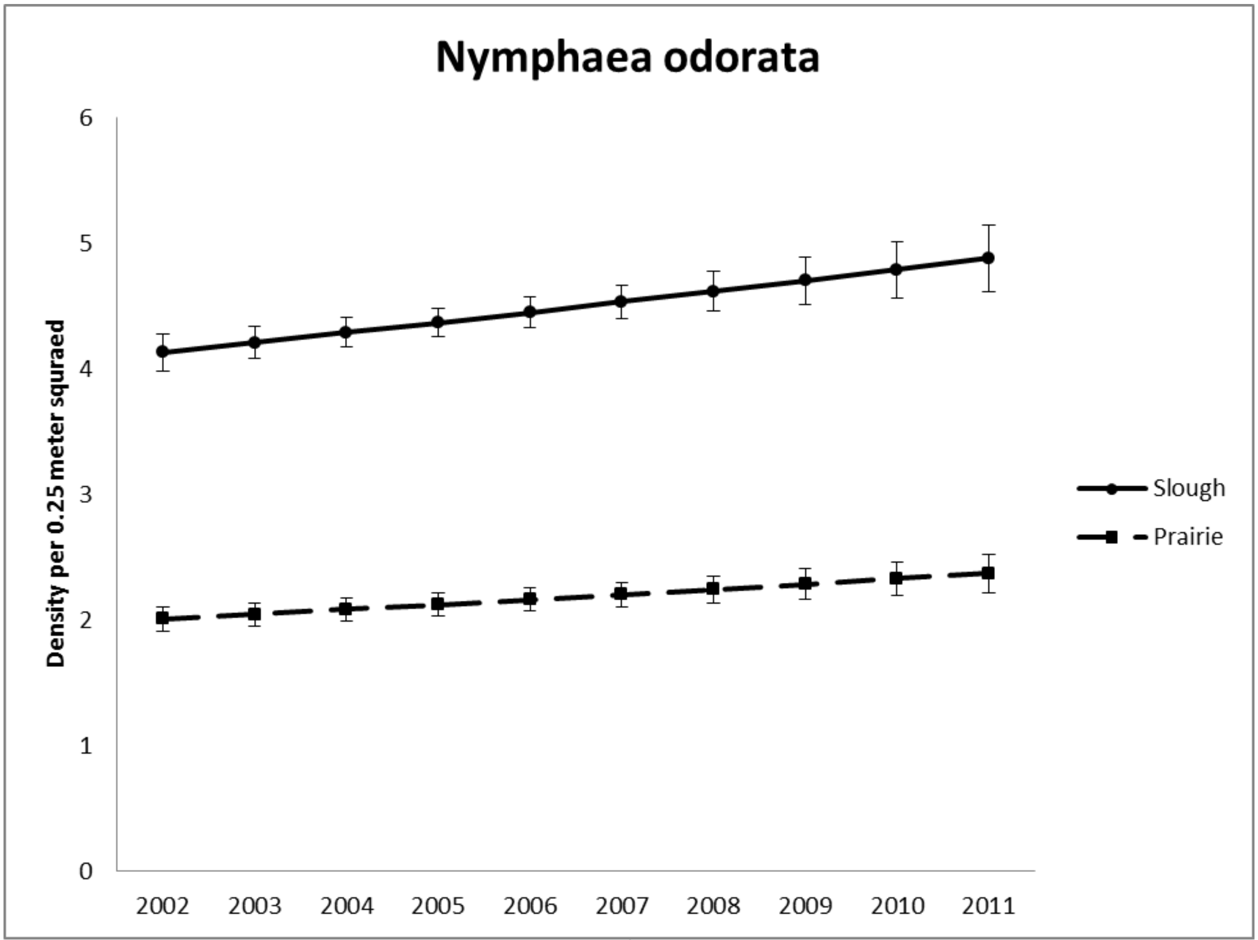

Figure 9: Modeled density estimates per 0.25-m^2, with standard errors, of *Nymphaea odorata* within our study area in Water Conservation Area 3A South.