

SNAIL KITE DEMOGRAPHY
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By
Robert Fletcher, Christopher Cattau, Rebecca Wilcox, Christa Zweig, Brian Jeffery, Ellen
Robertson, Brian Reichert, and Wiley Kitchens

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U. S. Geological Survey
Florida Cooperative Fish and Wildlife Research Unit
Department of Wildlife Ecology and Conservation
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 explanation 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. We emphasize 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 expressed prior written consent of the authors.

ABSTRACT

This report provides an annual update regarding ongoing population monitoring of the endangered snail kite (*Rostrhamus sociabilis plumbeus*). The over-arching goals of this monitoring are to provide reliable information on population size and trends, as well as key demographic, habitat, and foraging information of relevance to the recovery of this species. This report concentrates on demographic data collected during 2013, but also incorporates data collected since 1992. Recent demographic results reveal that snail kite abundance drastically declined between 1999-2008, with the population approximately halving from 2000 to 2002 and again from 2006 to 2008. Each of these two periods of population decline coincided, in part, with drought conditions throughout the southern portion of the kites' range. The 2001 drought significantly, yet temporarily, affected adult apparent 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. By coupling the vital rates measured over this time period with the changes in population size (using a Life Table Response Experiment), it became apparent that the primary demographic factors contributing to this decline were changes in adult fertility (which is the product of young fledged per adult and 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 were concerning, as they indicated an increased risk of extinction when compared to results from a previous PVA conducted in 2006. Recent analyses also provide indications of an aging population with problems inherent to older individuals, including increased adult mortality rates and decreased probabilities of attempting to breed, both of which have been shown to be exacerbated during times of harsh environmental conditions. While from 1999-2008 population size declined, from 2009-2012 there were modest, but consistent positive population growth rates.

Multiple factors may be limiting kite reproductive ability and reducing the carrying capacity of several of the wetland units throughout the state, although most are likely tied to both short-term natural disturbances (e.g., drought; Beissinger and Takekawa 1983) and long-term habitat degradations (e.g., the conversion of wet prairies to sloughs in WCA3A; Zweig and Kitchens 2008). These issues may alter both prey density and habitat conditions for foraging and successful reproduction. In general, during the period of observed declines, there was a notable

decline in snail kite production from two critical snail kite habitats, WCA3A and Lake Okeechobee. No young were fledged in WCA3A in 2001, 2005, 2007, 2008, or 2010. In 2012, only one successful nest, which fledged one young, was observed in WCA3A. In 2013, fecundity was increased in WCA3A as it contributed 13% of the overall observed nesting effort and 7% of the fledgling production. The decline in breeding activity and nest productivity observed in WCA3A over recent years may reflect deteriorating habitat quality, although data from 2013 are promising. Conditions suitable to snail kite reproduction in Okeechobee, on the other hand, may have improved over the past few years. In 2010, nesting was observed on Okeechobee for the first time since 2006. Then in both 2011 and 2012, Okeechobee was the third most productive wetland (in terms of kite reproduction) range-wide. In 2013, Okeechobee was the most productive wetland, in terms of overall kite production, with 24% of observed fledglings across all sites monitored arising from Lake Okeechobee.

The relatively low reproductive output from Okeechobee (1997-2010) and WCA3A (2001-present) left the kite population heavily concentrated in and dependent upon the Kissimmee River Valley in recent years, 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. In 2012, Toho accounted for 25% and 24% of all successful nests and fledged young, respectively. In 2013, Toho was the second largest contributor to observed nesting effort and was third in contributing to the number of observed fledglings. Two other notable changes occurred in 2013: we found a greater proportion of nests in Lake Kissimmee than in previous years, and we found a large number of nests in two Stormwater Treatment Areas (STA 1 and 5). For 2013, Lake Kissimmee ranked fourth in the number of observed fledglings and the STAs ranked third.

For 2013, we estimated the population size to be 1198 (with a 95% confidence interval of 1086-1309). This estimate suggests that population size did not substantially change from 2012 to 2013, although this estimate is higher than estimates from 2011 and 2010 (approximately 925 individuals were estimated in 2011 and 826 individuals in 2010). These estimates, along with the increased number of fledglings counted during the 2011, 2012, and 2013 breeding seasons are encouraging trends, yet it remains unclear whether such trends signify the beginning of a recovery phase. In this report we detail new findings related to snail kite demography and foraging, as well as a new assessment regarding the reliability of the monitoring effort. We also make specific recommendations that may help guide management decisions aimed at increasing

kite population growth rate.

INTRODUCTION

Understanding population biology is essential for management and conservation of species of interest, as well as interpreting effects of changing environments on the distribution and abundance of species (Williams et al. 2002). For the past 30 years, scientists, managers and conservationists have written about the need for long-term population monitoring, particularly in the context of endangered species conservation and understanding effects of habitat management and restoration (see, e.g., Lindenmayer and Likens 2009). Because changes in population size of species are directly linked to changes in demographic rates (Pulliam 1988), such as reproduction and survival, knowledge of how these rates change over time and what drives variation in these rates are necessary for effective conservation and management of imperiled species.

The snail kite (*Rostrhamus sociabilis plumbeus*) is a highly endangered, wetland-dependent raptor that has experienced a population decline over the past 15 years (Martin et al. 2007a). Snail kites display an extreme form of dietary specialization, feeding almost exclusively on a single genus of freshwater apple snail (*Pomacea spp.*) (Howell 1932; Stieglitz and Thompson 1967). Historically, snail kites in Florida foraged primarily on the native *Pomacea paludosa*, which is the only species of this genus native to Florida (Rawlings et al., 2007), but in recent years kites are commonly foraging on an invasive exotic snail, *Pomacea maculata* (Darby et al. 2007, Cattau et al. 2008). 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 and Kitchens 1997a). Three features that remain consistent 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 persistence 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 km² (2480 mi²) 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 (Figure 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 and Kitchens 1997a; 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 and/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 1997a; 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.

For snail kites, the U.S. Fish and Wildlife Service have identified 5 recovery criteria, four of which emphasize the need to monitor population size and demographic trends. *Criterion 1*: the 10-year average for total population size must be greater than 650 (with a coefficient of variation < 20%). *Criterion 2*: there cannot be an annual estimate of population size less than 500. *Criterion 3*: the rate of increase (i.e., population growth rate) must be greater than or equal to 1, sustained as a 3-year running average over 10 years. *Criterion 4*: the feeding range will not decrease from the current extent. *Criterion 5*: nesting must occur regularly over the 10-year period in Saint John's Marsh, the Kissimmee Chain of Lakes, Lake Okeechobee, and at least one Water Conservation Area (WCA). The first two criteria require reliable estimates of population size each year. We also note that for Criteria 1-2, these numbers were derived from annual

counts conducted from 1969-1992, which did not correct for imperfect detection. Since 1992, we have estimated detectability of kites to range from 0.25-0.63 (see below), which would alter a count of 650 to be between 1040-2555 birds, and a count of 500 to be between 800-1965 birds. Criterion 4 focuses on the need to acquire relevant information on foraging, whereas criterion 5 requires information on reproduction of kites. Over the years, this monitoring program has attempted to provide relevant information for assessing each of these criteria.

Our primary goal of this work is to provide reliable estimates of population trends and demography of kites, including identifying key demographic rates influencing trends and how natural and managed conditions alter these rates. This report presents demographic data on the snail kite in Florida. It concentrates on data collected in 2013, but also synthesizes data collected since 1992.

METHODS

Study Area

The Florida population of snail kites is best viewed as a 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, Meyer and Kent 2011). 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 (Figure 1).

Monitoring Protocol

Survey method for monitoring population size

Since 1969, snail kites have been monitored in Florida. After nearly 25 years of monitoring, it became apparent that monitoring approaches were insufficient for assessing population size and trends of kites (Bennetts et al. 1999). Counts were not comparable over time because of different amounts of effort across years (i.e., a lack of standardized sampling), observer effects (some observers tended to see more individuals than others), and other related issues of detectability of snail kites.

A second challenge regarding snail kite monitoring is that we know that individual kites

can move great distances throughout the year (Bennetts & Kitchens 1997a, 1997b, 2000; Martin et al. 2006, Meyer and Kent 2011), but particularly when they are not breeding (because when kites are associated with nests, they must return to the nest site frequently, which will likely constrain movements of breeding birds compared to those that are not breeding). Consequently, for estimating population size in snail kites, there are several attributes of a monitoring and estimation framework that are needed: 1) an approach that can use both information on marked birds and unmarked birds to estimate population size; and 2) an approach that acknowledges that kites can move in and out of wetlands within the sampling period.

The first component is useful because marked birds can provide reliable measures of detectability of individuals, movements into and out of sampled sites, and mortality of individuals. Unmarked birds should not be ignored when estimating population size because these individuals provide highly relevant information for interpreting population size. Simply put, counting all birds is incredibly important for monitoring, but we know it is not sufficient if we do not account for known biases. By integrating information on a portion of the population that is marked, we can provide more reliable estimates of population size.

The second component is critical to address because considerable movement can be problematic for interpreting counts. For example, individuals counted during one survey might not be the same individuals as those counted during another survey, due to movements of individuals in and out of a study site. Second, individuals may not temporarily occur in sites that are surveyed (termed ‘temporary emigration’ in population ecology models; Schaub et al. 2004), because they have moved to other areas not being surveyed.

To address these issues, Dreitz et al (2002) recommended an approach for estimating population size and growth rate in snail kites, which has been implemented since 1997. This approach uses multiple surveys during the breeding season where crews count all individuals observed (both marked and unmarked birds), coupled with banding and re-sighting information of individually marked snail kites, to estimate population size. This approach explicitly addresses the problem of imperfect detection of kites, movements of individuals into and out of the sampled sites, and potential mortality of individuals during the breeding season.

Multiple consecutive surveys were conducted by highly trained field technicians throughout the designated wetland units (Figure 1) from March 1st to June 30th at 2-3 week intervals of each year since 1997. This time period coincides with the occurrence of peak nesting

(Bennetts & Kitchens 1997a). Monitoring during the breeding season for birds is generally preferable to other times of year because most adults are nesting, which restricts movement and allows for population estimates to more closely reflect the breeding population than surveys conducted during non-breeding time periods. The surveys followed a format similar to the quasi-systematic transects conducted by airboat for former 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). This information is then used to estimate population size by adjusting counts for imperfect detection, potential mortality, and movements of birds into and out of study sites (see below).

Nest monitoring

Nest monitoring of kites is essential for three reasons. First, it provides critical information for Criterion 5 of the USFWS recovery plan for kites (see Introduction). Second, reproduction is directly linked to population change in snail kites and is a demographic rate that is amenable to site-specific management. Third, by banding fledglings at nests, it provides a low-cost approach for marking snail kites (an important element for reliable population size estimation; Williams et al. 2002). Focusing banding on juveniles has the added benefits of knowing the exact age of individually banded kites and obtaining a large sample size of marked juveniles, which show lower, and more variable, survival rates than adults (see Results). Understanding effects of age structure is important for population dynamics, and has been shown to be highly relevant to interpretation of reproduction (Reichert et al. 2012) and survival (Reichert et al. 2010b) of snail kites.

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) in an effort to monitor all known nests of kites. 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 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). We attempted to band all individuals that fledged from nests we monitored. At approximately 24 days of age, snail kite nestlings were banded with unique alpha-numeric bands. Each kite receives a unique band which can later be resighted-providing information on an individual's fate and location through capture-mark-resight analyses.

Data Reported and Statistical Analysis

Number of nests, nest success and nest productivity

We provide several different measures of reproduction that are of relevance to the production of snail kite offspring. We defined *active nests* as those that progress beyond the building stage to at least the egg-laying stage (Bennetts and Kitchens 1997a), and we only report counts of and perform analyses on active nests. Hereafter, we use the terms “nests” and “active nests” interchangeably. We make this distinction because some prior work on snail kite nesting biology tended to include nests that did not have eggs or nestlings in summaries (e.g., Snyder et al. 1989; for a discussion regarding the semantics of snail kite nesting biology see Chapter 5 in Bennetts & Kitchens 1997a); however, because kites can build nests without ever using them for reproduction, we do not include this information into our summaries. 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 to be consistent with prior work on snail kites: $\hat{S} = x/n$, 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 active nests with known fates (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 for which fate could not be determined were not included in our analyses of nest success or productivity. We also calculated *daily nest survival rates*, DSR, which is similar to nest success but accounts for variation in ‘exposure’ or the time we observed nests. DSR is generally considered to be a less biased estimator of nest success than apparent nest success. To do so, we used the logistic-exposure approach described by Rotella et al. (2004). Finally, we summarize nesting by calculating the *relative contribution of wetlands to the overall range-wide nesting effort*, on the basis of the number of nests found in a wetland divided by the total number of nests we monitored across all sites, as well as the *relative contribution of wetlands to the range-wide fledgling production*. We note that detectability of nests likely varies to some extent among sites (we have collected information on nest detectability and are currently analyzing it), but given preliminary analyses and the long-duration

of the nesting cycle for kites, the likelihood of detecting successful nests that produce fledglings is very high. That is, we may occasionally not find nests that failed because they were not active for long periods of time; however, nests that are successful are active, on average, for 58 days (Rodgers et al. 1995), such that we have much more time to eventually observe these nests in the field. We will provide these estimates in our next report.

Range-wide apparent annual survival

We used information on individually marked kites and extensions of the Cormack-Jolly-Seber model (CJS) (Cormack 1964; Jolly 1965), implemented in program MARK 7.1 (White & Burnham 1999), to estimate apparent annual survival (Φ ; hereafter survival) and detection probabilities (p ; i.e., the probability of detecting a Snail Kite given that it is present in the study area during the period of sampling, or ‘imperfect detection’ of individuals). CJS models have a long history in population ecology and the principles of this approach are at the foundation of much of population analysis (Williams et al. 2002). Survival is considered ‘apparent’ because this approach does not distinguish between mortality and permanent emigration from our study sites (Fig. 3). So, if an individually banded kite leaves all of the sites we survey and never returns over the years, this approach would lead to lower estimates of survival than true survival rates. However, given the extent of our sampling across the geographic range (Figure 1), the annual time frame of this analysis (i.e., we pool all information for the entire sampling period within a year), and the frequent movements of snail kites (Bennetts & Kitchens 1997a, 1997b; Martin et al. 2006; Meyer and Kent 2011), we expect this to be rare across years.

We developed a set of biologically relevant models that allowed apparent survival to vary temporally (according to ecological/environmental effects; see below) and to differ between age classes (Juveniles: 30 days – 1 year; Adults: >1 year). We treated imperfect detection as a nuisance variable and incorporated time-dependent detection probabilities in all models (i.e., separate parameters were assigned to detection for each year). Hypothesized models were ranked using Akaike’s Information Criterion adjusted for small sample sizes (AICc) (Burnham and Anderson 1998). AICc compares models based on the relative fit of the model to the field data and penalizes this fit based on model complexity (i.e., a more complex model should provide much better fit for its use in comparison to a simpler model). Based on the fits of these models, we interpret specific effects based on parameter estimates and their associated uncertainty (e.g.,

if a parameter has a 95% confidence interval, CI, that overlaps zero, then we interpret this effect to be so weak that it is essentially not meaningful). Background information on this approach has been published elsewhere (for details see Bennetts & Kitchens, 1997a; Martin et al., 2006, 2007a, 2007b; Reichert et al. 2010b).

Historically, range-wide adult survival has shown little year to year variation except during severe drought conditions; however, state-dependent adult survival can differ among regions, and has been shown to be lower in the KCOL than in the Everglades and Okeechobee (Martin et al. 2006, 2007b). To assess the effects of regional hydrologic conditions in South Florida on range-wide adult survival, we used the 4-month (May–August) average Palmer Hydrological Drought Index (PHDI) for Florida Climate Division 5. PHDI is a long-term, lagging indicator for drought and wet conditions, reflecting groundwater conditions and reservoir levels (<http://www.ncdc.noaa.gov/oa/climate/research/prelim/drought/palmer.html>, *see* Figure 2). This index was developed by the National Oceanic and Atmospheric Administration (NOAA) and has been used extensively for assessing dry conditions that occur from variation in precipitation and reservoir/groundwater conditions. We focus on Division 5, which incorporated much of the geographic region of kites, but we have also correlated these PHDI measurements with Division 4 and have found that these two regions are highly correlated ($r = 0.83$). May–August corresponds to primarily the post-fledging “Summer” season (*sensu* Bennetts and Kitchens 1997a). This period encompasses the transition from the dry to the wet season (annual low water levels in South Florida wetlands are typically reached in May/June). A low average PHDI value for May–August may reflect a drier than average dry season and/or a drier than average (or delayed) onset of the wet season. Such conditions can lead to low snail density and availability (Darby et al. 2008), particularly in the palustrine systems typical of South Florida, which may influence Snail Kite survival (Bennetts & Kitchens 1997a, 2000; Kitchens et al. 2002; Martin et al. 2006, 2007a, 2008). We used the average (May–August) PHDI to categorize severe (PHDI: -2.00 to -2.99) and moderate (PHDI -1.00 to -1.99) drought years (but also considered PHDI as a continuous measure as well; *see* below). According to the Weather Bureau/NOAA's categorization, the PHDI ranges from -2.00 to -2.99 and from -1.00 to -1.99 are technically labeled as “moderate” and “mild” drought years, respectively (Palmer 1965), which correspond to years of “severe” and “moderate” droughts described previously for kites (e.g., Reichert et al. 2012); for consistency with prior work on snail kites, we keep the term “severe”

and “moderate”. Severe droughts occurred in 2001, 2007, 2011 and 2012, while moderate droughts occurred in 2000 and 2009 (Figure 2). Other authors have drawn similar conclusions when classifying drought years in South Florida (e.g., Abtew et al. 2002, 2010, 2013; Martin et al. 2006, 2007a; Reichert et al. 2012). We considered models that included categorical drought effects (i.e., survival was allowed to differ between drought and non-drought years, as well as between moderate and severe drought years). We also considered models that included PHDI as a continuous covariate. Because kites have been heavily concentrated in the KCOL during the breeding season since 2005 but were concentrated in south of KCOL from 1992-2004, we included models with categorical era effects (i.e., a pre-range-shift era, Era 1: 1992-2004; and a post-range-shift era, Era 2: 2005-2013). Moreover, as drought has little impact on adult survival in the KCOL (Martin et al. 2006, 2007b), we considered models in which drought effects differed between eras. We considered models with time-dependent juvenile survival because (unlike adult survival) juvenile survival shows significant annual variation and previous analyses have found strong support for a fully time-dependent parameterization of juvenile survival (Martin et al., 2006, 2007a, b). Nonetheless, we also assessed models in which juvenile survival was constrained to similar drought and era effects as those of adults, and we considered both additive and interactive effects of age with these parameters. We used an information theoretic approach of model averaging across the most parsimonious models to produce model averaged estimates (Lebreton et al. 1992, Anderson 2008) of range-wide detection and apparent survival of juveniles and adults. Model averaging incorporates uncertainty in model selection and helps guard against model selection bias.

To assess goodness of fit of these models, we estimated the median \hat{c} (Cooch and White 2008) using the most general model in our set (i.e., the model in which a separate parameter for each year was assigned to juvenile, to adult survival, and to detection; Model 8, Table 1). The approach yielded a $\hat{c} > 1$. Therefore, parameter variance components were adjusted to account for over-dispersion, and we used quasi-AIC (QAICc) for model selection instead of AICc, and we adjusted standard errors and confidence intervals accordingly.

Population size

Victoria Dreitz and colleagues developed a ‘superpopulation’ estimator for snail kites that acknowledges the issues described above regarding imperfect detection of kites, frequent

movements of kites among wetlands, and potential mortality (Dreitz et al. 2002; see also Schwarz and Arnason 1996, Schwarz and Stobo 1997 for a general assessment of similar estimators). This estimator uses information on both marked and unmarked birds to count the population, while adjusting for imperfect detection, possible mortality of individuals, and movement of birds into and out of the surveyed sites. It assumes an ‘open’ population (i.e., individuals may move in or out of the sampled area, and mortality may occur). It also assumes that information gleaned from banded (marked) birds applies to unmarked birds. This assumption is necessary for inclusion of the rich information regarding counts of unmarked birds into population estimates.

To provide context for this approach, consider the following. With most count-based approaches to estimating population size, with relevant field data we can adjust for imperfect detection as:

$$\hat{N} = \frac{C}{p}$$

Where \hat{N} is the population size (the ‘hat’ above N is to emphasize that this is an estimate of N), C is the count (the number of birds we observe in the field), and p is the probability of detecting an individual if it is in fact there, which can range from 0-1. Since p is nearly always <1 , N will be greater than C . p is estimated from banded birds by assessing re-sighting of individuals within a breeding season (see below).

The field protocol for counting kites occurs during 6 surveys (approximately 3 weeks apart). So, we could use the above formula to estimate the number of kites during each survey, i ($i = 1, 2, \dots, 6$), as:

$$\hat{N}_i = \frac{C_i}{p_i}$$

We estimate p_i using information from banded individuals using Cormack-Jolly-Seber mark-recapture models that estimate this parameter along with apparent survival, ϕ (see below). In this case, p_i summarizes two components of detection: 1) the likelihood that individuals were detected, given that they were in the area; 2) the likelihood that individuals were available to be detected (which is an issue of temporary emigration). While the approach does not distinguish these two issues, we note that mark-recapture models could be fit to the data that do partition

these issues; we have fit these more complex models and general trends remain the same.

While we can adjust individual counts for imperfect detection, how would we add up these survey estimates to get a measure of total population size? We cannot simply sum these \hat{N}_i because many of the same birds counted in one survey could be the same as those counted in another survey. The superpopulation estimator deals with this issue by first estimating the number of birds in the first survey and then determining how many new birds, B , *moved into* the sample area in subsequent surveys: The superpopulation estimator uses information on both the counts and the marked birds to estimate B for each survey i (surveys 2-6):

$$\hat{B}_i = \hat{N}_i - \hat{N}_{i-1}\phi,$$

or the number of birds observed currently minus the number of birds observed in the previous survey times ϕ . ϕ is commonly referred to as ‘apparent survival’ and is estimated from banded birds using CJS models (described above), although in this case estimates occur across individual surveys within a year, such that ϕ is the probability of an individual surviving and staying (not emigrating) in the study area between surveys (3 weeks). Because over the survey breeding season the probability of kites surviving is likely very high (from annual estimates, we would expect that survival over 3 weeks may be as high as ~0.99), ϕ biologically represents *fidelity* to the study area and $1 - \phi$ can be thought of as primarily estimating the probability that birds *moved out* of the study area.

With this information, the ‘superpopulation’ is estimated as:

$$\hat{N} = \hat{N}_1 + \sum_{i=2}^6 \hat{B}_i.$$

Therefore, the estimated population size (i.e. ‘super-population’) is equal to the abundance of all snail kites that were in the *study area* during at least one survey within the breeding season. For example, even if individuals spent a large proportion of their time outside of all of our survey areas, there is a high likelihood that they would be available to count in at least one of our sites during one of our surveys (Fig. 4).

Concerns of this approach. While the use of mark-recapture is at the foundation of most population analyses (Williams et al. 2002), some scientists have voiced concerns regarding mark-recapture estimation techniques (Johnson 2008), because of some of the assumptions involved in the implementation of some of these methods. However, the superpopulation

estimator avoids all of the major concerns that are commonly mentioned (e.g., population closure, not accommodating movement, ignoring unbanded individuals). Ultimately, the two key parameters estimated from banded birds are p and ϕ , which has been done using the long-standing Cormack-Jolly-Seber method from mark-recapture (Williams et al. 2002).

Simulations to determine the sensitivity of this approach. To address some potential concerns regarding the reliability of this approach in estimating population size of snail kites, we developed a simulation model. Simulation models are useful in this regard because in such models, investigators know the ‘truth’ regarding the status of the population and can then determine if sampling and estimation can provide reliable measures of the true status of the population. Such assessments are impossible when relying on comparing estimates and approaches from field data alone. Consequently, we can use simulations to determine the adequacy of the way snail kite population size is estimated under a range of scenarios regarding potential variation in the biology of kites and sampling approaches.

We developed these simulations to capture general potential issues regarding snail kite sampling and population size estimation. Our model started with a scenario where there were 33 wetlands that kites use, 90% of which (30) are sampled; 30 sites sampled reflects approximately the number of spatial replicates of field surveys throughout the field sampling within wetlands. The model then simulated kite movement and mortality among wetlands over 6 time periods, reflecting the repeated surveys conducted in field sampling between March-June. For each time step, the numbers of kites in each wetland were summarized and the total population size across all sites (true population size) was tallied. This model included the fact that some kites are banded whereas others are not banded (unmarked) and that beyond the fact that a small portion of sites were not sampled, individuals could temporarily move out of the observed wetland (e.g., to forage in nearby areas).

We then sampled from this true state of population size in a way that reflects the estimation approach used in the superpopulation estimator (Dreitz et al. 2002). This included acknowledging that individuals may not be detected (detectability, $p < 1$) and that not all sites are sampled. With this sample of observed marked and unmarked birds, we calculated population size using the superpopulation estimator and contrasted this with simply using the maximum count of birds over the six surveys and the average count of birds over the six surveys (Martin et al. 2007a).

With this general template, we varied parameters in a factorial fashion to assess the reliability and limits of the superpopulation estimator. We varied true population size, N , from 300-5000 birds, the proportion of birds in the population that are marked from 0.05-0.9, the true detection probability, p , of birds from 0.15-0.9, the rate of temporary movement outside of our sites from 0-0.9. For each combination, we ran 25 replicate simulations, resulting in 76142 simulations. For each simulation, we estimated population size, detection probability (note that observed detection probability estimates may capture both the underlying ‘true’ detection probability in the model and the issue of temporary movement), the time that individuals spent outside of the sites sampled (either due to being in sites not sampled or from temporary movements outside of sites; see above), the maximum count of birds over the six surveys, the average count of birds over the surveys, and measures of bias of the estimator (relative bias, or the proportional difference of the estimate from the true population size and root mean squared error).

RESULTS

Reproduction

Number of nests counted

A total of 344 active nests (i.e., containing eggs or nestlings) were located range-wide in 2013 (Table 2). Of the nests monitored, forty-five percent of the nests occurred in the KRV, with Lakes Toho and Kissimmee each accounting for 17% of the range-wide nesting effort (Table 2). Nesting effort in Lake Okeechobee and WCA 3A increased in 2013 in comparison to 2012: Okeechobee accounted for 25% of the range-wide nesting effort in 2013, and WCA3A accounted for 13% (Table 2). As has largely been the case since 2005, the majority of the nesting effort continues to be concentrated in the KRV (Figure 3a).

Number of juveniles banded and number of young fledged

We were able to confirm the nest fate for 308 of the 344 active nests (Table 2). From these nests of known fate, 174 young were confirmed to have fledged (Table 2). We banded 251 nestlings during the pre-fledging stage and six additional fledglings just after these birds fledged from their nests. 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. In Figure 5, we also provide summaries of the number of birds banded over the years of this program, the number of birds re-sighted during breeding surveys (note that across the entire year, we re-sight more individuals), and the minimum number of band birds known to be alive in each year (based on re-sighting birds in the year of interest or in future years).

The total number of young fledged throughout the entire state that we sampled dropped substantially after 1998 (Figure 6). In 2011 and again in 2012 the number of young fledged was greater than in any year since 1998, yet the number of young observed to fledge in 2013 was down slightly from 2011-2012 (Figure 6). In 2013, Okeechobee contributed 24% of young fledged range-wide (Table 2). The STAs contributed 18%, Lake Toho 17%, and Lake Kissimmee 14% of young fledged range-wide, while other wetlands made relatively minor contributions (Table 2). As has largely been the case since 2005, the KRV is still responsible for the majority of all kite production, but Lake Okeechobee has been an increasingly major contributor to kite production (Figures 6, 7). We emphasize that these numbers are uncorrected counts that ignore potential variation in the ability to find nests across wetlands. We are currently assessing variation in nest detectability among wetlands and documenting fledging events; these findings will be discussed in a subsequent report.

Nest success and productivity

In 2013, the range-wide average of apparent nest success was 0.43 (SE = 0.03). Lake Okeechobee, Lake Toho, and the STAs had the largest number of successful nests (Figure 8). Bringing down the range-wide average were wetlands with substantial nesting effort but low nest success, including Lake Kissimmee and WCA3A (Table 2). Annual estimates of nest success (1995-2013) in WCA3A and across the entire range are presented in Figures 9a and 9b, respectively. In 2013, range-wide nest productivity (i.e., mean number of young fledged per successful nest) was 1.33 (SE = 0.04), down from 1.98 in 2012. Nest productivity showed some variation among wetlands in 2013, with WCA 3A and Lake Istopoga having the lowest productivity and Lake Kissimmee having the highest productivity (Table 2, Figure 10).

Apparent 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 (QAICc weight = 1; Table 1). 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 was best modeled by drought and era effects. The models most supported by the data (Model 1 and Model 2 in Table 1, which held 75% and 16% of the QAICc weight, respectively) assumed that adult survival differed between eras and was influenced by drought during the pre-range-shift era. In Model 1, drought effects during the pre-range-shift era were assumed to differ between moderate and severe droughts (i.e., 2000 \neq 2001), while in Model 2 these drought effects were assumed to be similar (i.e., 2000=2001) (Table 1). Beta parameter estimates (beta parameters are parameters on the link scale used in the model, in this case it is a logit, or log odds scale) of Model 1 indicated a stronger negative effect of the severe 2001 drought ($\beta_{2001} = -1.74$, 95% CI = -2.21 – -1.27) relative to the moderate 2000 drought ($\beta_{2000} = -0.36$, 95% CI = -1.43 – 0.70) and that adult survival was lower during the post-range-shift era ($\beta_{\text{Era2}} = -0.52$, 95% CI = -0.77 – -0.26). Note, however, that β_{2000} and β_{2001} have overlapping confidence intervals and that the confidence interval of β_{2000} overlaps 0. Beta parameter estimates of Model 2 indicated similar effects ($\beta_{2000=2001} = -1.21$, 95% CI = -1.57 – -0.85; $\beta_{\text{Era2}} = -0.53$, 95% CI = -0.79 – -0.28).

Models that included drought effects during the post-range-shift era received some, albeit little, support from the data, as Model 3 and Model 4 carried 6 and 2% of the QAICc weight, respectively (Table 1). However, in Model 3, which also included an era effect, the beta parameter estimate for drought effect during Era 2 was minimal and confidence intervals overlapped 0 ($\beta_{2007=2009=2011=2012} = 0.004$, 95% CI = -0.547–0.555). In Model 4, which did not include an explicit era effect, the beta parameter estimate for drought during Era 2 was significant ($\beta_{2007=2009=2011=2012} = -0.62$, 95% CI = -0.96 – -0.28) but lower than the beta parameter estimate for drought during Era 1 ($\beta_{2000=2001} = -1.01$, 95% CI = -1.33 – -0.69).

Based on the PHDI index, drought conditions in South Florida occurred during 2000, 2001, 2007, 2009, 2011 and 2012. Apparent adult survival was strongly affected by drought during the pre-range-shift era; however, the effect of drought during the post-range-shift era is less clear. Adult survival appears to have declined slightly during the post-range-shift era (Figure 11a), and this may be due greater drought frequency during Era 2 and/or to factors

associated with increased use of the KRV during the breeding season (e.g., colder temperatures). Although nesting outside the KRV and Lake Okeechobee was been minimal between 2007-2012 (Figure 7), adults do continue to utilize palustrine systems in South Florida, especially during the non-breeding season. Therefore, adult survival may still be influenced by the hydrological conditions in the southern portions of the range. However, model selection suggests that something other than regional hydrologic conditions in South Florida may be affecting range-wide adult survival during Era 2 (Table 1).

Juvenile survival has varied widely over time and reached a record low in 2000-2001 (Figure 11a). Juvenile survival also decreased significantly in the years 2004 to 2006 and rebounded in 2007 (Figure 11a). Although South Florida experienced drought conditions in 2007, 2009, 2011 and 2012, range-wide juvenile survival seemed to be unaffected by these conditions. This is likely due to the fact that the majority of young fledged in these years came from the KRV (Figure 6), and tended to remain in these lacustrine wetlands until reaching adult age. Lake levels in the KRV have historically been less affected by adverse drought conditions (Bennetts & Kitchens 1997a).

Detection

Annual estimates of detection probability for survival analyses have varied from year to year, but generally increased since 1992 (Figure 11b). In addition to differences in detection through time, preliminary evidence suggests that encounter rates (a component of detection probabilities) differ between wetland types (see Appendix A in Reichert et al. 2010a). Although the current modeling approach does not account for site-specific detection, we are actively working on methods to account for this variation.

Population Size

Estimates

The snail kite population in Florida progressively and dramatically decreased between 1999 and 2002 from approximately 3400 to 1700 adult birds (Figure 12). 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 (Figure 12). It seems that the population has slowly been increasing since 2009, and the population size estimate for 2012, 1218 (1088-1347 CI), is significantly greater than the number of individuals estimated for 2011 (Figure 12). In 2013, the population estimate was 1198 (1086-1309 CI), essentially stable from 2012. We are cautiously optimistic that the population is entering a biologically meaningful recovery phase.

We also report population growth rates on the basis of these population estimates (Figure 13). Our best estimates of population growth rates show a positive growth rate from 2009-2012, and an essentially stable growth rate (0.98) from 2012 to 2013.

Assessing limitations of the estimator

We provide preliminary results from 76,142 simulations of factors influencing the superpopulation estimator. Overall, the estimator used for assessing population size of snail kites was accurate across a wide-range of conditions simulated, whereas simply using a maximum count over the breeding season provided highly biased results, with population sizes always underestimated. We note that using average counts showed nearly identical patterns to using maximum counts, so we do not show those results here. In Figure 13, we show results of the simulations for parameter combinations that reflect realistic scenarios that hold most parameters at a realistic value based on field data and then alters other key parameters of interest. We emphasize that for estimates from field data, observed detection probability for breeding season surveys has ranged from 0.26-0.57 from 1997-2013, and we expect that the proportion of the population that is banded has ranged from approximately 25-50% over the years (see Figure 5 for banding information). Based on recent radio-telemetry work by Ken Meyer (Meyer and Kent 2011), we expect that individual birds might spend approximately 50% of their time outside our sample sites.

When varying true population size from 300-5000 individuals, we find that our population estimator reliably estimates population size when birds spend 50% of their time outside our sample and observed detection probability is low but within the bounds of field data (0.33). Varying the proportion of birds marked does not change population estimates considerably, although uncertainty in estimates increases slightly with a smaller proportion of birds marked. The estimator appeared most sensitive to increasing amount of time birds spend

outside the sampled area (Figure 14), but this sensitivity only occurred in extreme situations (Figure 15): when birds were simulated to spend very large amounts of time spent outside the sample ($>90\%$ of time) and when observed detection probability was very low ($p < 0.1$) and true population size was very low (<600 birds).

To more formally identify the factors most responsible for bias in our estimator, we used both a general linear model and a regression tree approach. General linear models (GLM) allow formal consideration of linear effects and their interactions, whereas regression trees allow identification of possible key thresholds (non-linearity) in effects, which we expected. The GLM found that the only factors influencing bias was time outside the sample (t-value = 2.73, $P < 0.006$) and the interaction of time outside the sample and population size (t-value = -2.13, $P < 0.035$). A regression tree based on the relative bias of the estimator found that bias could best be explained by the time birds spent outside the sample, breaking the data into $>90\%$ and $<90\%$.

Ultimately, these simulations suggest that the estimator is very reliable, on average, under situations that appear to be occurring in the field and is generally robust across variation in true population size, the proportion of birds marked in the population, the amount of time kites spend outside our sample, and detection probability of kites. However, extreme combinations of these factors may cause the estimator to perform poorly (note that simple counts also had similar problems). In addition, the summaries provided focus on average estimates under these scenarios; we are currently assessing how these issues influence the uncertainty around estimates of population size. Based on these preliminary results, a useful barometer in the context of current monitoring might be to closely track variation in estimates of detection probability from breeding surveys. If estimates of observed detection probability begin approaching values of 0.1 or lower (the lowest estimate from 1997-2013 was 0.25 in 2004), then this estimator may become less reliable. We are continuing to refine these simulations and further assess issues regarding the adequacy of monitoring and estimation of population size.

DISCUSSION

Trends from our demographic studies revealed a drastic decline in the snail kite population in Florida between 2000-2008, with the population essentially halving from 2000 to 2002 and again from 2006 to 2008 (Figure 12) (see also Chapter 3 of Martin et al. 2007c). While population estimates have steadily increased or remained stable since 2009 (Figures 12, 13), it

remains unclear whether this trend signifies the beginning of a population rebound. For example, the observed trends from 2009-2013 are similar to patterns observed from 2003-2006.

Concurrent with the population decline were declines in the number of nesting attempts and the number of young fledged (Figures 6, 7), as well as a period of generally lower and more variable nest success (Figure 9) (see also Chapter 4, Martin et al. 2007c). Several lines of evidence suggest that these estimates and trends are generally reliable. First and foremost, preliminary results from simulations suggest that our estimator is robust to potential issues that could affect this monitoring program, particularly under scenarios that are likely occurring in the field (Figures 14, 15). While the superpopulation estimator was designed to deal with many of these factors, until now the approach had not been thoroughly assessed in a standardized way to interpret the reliability of this approach. The reliability of this approach stems from several aspects of the monitoring program, including the fact that six repeated surveys are conducted throughout the breeding season, every attempt is made to band all nestlings fledged from nests (which is a low-cost approach to obtaining a reasonable portion of the population being marked), and monitoring is relatively comprehensive across the region. The superpopulation estimator directly deals with the problem of imperfect detection, movement of kites into and out of the study area, and potential mortality of individuals (Dreitz et al. 2002). Furthermore, it is much less sensitive to survey effort than count-based approaches (Bennetts et al. 1999). Estimates for each year are also calculated only with information collected in that year. Consequently, if observed population declines were simply from birds moving out of study sites in a given year from poor environmental conditions (note they would have to move and stay out of all of the sites sampled in a given year), we would expect numbers to rebound in subsequent years, which has not been the case. Annual survival estimates (Fig. 11) and observed variation in reproduction (Fig. 6, 9), both independent data from breeding survey numbers, can adequately explain these changes.

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), as well as reductions to native apple snail populations (Darby et al. 2009). First of all, one of the major historic components of the kites' habitat network in Florida, Lake Okeechobee, remained relatively unproductive for over a decade, contributing minimally to kite reproduction from 1996 to 2010. This has been attributed, in part, to a shift in the water management regime (Bennetts & Kitchens, 1997a). Hurricanes in 2004

further degraded kite nesting and foraging habitat in Okeechobee (*personal observation*). From 1996-2006 most kite nesting attempts on Okeechobee occurred in the Clewiston Flats southwest of Moonshine Bay; an area suitable for kite nesting and foraging at lake stages above 15 feet NGVD. Since then, lake schedules have been lowered and normally range from 12-15 feet NGVD, and kite nesting has shifted accordingly. 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 and again in 2012, making Okeechobee the third most productive habitat, in terms of kite reproduction, in those years and production in 2013 was very high again (Table 2). In these recent years, nesting kites on Okeechobee have utilized exotic snails (primarily *Pomacea maculata*) almost exclusively (see also Reichert et al. 2011). Although native egg clusters were observed in several locations (especially near Observation Island), the re-colonization of Lake Okeechobee seems to be, at least in part, consistent with the recent appearance of exotic apple snails and the recovery of littoral marshes under the new lake schedule.

While the jump in production from Okeechobee over the last two years 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 will 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 increased frequency of local disturbance events due to cold weather and the treatment of *Hydrilla* (the effects of which are currently being investigated).

WCA3A has often been considered one of the 'most critical' wetlands within the range of the kite in Florida, acting as a stronghold for kite reproduction for several decades (Beissinger 1995; USFWS 1999, 2010; Martin et al. 2007c, 2008); however, reproductive effort and success in WCA3A decreased sharply after 1998 (Martin et al. 2008), with no young fledged there in

2001, 2005, 2007, 2008, or 2010. In 2012, only one successful nest, which fledged one young, was observed in WCA3A, but in 2013 more successful nesting occurred (Table 2). The low reproduction in WCA3A has 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 (Cattau et al. 2008) 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, USFWS 2010) and suppressed native apple snail populations (Darby et al. 2009). While cooperative efforts have been undertaken to minimize potential deleterious effects of water management (*see* USFWS 2010), the low levels of reproductive activity and success observed in WCA3A in recent years (Figures 7, 8, 9, 16) suggest that habitat conditions, such as snail density and vegetation, may remain poor throughout much the wetland (see Appendix A for more information on vegetation changes in WCA3A). The majority of the nesting kites in WCA3A that have been observed during the last two years were concentrated near the southern boundary, as they were foraging on exotic snails that likely entered WCA3A via the canal associated with Tamiami Trail (US 41) (Figure 17).

The relatively low reproductive output from Lake Okeechobee (1997-2010) and WCA3A (2001-present) left the kite population heavily concentrated in and dependent upon the KRV (Figure 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. This year, Toho accounted for 17% and 17% of all successful nests and fledged young, respectively.

2013 saw an increased in nesting effort on Lake Kissimmee and in storm water treatment areas (STA) 1E and 5. On Kissimmee some breeding individuals utilized novel nesting substrates, most notably a large floating tussock made of Cuban bulrush and primrose. The age distribution of individuals present on Kissimmee was younger than the range-wide age distribution in 2013. The majority of those individuals on Kissimmee were born in the northern portion of the breeding range (East Lake Toho and Lake Toho). STAs 1E and 5 supported a long nesting period starting in January and extending through to mid-October. Individuals breeding in STA 5 built nests in recently defoliated *Salix*. Nests in STA 1E were all located in *Typha* and experienced high apparent nesting success compared to other portions of the range (Table 1).

STA 1E contained very high densities of exotic snails (based on communications with SFWMD), and all snail shells collected from STA nests were exotic (n=16).

Shells collected at nests in recent years reveal that a large proportion of snails utilized by kites are exotic (see also Cattau et al. 2012). Extensive use of exotic snails by kites has occurred in the KRV since 2005 and more recently in Okeechobee, the STAs, and some WCAs, likely due to the continued spread of the exotic snail. 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 (see also Darby et al. 2007). 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 remained relatively stable (Figure 18; note that there was an error in reporting these data in the 2012 report). In addition, handling times remain higher for exotic snails than native snails (Table 3). 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). While kites appear to continue to be foraging on relatively large exotic snails, there have been no data to suggest detrimental demographic effects, as hypothesized in Cattau et al. (2010). There are several reasons why this may be the case, such as changes in foraging behavior (but see Table 3) or that the energetic deficit shown in Cattau et al. (2010) may not have translated to demographic changes in survival or reproduction. The increased nesting effort and reproductive output in many wetlands during the last two years may be attributable, at least in part, to the presence and abundance of the exotic apple snail. We are continuing to assess these potential linkages.

Additional work related to the nesting and foraging ecology of kites on Toho has recently been completed (Pias 2012, Olbert 2013), and should improve our understanding of the conditions necessary to facilitate successful reproduction and survival. Specifically, we found that the primary cause of nesting failure was the yellow rat snake (*Elaphe obsoleta quadrivittata*), which was the most common predator of both eggs and young (Olbert 2013). Contrary to previous literature (Sykes et al. 1995), nest collapse was not found to be an important cause of nest failure, even in non-woody substrates. In fact, nest collapse was only observed once

out of 92 nests (82 in non-woody substrate), and that nest was in a woody substrate. This suggests that high rates of collapse noted in earlier studies were probably due to predation that occurred between nest checks. We are currently attempting to isolate the importance of nest predation on population growth rates in snail kites.

Additionally, results indicate that habitat characteristics of snail kite nesting areas affected predation events; different predators were influenced by different habitat variables. For some terrestrial predators nest access was affected by distance of the nest patch to the shore, water depth, and nest height. While semi-aquatic predators, such as marsh rice rats (*Oryzomys palustris*), were influenced by the distance of the nest patch to shore, water depth, and minimum daily temperature. Also, nests initiated in February and March of the 2010 nesting season were more likely to be abandoned than any other nests observed during the two year study. This high rate of abandonment was likely due to unusually cold temperatures in the early breeding season. Peak nest initiation was also delayed during the cold weather events, until adults could provision young at a consistent rate.

The results of a study (Pias 2012) examining the foraging behavior of snail kites on Lake Toho indicate that foraging substrates that occurred off the main body of lake may have had higher snail availabilities, but snail kite nests built in off-lake areas may be more vulnerable to predation. Therefore, in spite of relatively lower snail availabilities, foraging substrates that occur on Lake Toho may be more valuable to snail kites than off-lake areas. Additionally, the availability of snails in on-lake foraging substrates is likely driven in part by water levels, highlighting the role of water management in maintaining foraging habitat for breeding snail kites.

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 KRV, leading to increased encounter rates for survey methodologies inherent to these areas. We are currently investigating possible improvements for modeling procedures of survival and population size (see Appendix A in Reichert et al. 2010a).

Recent mark-recapture analyses have elucidated age-dependent variation within adult snail kite demographic parameters. Specifically, results provide evidence of senescent declines in

snail kite survival and breeding probabilities, an effect that is exacerbated during harsh environmental conditions (Reichert et al. 2012). Based on a declining proportion of snail kites attempting to breed since 1996 (Reichert 2009) 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. 2010a). 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 (Martin et al. 2008), 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 important 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 to levels of reproduction observed in previous years. All proposed water and vegetation management actions should undergo critical evaluation processes to minimize the likelihood of 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. 2010a), which will aid future management decisions.

When the water regulation schedule in WCA3A does not mimic the seasonal patterns driven by the natural hydrological cycle, hydrology 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 (Cattau et al. 2008, 2012). 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 and 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.

Management of the kite population will also require landscape-scale considerations as movements appear more structured than previously thought (Martin et al. 2006, Fletcher et al. 2011, 2013). While kites can, and do, move long distances (see Bennetts and Kitchens 1997a, Meyer and Kent 2011), movements observed from standardized mark-resight data suggest that individuals tend to concentrate movements during the breeding season either within KCOL or throughout the much of rest of the range (Fletcher et al. 2013). 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 wetlands within the region to be affected, to serve as refuge for snail kites. Draw-downs of local wetlands 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 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 (e.g., hydrologic conditions, vegetative community compositions,

snail densities) in WCA3A and elsewhere. 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. 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. Single-state models of annual apparent survival and detection probabilities of Snail Kites in Florida from 1992-2013. Models included effects of Age (JUV: 30 days to 1 year old; AD: > 1 year), time (t), Era (pre-range-shift: 1992-2004; post-range-shift: 2005-2013), and regional hydrologic conditions in South Florida based on the 4-month (May–August) average Palmer Hydrological Drought Index (PHDI) for Florida Climate Division 5. Hydrologic conditions were modeled using either categorical drought effects (e.g., Drought_(year)) or a continuous covariate (i.e., PHDI).

| Model # | Apparent survival (Phi) | Detection (p) | Number of Parameters | QAICc | ΔQAICc | Weight | Deviance |
|---------|---|---------------|----------------------|---------|--------|--------|----------|
| 1 | JUV(t) + AD(Drought _{(2000)≠(2001)} + Era) | t | 46 | 9525.88 | 0.00 | 0.75 | 2960.29 |
| 2 | JUV(t) + AD(Drought ₍₂₀₀₀₌₂₀₀₁₎ + Era) | t | 45 | 9528.94 | 3.05 | 0.16 | 2965.38 |
| 3 | JUV(t) + AD(Drought _{(2000=2001)≠(2007=2009=2011=2012)} + Era) | t | 46 | 9530.97 | 5.09 | 0.06 | 2965.38 |
| 4 | JUV(t) + AD(Drought _{(2000=2001)≠(2007=2009=2011=2012)}) | t | 45 | 9533.52 | 7.64 | 0.02 | 2969.96 |
| 5 | JUV(t) + AD(Drought ₍₂₀₀₀₌₂₀₀₁₌₂₀₀₇₌₂₀₀₉₌₂₀₁₁₌₂₀₁₂₎) | t | 44 | 9535.53 | 9.65 | 0.01 | 2974.01 |
| 6 | JUV(t) + AD(Drought _{(2000=2009)≠(2001=2007=2011=2012)}) | t | 45 | 9537.08 | 11.19 | 0.00 | 2973.52 |
| 7 | JUV(t) + AD(Drought _{(2000=2009)≠(2001=2007=2011=2012)} + Era) | t | 46 | 9538.53 | 12.65 | 0.00 | 2972.94 |
| 8 | Age * t | t | 62 | 9540.47 | 14.59 | 0.00 | 2942.24 |
| 9 | JUV(t) + AD(PHDI) | t | 44 | 9543.09 | 17.21 | 0.00 | 2981.56 |
| 10 | JUV(t) + AD(PHDI + Era) | t | 45 | 9544.59 | 18.71 | 0.00 | 2981.03 |
| 11 | JUV(t) + AD(Era) | t | 44 | 9550.77 | 24.89 | 0.00 | 2989.24 |
| 12 | Age + t | t | 43 | 9559.26 | 33.38 | 0.00 | 2999.76 |
| 13 | Age + Drought _{(2000=2001)≠(2007=2009=2011=2012)} + Era | t | 26 | 9592.43 | 66.55 | 0.00 | 3067.37 |
| 14 | Age + Drought ₍₂₀₀₀₌₂₀₀₁₎ + Era | t | 25 | 9601.77 | 75.89 | 0.00 | 3078.72 |
| 15 | Age + Drought _{(2000)≠(2001)} + Era | t | 26 | 9602.22 | 76.34 | 0.00 | 3077.16 |
| 16 | Age + Drought _{(2000=2001)≠(2007=2009=2011=2012)} | t | 25 | 9616.38 | 90.50 | 0.00 | 3093.33 |
| 17 | Age + Drought _{(2000=2009)≠(2001=2007=2011=2012)} | t | 25 | 9624.22 | 98.34 | 0.00 | 3101.18 |
| 18 | Age + Drought _{(2000=2009)≠(2001=2007=2011=2012)} + Era | t | 26 | 9626.24 | 100.36 | 0.00 | 3101.18 |
| 19 | Age + PHDI + Era | t | 25 | 9640.67 | 114.78 | 0.00 | 3117.62 |
| 20 | Age + PHDI | t | 24 | 9641.73 | 115.85 | 0.00 | 3120.70 |
| 21 | Age + Drought ₍₂₀₀₀₌₂₀₀₁₌₂₀₀₇₌₂₀₀₉₌₂₀₁₁₌₂₀₁₂₎ | t | 24 | 9649.97 | 124.09 | 0.00 | 3128.94 |
| 22 | Age | t | 23 | 9661.90 | 136.02 | 0.00 | 3142.89 |
| 23 | Age + Era | t | 24 | 9662.50 | 136.61 | 0.00 | 3141.47 |
| 24 | Age * Drought _{(2000=2001)≠(2007=2009=2011=2012)} + Era | t | 28 | 9743.28 | 217.40 | 0.00 | 3214.17 |
| 25 | Age * Drought ₍₂₀₀₀₌₂₀₀₁₎ + Era | t | 26 | 9745.77 | 219.88 | 0.00 | 3220.70 |
| 26 | Age * Drought _{(2000)≠(2001)} + Era | t | 28 | 9749.32 | 223.44 | 0.00 | 3220.21 |
| 27 | Age * Drought _{(2000=2009)≠(2001=2007=2011=2012)} + Era | t | 28 | 9797.30 | 271.41 | 0.00 | 3268.19 |
| 28 | Age * Drought _{(2000=2001)≠(2007=2009=2011=2012)} | t | 26 | 9825.68 | 299.79 | 0.00 | 3300.61 |

Table 1 continued. Single-state models of annual apparent survival and detection probabilities of Snail Kites in Florida from 1992-2013. Models included effects of Age (JUV: 30 days to 1 year old; AD: > 1 year), time (t), Era (pre-range-shift: 1992-2004; post-range-shift: 2005-2013), and regional hydrologic conditions in South Florida based on the 4-month (May–August) average Palmer Hydrological Drought Index (PHDI) for Florida Climate Division 5. Hydrologic conditions were modeled using either categorical drought effects (e.g., Drought_(year)) or a continuous covariate (i.e., PHDI).

| Model | | Detection | Number of | | | | |
|-------|--|-----------|------------|---------|--------|--------|----------|
| # | Apparent survival (Phi) | (p) | Parameters | QAICc | ΔQAICc | Weight | Deviance |
| 29 | Age * PHDI + Era | t | 26 | 9826.71 | 300.83 | 0.00 | 3301.65 |
| 30 | Age * Era | t | 24 | 9834.40 | 308.52 | 0.00 | 3313.37 |
| 31 | Age * Drought _{(2000=2009)≠(2001=2007=2011=2012)} | t | 26 | 9838.11 | 312.23 | 0.00 | 3313.04 |
| 32 | Age * Drought ₍₂₀₀₀₌₂₀₀₁₌₂₀₀₇₌₂₀₀₉₌₂₀₁₁₌₂₀₁₂₎ | t | 24 | 9862.46 | 336.57 | 0.00 | 3341.43 |
| 33 | Age * PHDI | t | 24 | 9909.00 | 383.11 | 0.00 | 3387.97 |
| 34 | . | t | 22 | 9961.77 | 435.89 | 0.00 | 3444.78 |

Table 2. Wetland-specific nesting summary statistics for the 2013 snail kite breeding season. The category “Other” includes the following wetlands (with the number of active nests in parentheses): Everglades National Park (2), St. Johns Marsh (1), Grassy Waters Preserve (2), Hungryland (2), and Loxahatchee NWR (1). Stormwater Treatment Area (STA) below is a combination of activity at STAs 1 and 5. Six additional fledglings (not included in the table) were banded, 5 on Kissimmee and 1 on Lake Istokpoga.

| Wetland | Total number of active nests ^A | Number of nests with known fate ^B | Number of successful nests ^C | Number of young banded | Observed number of young fledged ^D | Daily survival rate | Relative contribution to range-wide nesting effort | Relative contribution to range-wide fledgling production | Apparent nest success (\pm SE) | Average nest productivity (\pm SE) ^E |
|--------------|---|--|---|------------------------|---|---------------------|--|--|-------------------------------------|--|
| WCA3A | 43 | 35 | 12 | 23 | 13 | 0.975 | 0.13 | 0.07 | 0.34 (\pm 0.08) | 1.08 (\pm 0.08) |
| WCA3B | 7 | 6 | 5 | 4 | 6 | 0.986 | 0.02 | 0.03 | 0.83 (\pm 0.15) | 1.20 (\pm 0.20) |
| Okeechobee | 86 | 77 | 29 | 50 | 42 | 0.975 | 0.25 | 0.24 | 0.38 (\pm 0.06) | 1.45 (\pm 0.11) |
| STA | 44 | 39 | 23 | 49 | 31 | 0.988 | 0.13 | 0.18 | 0.59 (\pm 0.08) | 1.35 (\pm 0.10) |
| Istokpoga | 11 | 10 | 6 | 10 | 6 | 0.984 | 0.03 | 0.03 | 0.60 (\pm 0.15) | 1.00 (\pm 0.00) |
| Kissimmee | 58 | 51 | 15 | 44 | 24 | 0.974 | 0.17 | 0.14 | 0.29 (\pm 0.06) | 1.60 (\pm 0.16) |
| Toho | 57 | 54 | 24 | 47 | 30 | 0.982 | 0.17 | 0.17 | 0.44 (\pm 0.07) | 1.25 (\pm 0.09) |
| East Toho | 23 | 23 | 10 | 15 | 12 | 0.983 | 0.07 | 0.07 | 0.43 (\pm 0.10) | 1.20 (\pm 0.13) |
| Lake Jackson | 7 | 6 | 2 | 5 | 3 | 0.975 | 0.02 | 0.02 | 0.33 (\pm 0.19) | 1.50 (\pm 0.50) |
| Other | 8 | 7 | 5 | 4 | 7 | 0.980 | 0.02 | 0.04 | 0.71 (\pm 0.17) | 1.40 (\pm 0.24) |
| Total | 344 | 308 | 131 | 251 | 174 | 0.979 | | | 0.43 (\pm 0.03) | 1.33 (\pm 0.04) |

^A Used to calculate relative contribution to range-wide nesting effort

^B Sample sizes for estimates of apparent nest success

^C Sample sizes for estimates of nest productivity

^D Includes some nestlings that were found when ≥ 24 days old whose nests were not used to estimate nest productivity

^E Mean number of young fledged per successful nest

Table 3. Results of time activity budgets conducted in the Kissimmee Chain of Lakes and Water Conservation Area 3A during 2013. Foraging snail kites are chosen at random and observed for up to one hour. Behaviors are observed and recorded to the nearest second. Snail shells of snails captured by kites during observations are collected so that snail species and size can be directly linked to individual kites and extraction times. This is an update of work that is currently ongoing.

| | Average Handling Time (sec) | Handling Time SE | N | Average Snail Length (mm) | Snail Length SE |
|--------------------------|-----------------------------------|---------------------|----|---------------------------------|--------------------|
| Adult – Exotic Snails | 228.12 | 24.13 | 52 | 51.64 | 1.64 |
| Adult – Native Snails | 107.84 | 9.42 | 81 | 37.57 | 1.80 |
| Juvenile – Exotic Snails | 237.00 | 64.80 | 7 | 47.22 | 5.57 |

Figure 1. Study area, with the numbers indicating the wetland regions sampled during annual population monitoring surveys. Kissimmee-Chain-of-Lakes includes Lakes Tohopekaliga, East Toho, Kissimmee, Runnymede, Jackson, Marian, Cypress, Weohyakapka, Arbuckle, and Hatchineha. St. John's Marsh includes East and West portions of the Blue Cypress Marsh Complex and surrounding wetlands. Loxahatchee Slough includes Grassy Waters, Hungryland Wildlife Management Area, Corbett WMA, and surrounding wetlands.

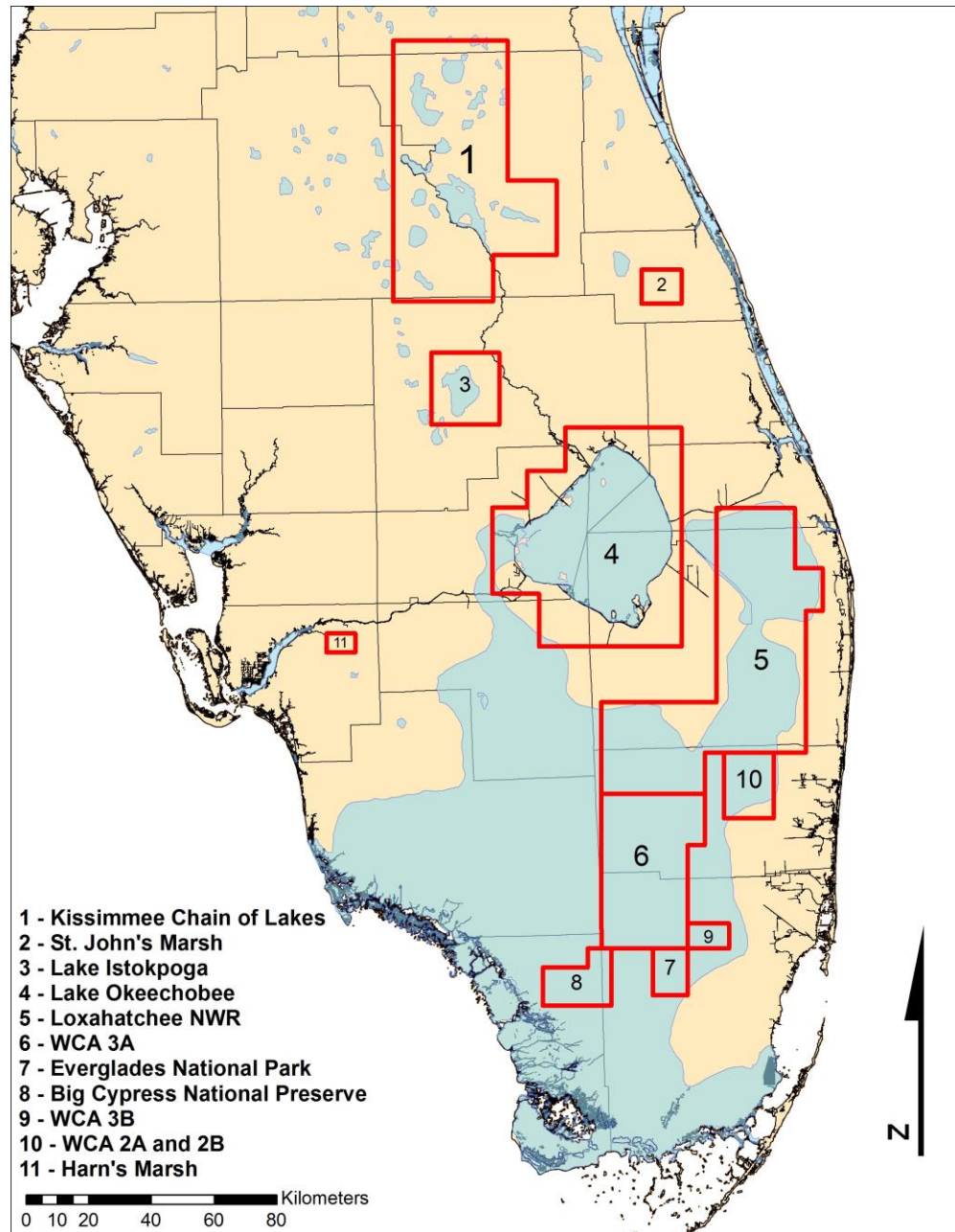
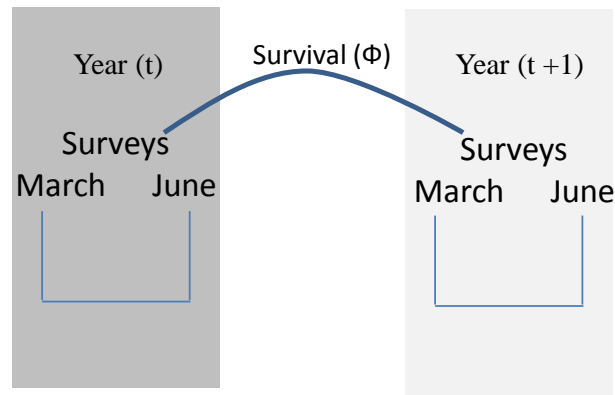


Figure 2. (a) Diagram illustrating apparent survival estimation with Cormack-Jolly-Seber models, and (b) example detection histories of re-sighting kites and their interpretation.

(a) Apparent survival (ϕ)



(b) Example detection histories across 5 years

11100: *individual was alive in years 1-3; for years 4-5, either individual suffered mortality or was alive and not detected (mark-recapture analysis acknowledges both possibilities)*

10001: *individual was alive across all five years, but not re-sighted in years 2-4. This information is used to estimate detectability of individuals. By knowing detectability, we can better interpret the above situation regarding whether individuals suffer mortality or are simply not detected*

Figure 3. Four-month (May–August) average Palmer Hydrological Drought Index (PHDI) for Florida Climate Division 5. PDHI is a long-term, lagging indicator for drought and wet conditions, reflecting groundwater conditions and reservoir levels. The period May–August corresponds to the post-fledging “Summer” season (*sensu* Bennetts & Kitchens 1997). This period encompasses the transition from the dry to the wet season (annual low water levels in South Florida wetlands are typically reached in May/June). A low average PHDI value for May–August may reflect a drier than average dry season and/or a dryer than average (or delayed) onset of the wet season.

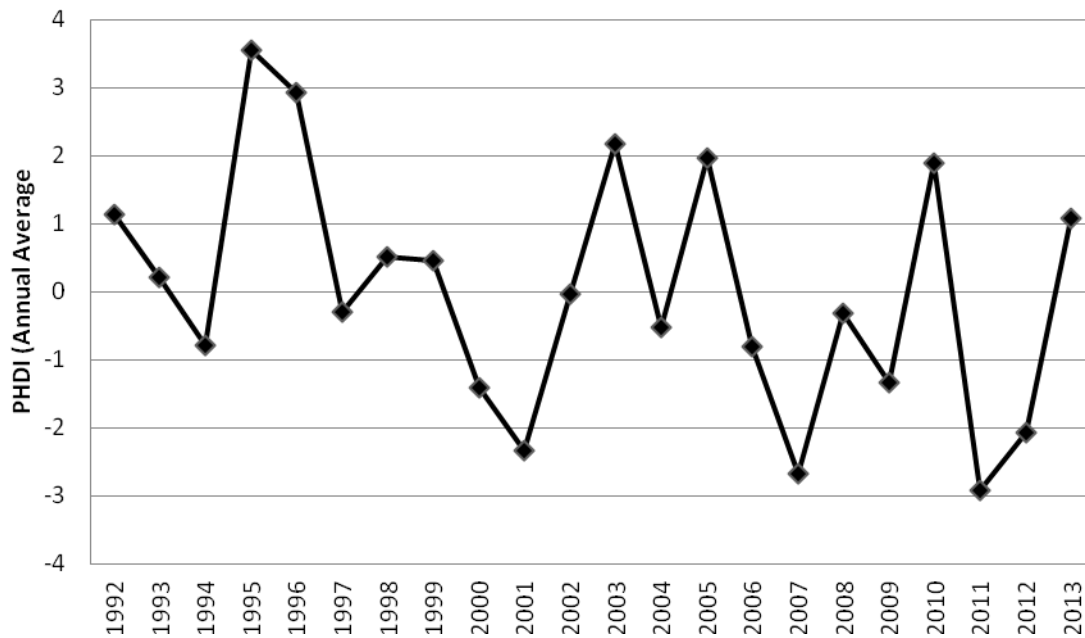


Figure 4. Amount of time kites may spend in our study area and availability of kites to be sampled. Because we re-visit each wetland 6 times, individuals only need to be present in the wetland in 1 of 6 visits. So, the probability that an individual will be in the wetlands at least on one occasion = $1 - (1 - \text{time inside})^6$, or the 1 minus the probability that it is outside the study area for all six occasions. Note that the shown relationship does not account for whether or not we detect an individual given that it was available to be counted, but the superpopulation estimator does account for this issue.

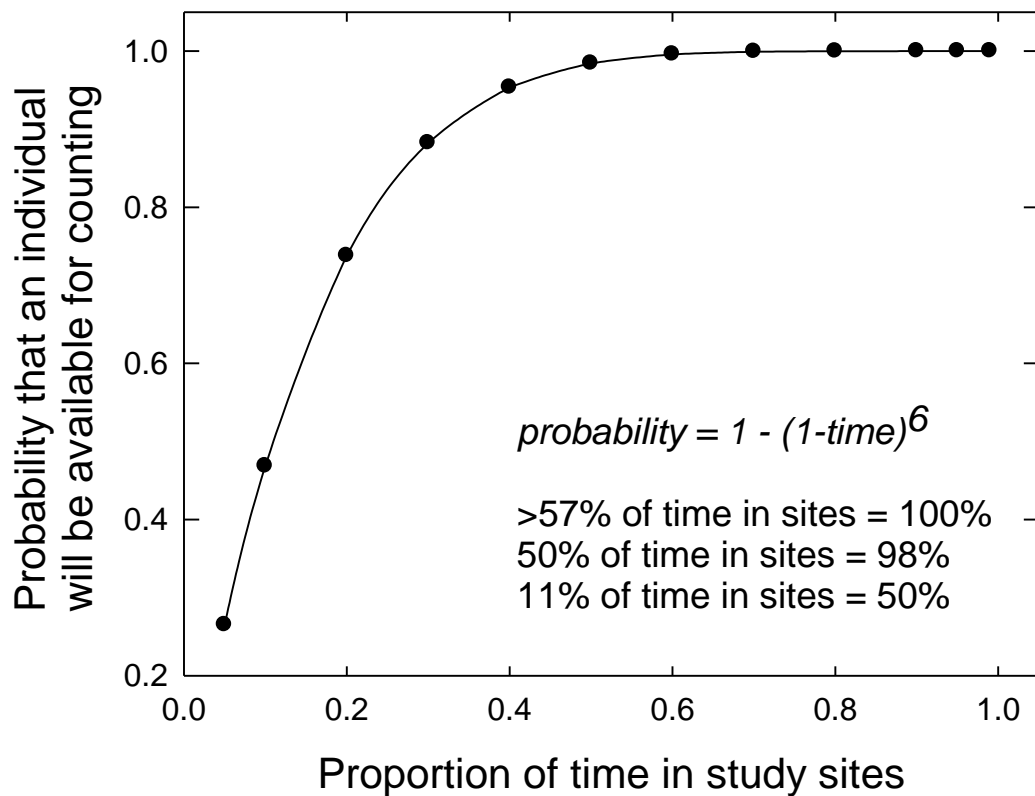


Figure 5. The number of birds banded each year, the number of banded adults re-sighted annually (during breeding surveys, March-June), and the minimum number of banded adults known to be alive each year. The latter includes all adults re-sighted each year i , plus adults re-sighted in subsequent years that were originally banded in years prior to year i . Gray area shows time period before population size estimation that uses information on banded birds began.

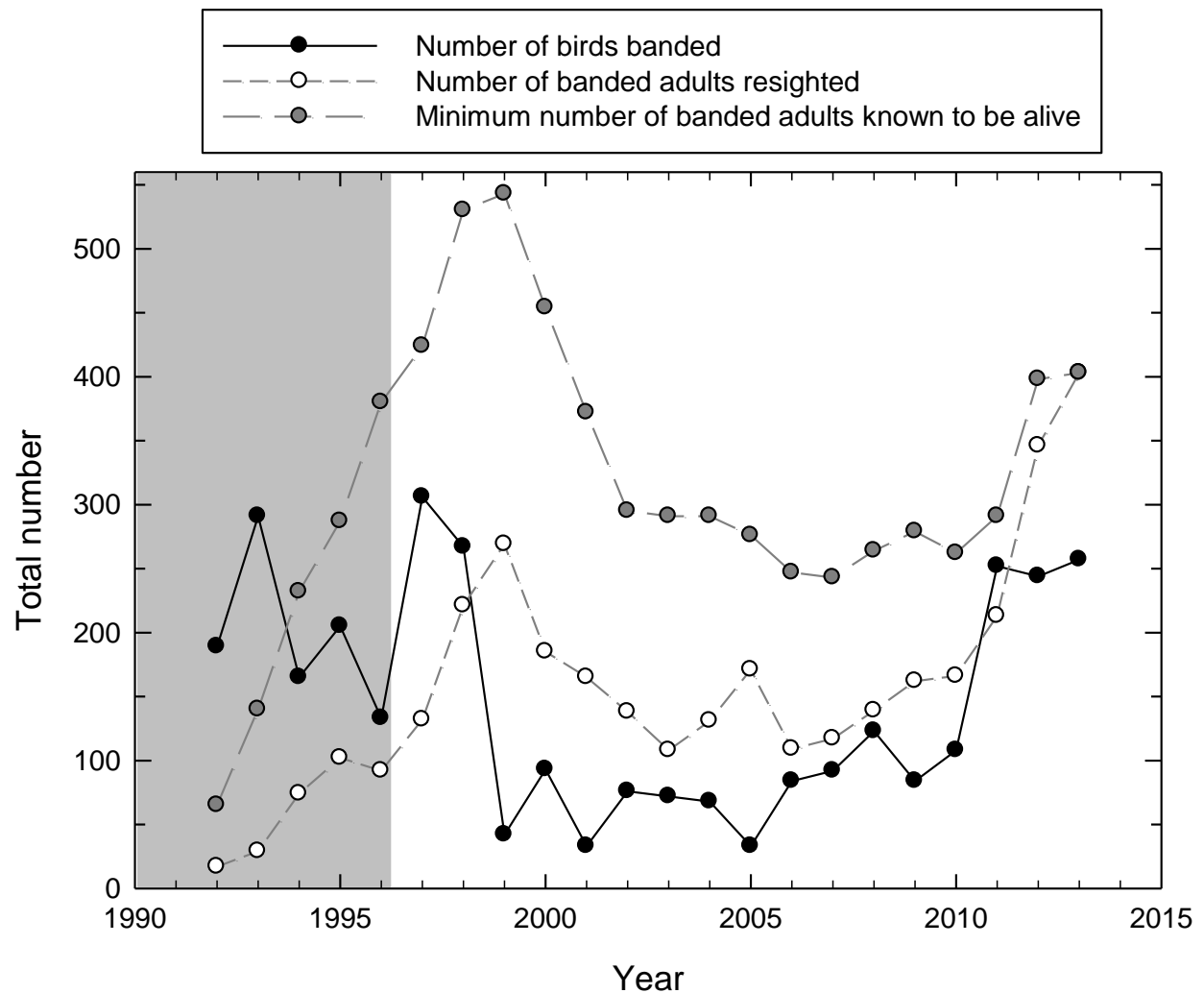


Figure 6. Number of young confirmed to have fledged, 1992-2013. Note that these values represent raw counts (uncorrected for detection) of young that reached the minimum fledging age (i.e., 24 days old) in monitored nests. KRV = Kissimmee River Valley, which includes the Kissimmee Chain of Lakes and Lake Istokpoga; SJM = St. John's Marsh; Everglades includes Everglades National Park, Big Cypress National Preserve, and all WCAs and STAs.

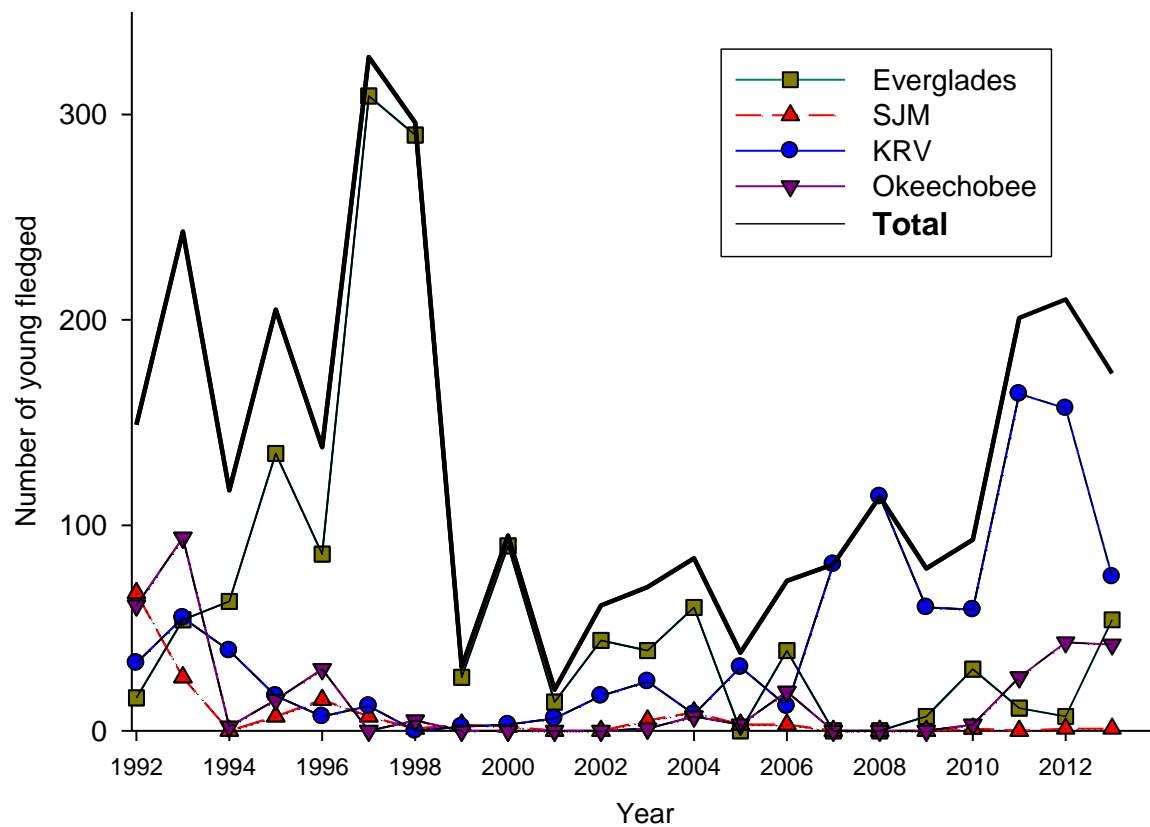


Figure 7. Relative annual contribution of wetland regions to the total number of (a) active nests and (b) young fledged, on a range-wide basis from 1995–2013. Everglades includes Everglades National Park, Big Cypress National Preserve, and all WCAs and STAs; KRV = Kissimmee River Valley, which includes the Kissimmee Chain of Lakes and Lake Istokpoga; Other includes St. John’s Marsh, Grassy Waters, and other peripheral areas.

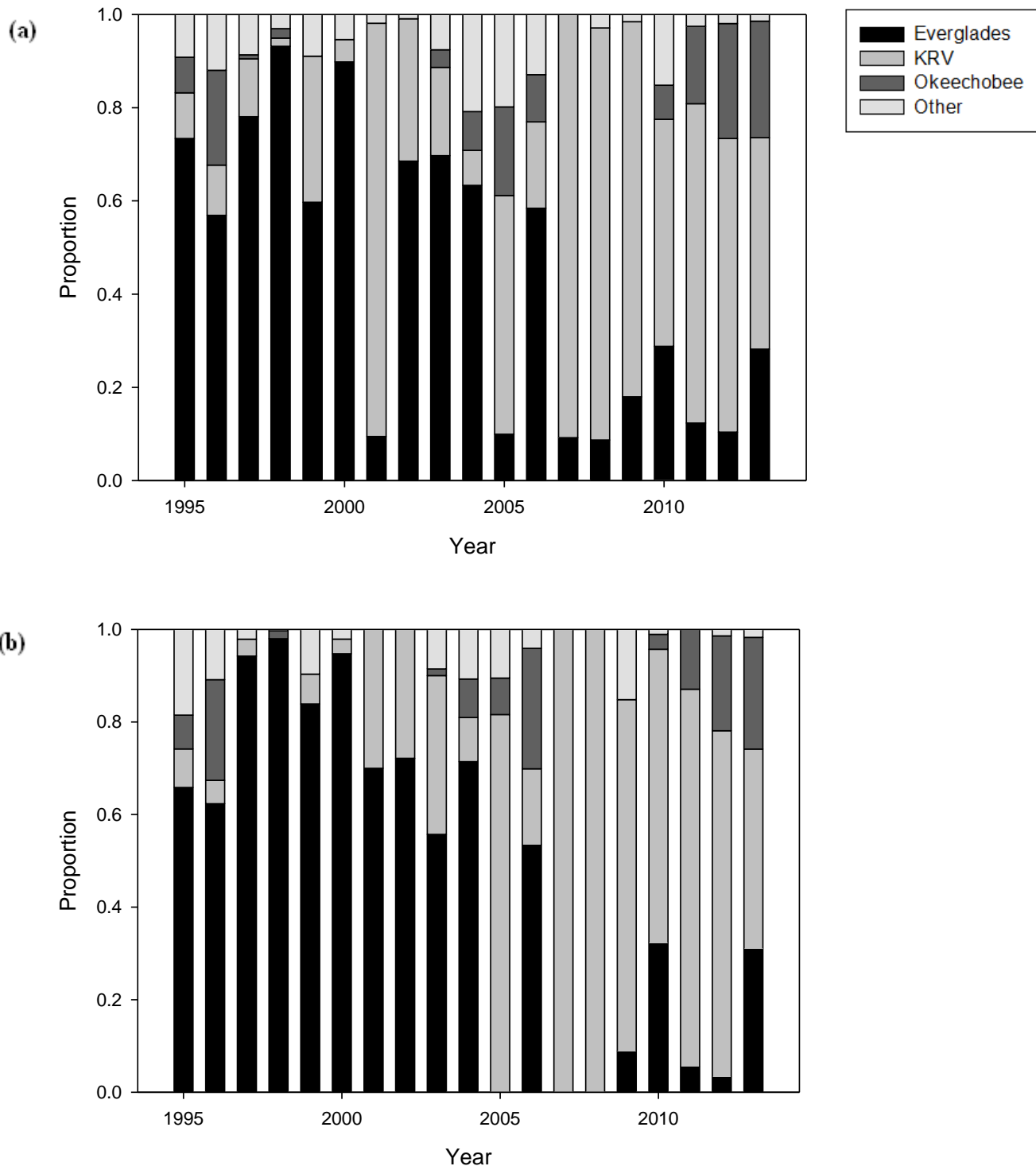


Figure 8. Map of the relative proportion of successful nests across the range, 2013.

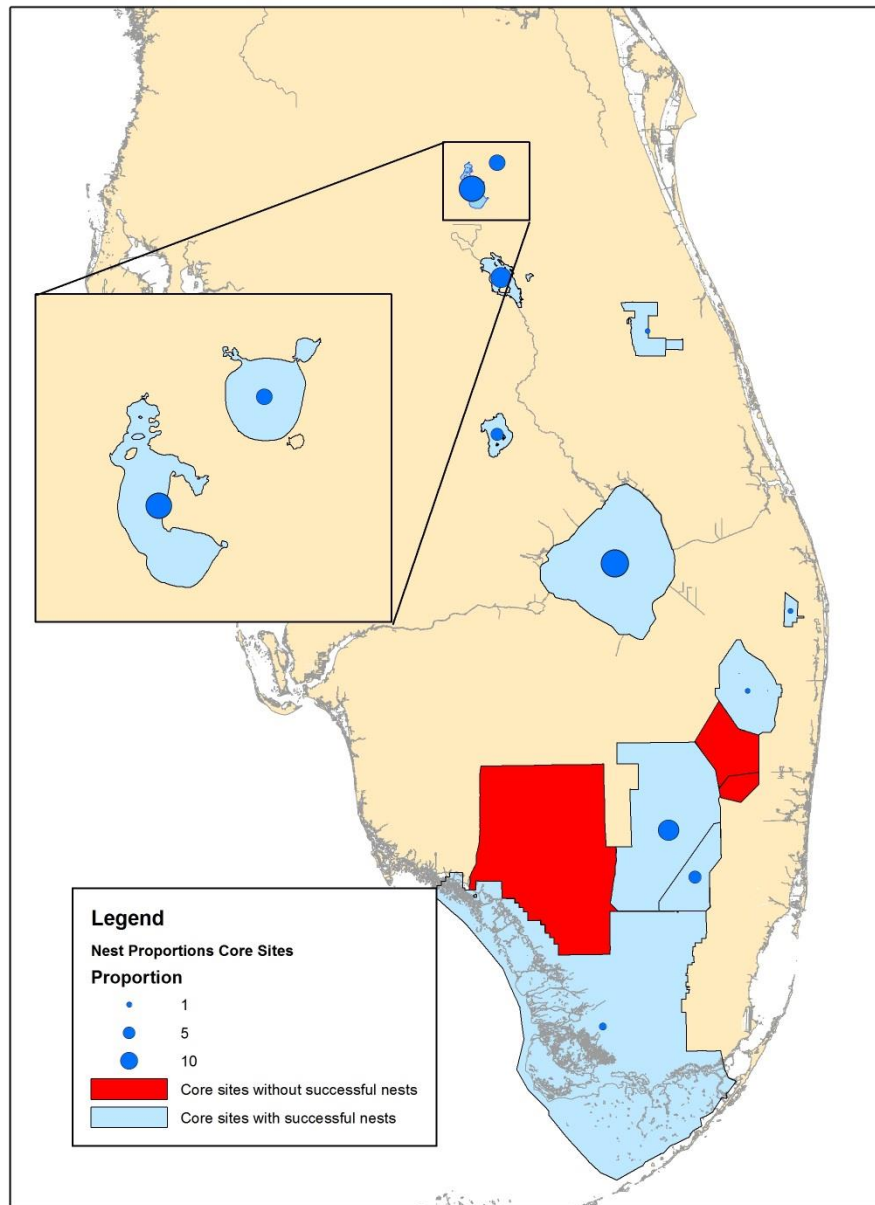


Figure 9. Apparent nest success (\pm SE), 1995-2013 in (a) WCA3A, and (b) range-wide. Note that in 2005, 2007 and 2010, all nests in WCA3A failed, while in 2001 and 2008, no active nests were observed in WCA3A.

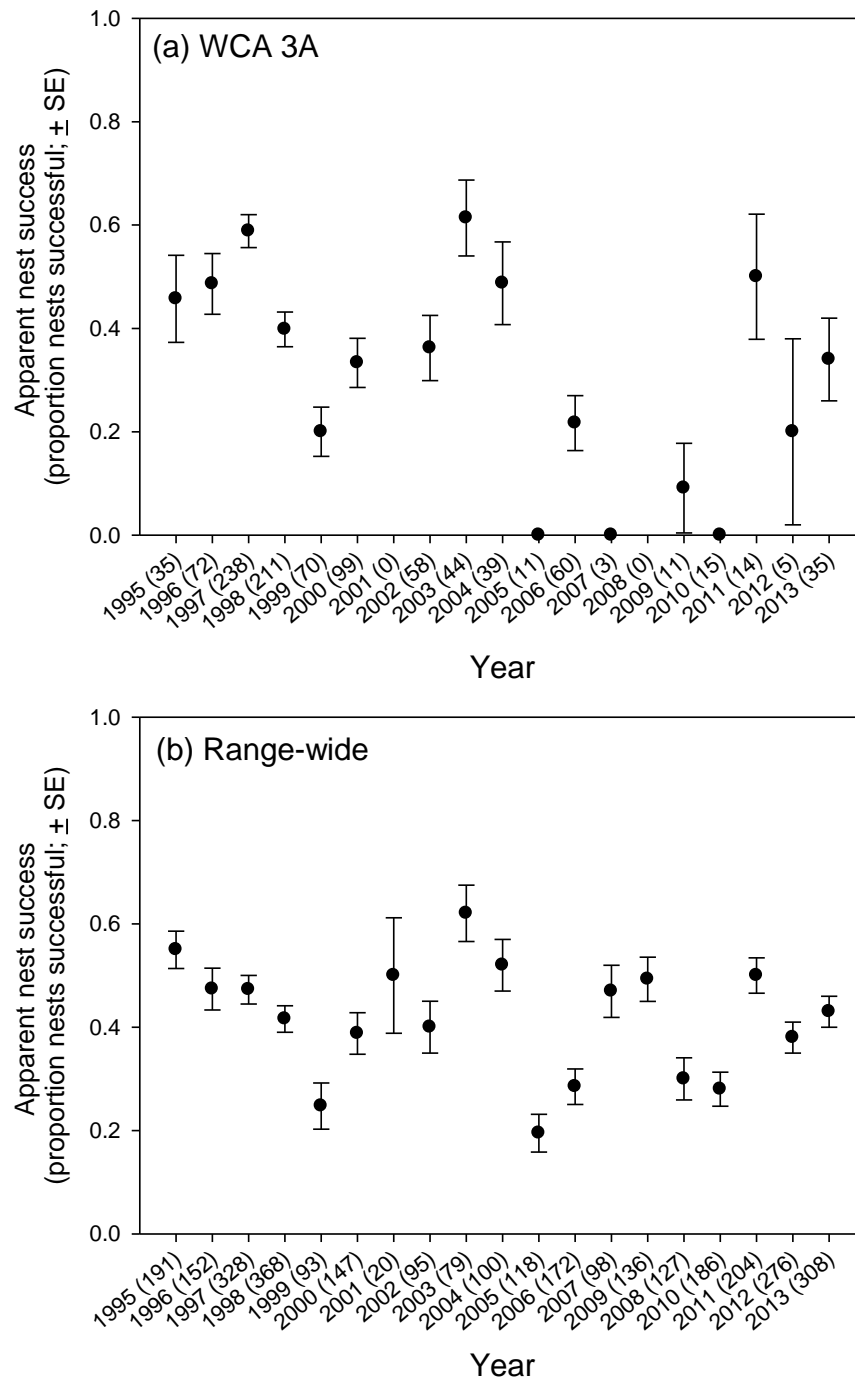


Figure 10. Mean (\pm 95% CI) number of young fledged per successful nest in select wetlands during 2013. Note small sample sizes for many wetlands.

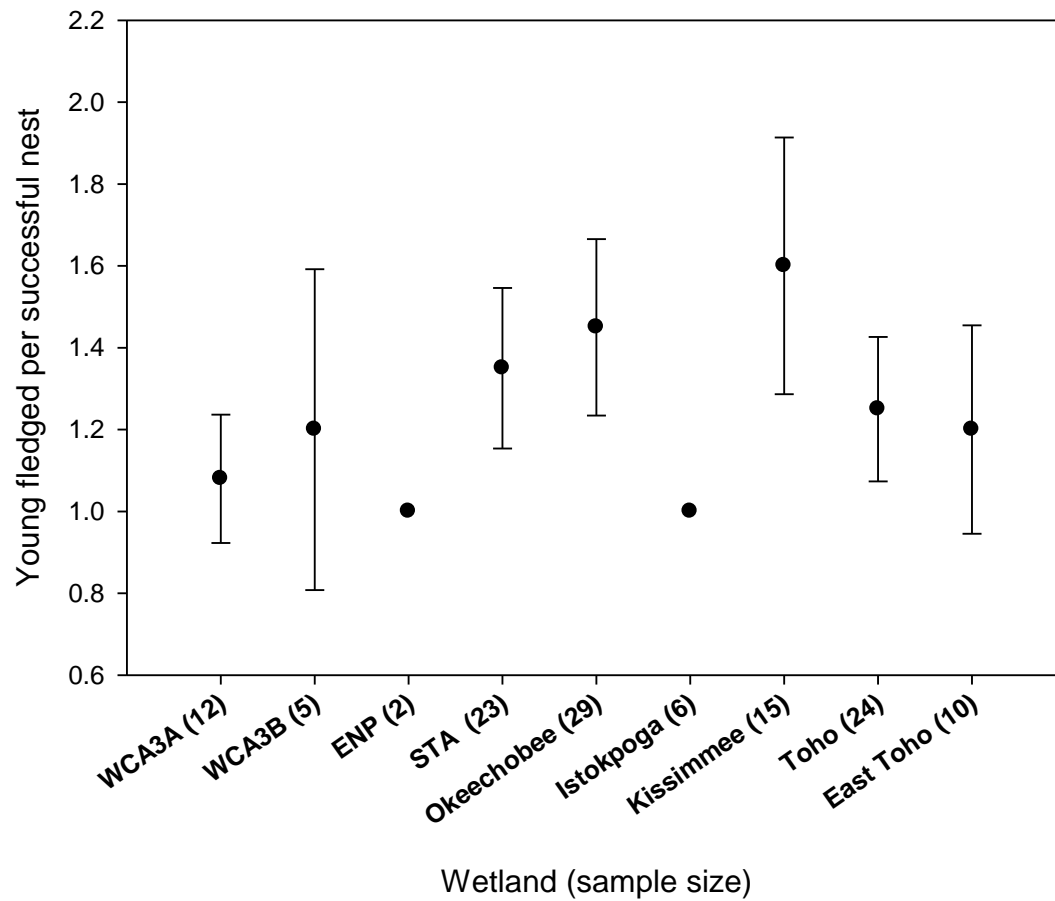
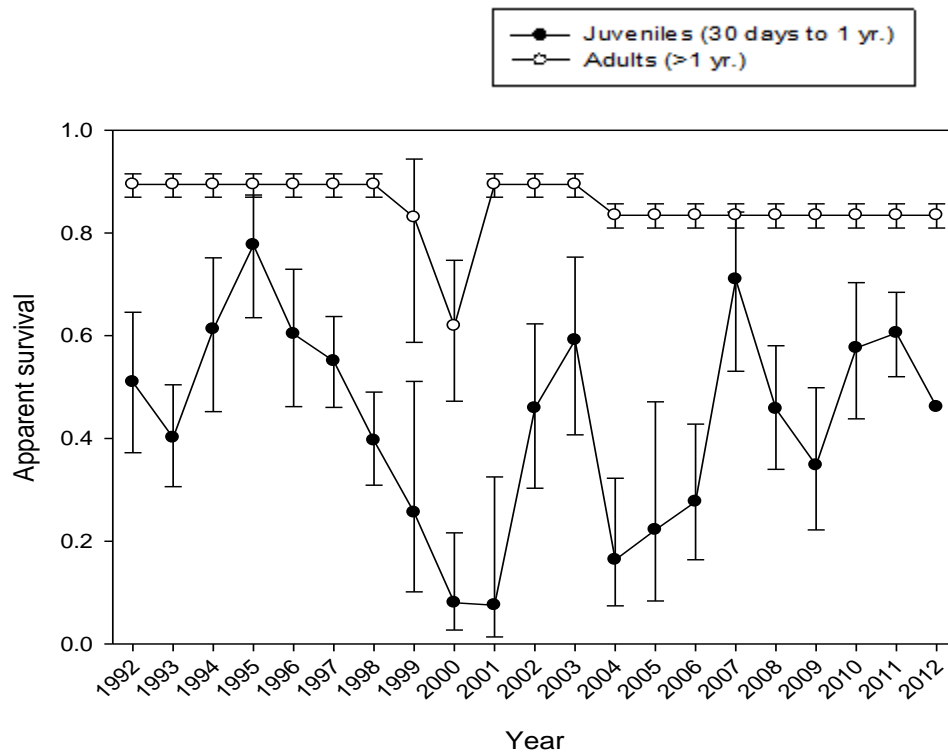


Figure 11. (a) Model-averaged estimates of adult (white circles) and juvenile (black circles) apparent annual survival, 1992-2012; (b) Annual detection probability relevant to apparent survival estimates, 1992-2012. Error bars correspond to 95% confidence intervals.

(a).



(b).

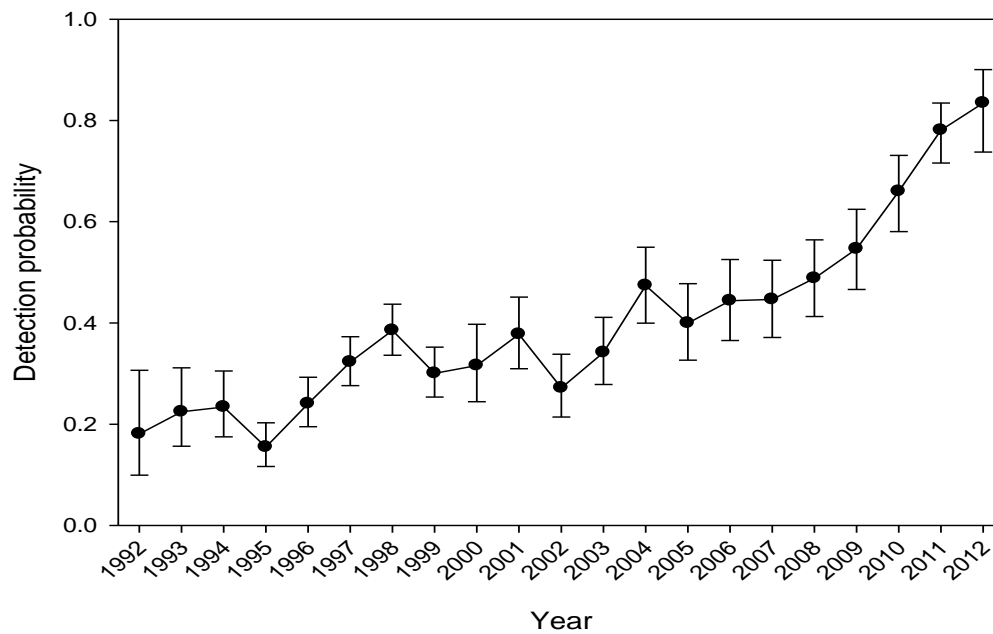


Figure 12. Population size of snail kites, 1997-2013, estimated using the super-population approach (Dreitz et al. 2002; Martin et al. 2007c). Black dots (and error bars) show population size estimates for each year (and 95% confidence intervals); note that each year is estimated independently (i.e., only information from that year is used in calculations). The black line shows the 3-year running average and gray shaded region shows the uncertainty around the 3-year running average (95% CI, taken from parametric bootstrapping).

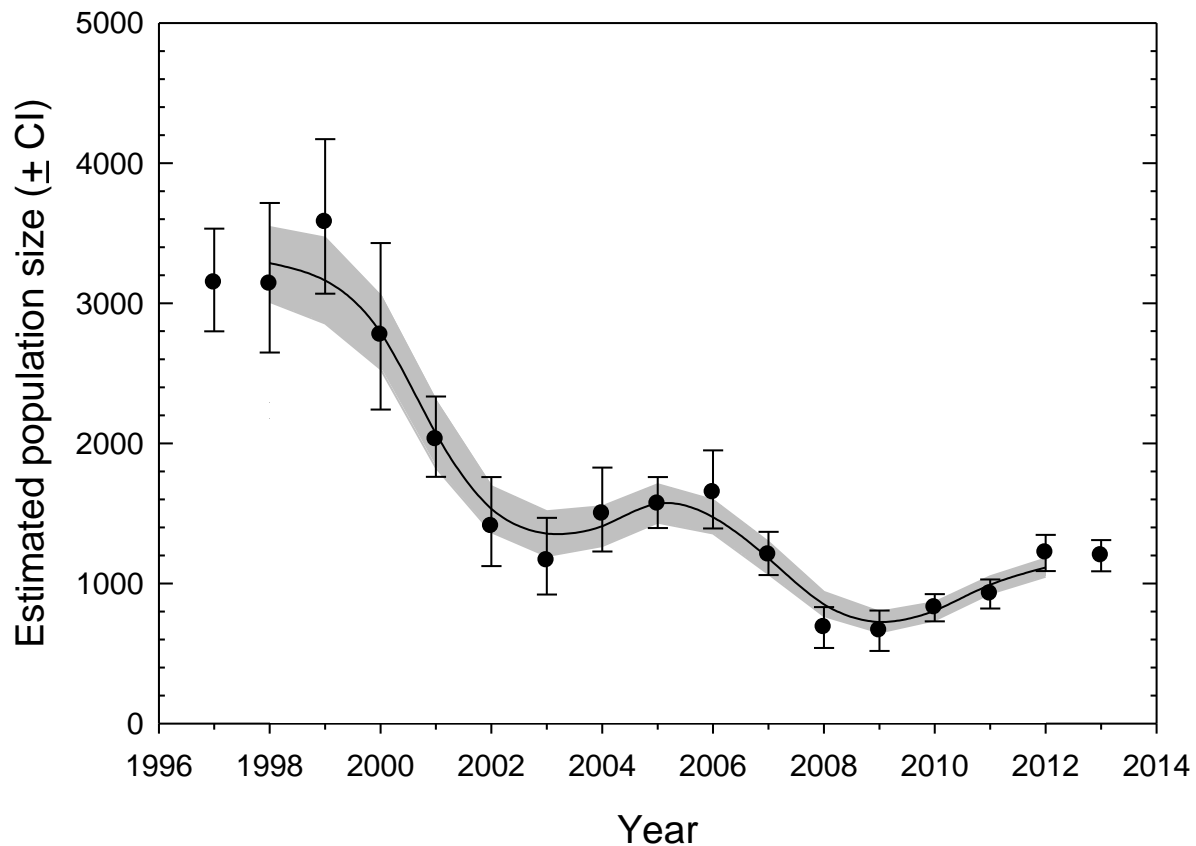


Figure 13. Population growth rates (finite rate of increase, N_t/N_{t-1}) of snail kites, 1997-2013, derived from the super-population estimates (shown in Figure 7). The black line shows the 3-year running average and gray shaded region shows the uncertainty (95% CI, taken from parametric bootstrapping) around the 3-year running average. Uncertainty around estimates is preliminary—we are currently assessing the utility of this approach to uncertainty estimation.

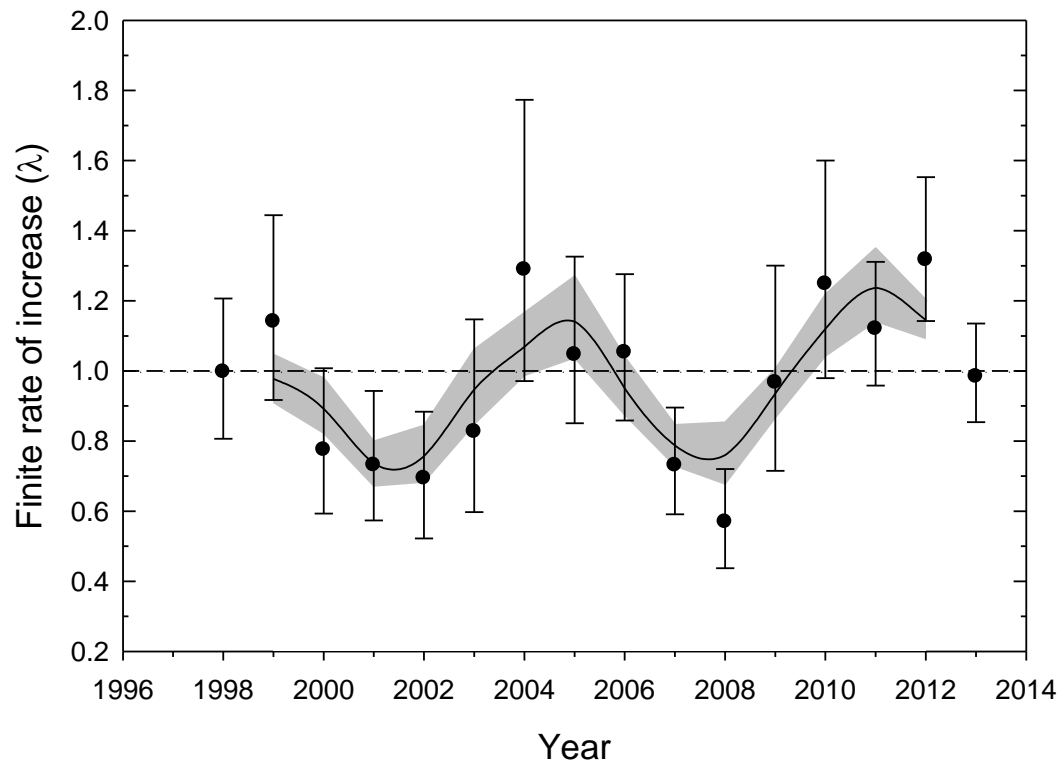


Figure 14. Simulation results (mean \pm 2 SD) comparing the utility of the superpopulation estimator and using the maximum count under various scenarios. (a) Comparisons under a range of true population sizes ($N = 300 - 5000$), with proportion of marked birds, observed detection probabilities, and time spent outside sample set to realistic values from field monitoring. (b) Comparisons for regarding the proportion of the population banded, when population size is 900 individuals. (c) Comparisons regarding the amount of time spent outside the sample, when the population size is 900 individuals.

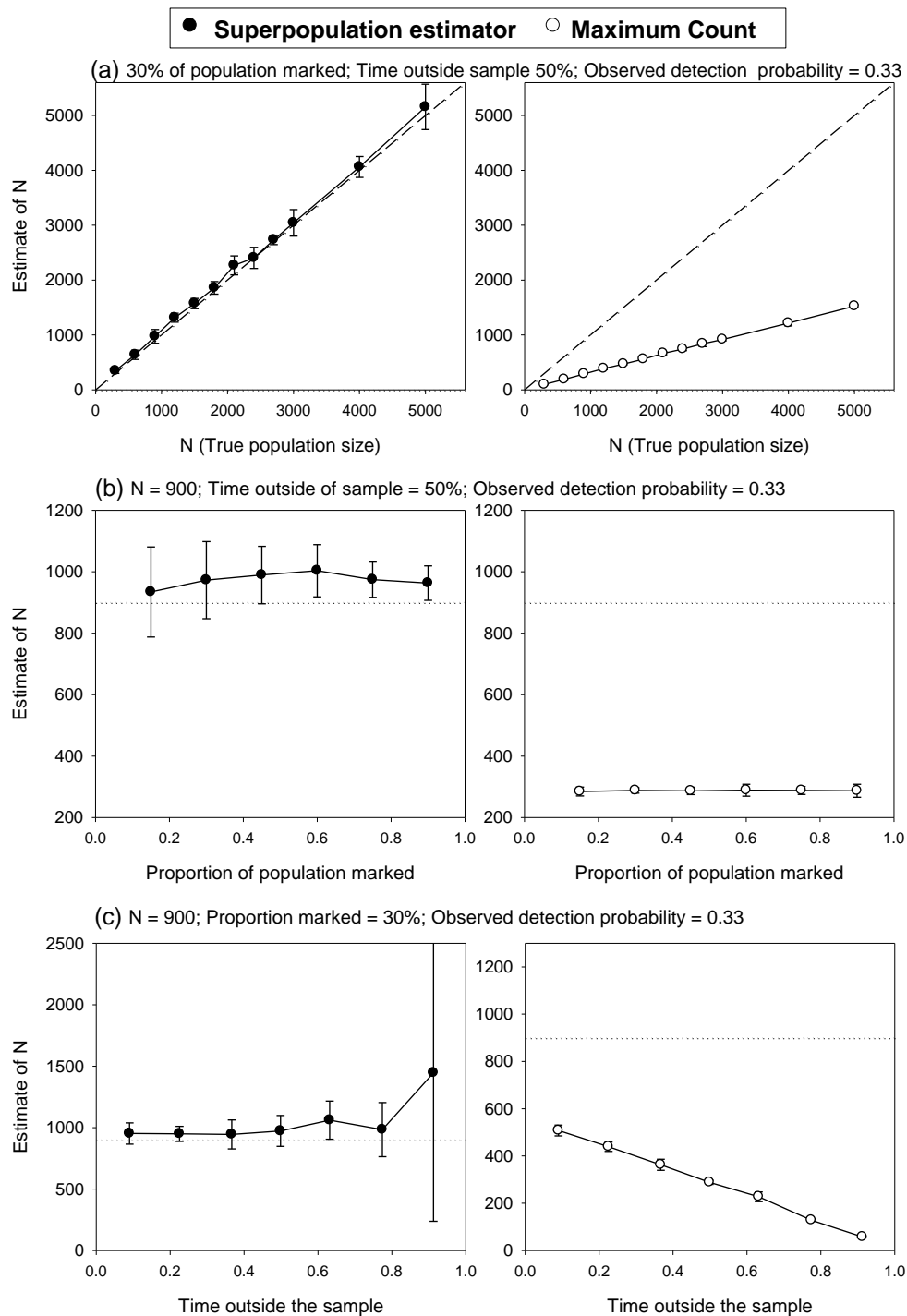


Figure 15. Simulation results for conditions most sensitive to estimates. (a) Comparisons under a range of true population sizes ($N = 300 - 5000$), with changes in the amount of time kites spend outside the study sites. (b) Similar to (a), but focusing on situations where observed detection probability is very low ($p \leq 0.10$).

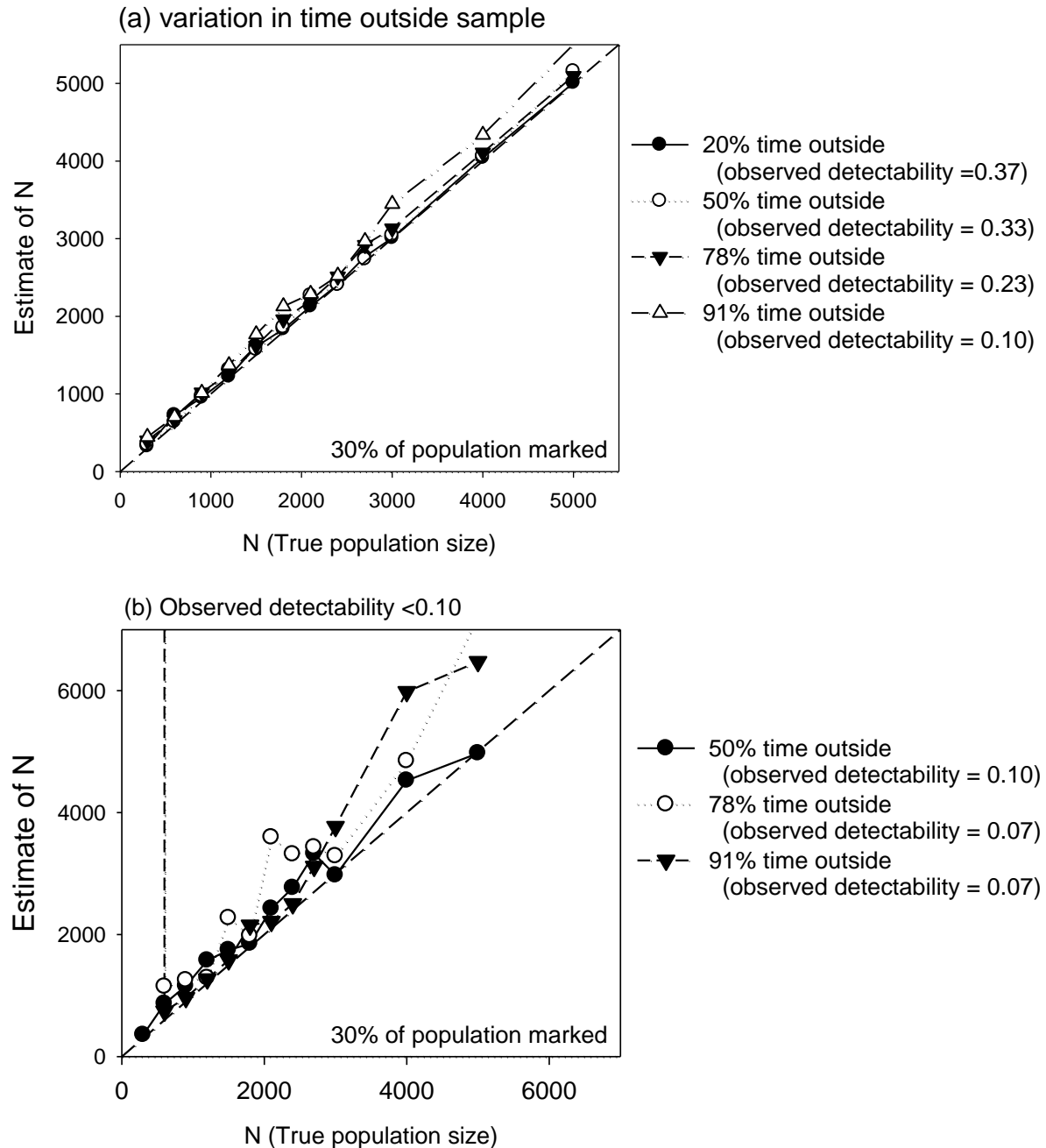


Figure 16. Mean (\pm SE) number of young fledged per successful nest in WCA3A, 1995-2013. No successful nests were observed in WCA3A in 2005, 2007, 2008 or 2010. Only one successful nest was observed in 2009 and 2012 (white circles). All other years (black circles) had sample sizes greater than six. Model predicted mean (solid line) and standard error (dashed lines) represent results from a simple linear regression of mean number of young fledged per successful nest as a function of time.

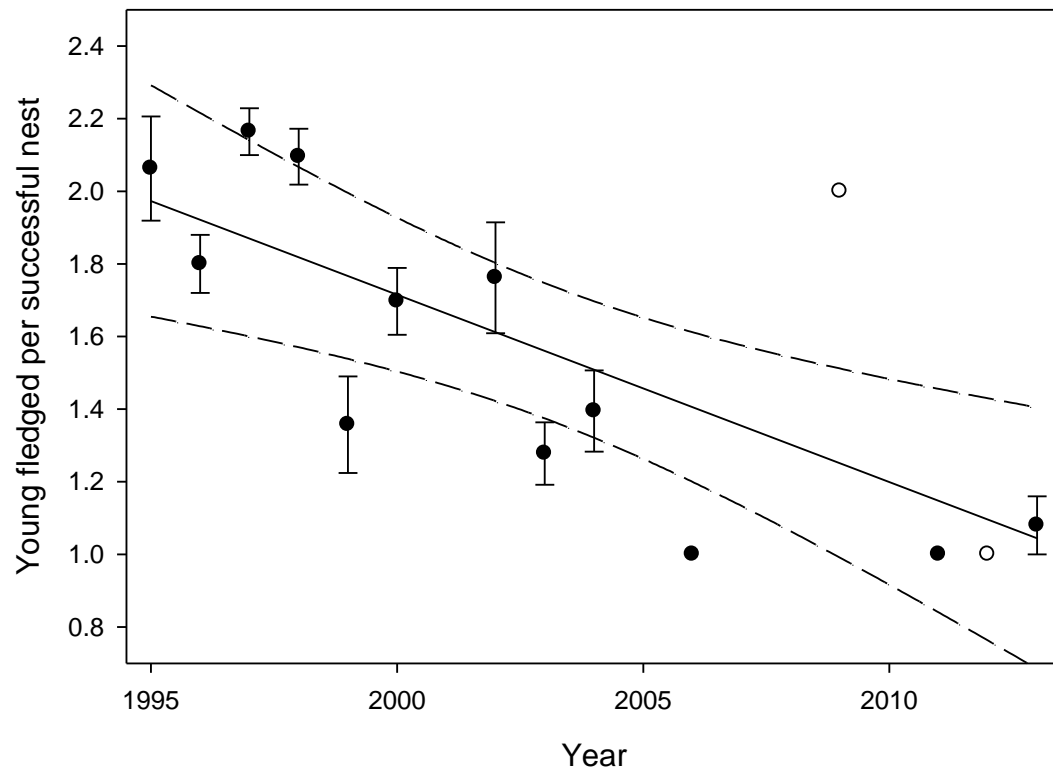


Figure 17. Proportion of collected snails that were exotic relative to the distance of collection site from a canal in Water Conservation Area 3A. All snail shells were collected at snail kite nests and under observed extraction perches (n = 162 sites within WCA3A; 503 snail shells collected). Lines based on a logistic regression model, pooling data collected from 2011-2013.

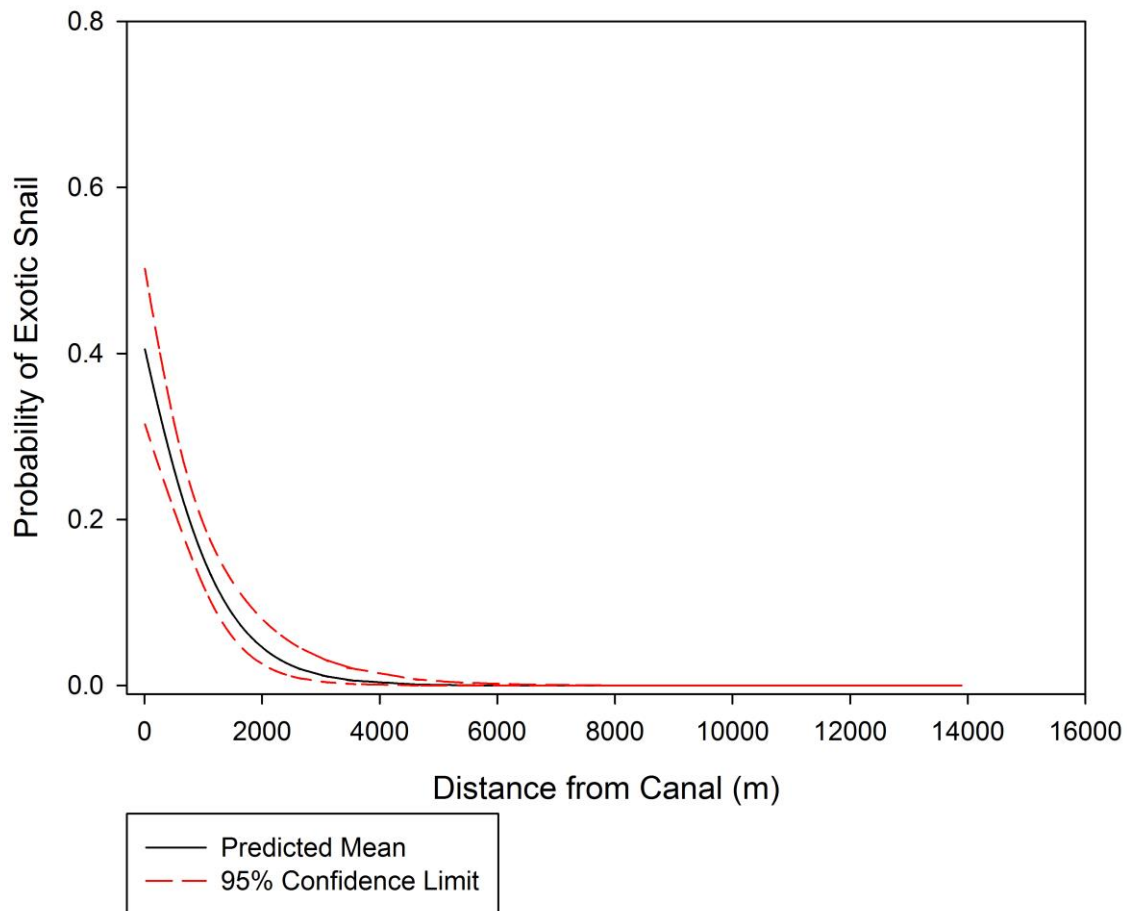
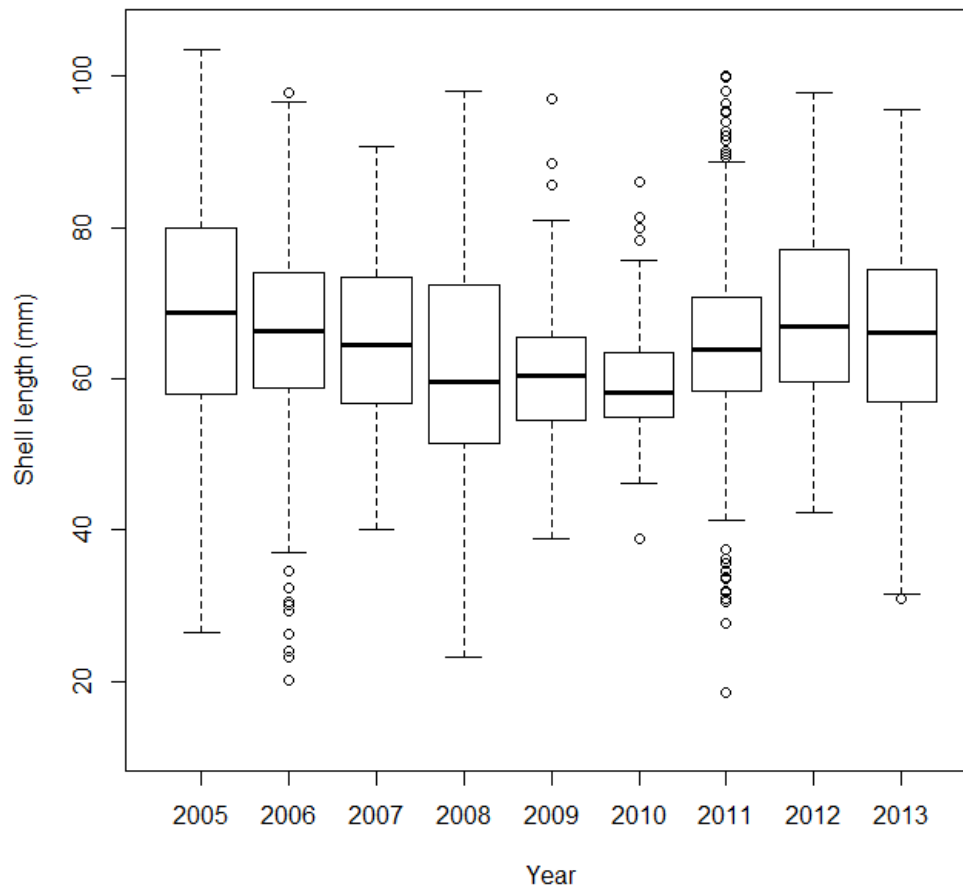


Figure 18. Median shell size of exotic snails consumed by snail kites on Toho, 2005-2013. Shells were collected under feeding perches and at nests.



Appendices

Appendix A. Update on habitat monitoring for the snail kite.

Appendix B. Publications and theses supported by this work from 2010-2014 (5-year period).

Appendix A

Habitat monitoring for the Snail Kite

Introduction

The Florida Snail Kite (*Rostrhamus sociabilis plumbeus*) 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 (Bennetts 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 (Figure S1) 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 (Figure S1) 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 m² 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 2007-2013. These were sorted by species, counted, dried to a constant weight, and weighed to the nearest 0.1 g. Approximately 14,600 samples were collected and processed between 2002 and 2013. 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 $i = ((R_{di} + R_{bi})/2) * 100$,

where R_{di} is the relative density of species i and R_{bi} 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 m² 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

We modeled count data per 0.25 m² sample for only prairie/sloughs, by species, with a negative binomial generalized linear model in R (glm.nb in MASS): Density ~ Year + Community (Prairie or Slough). Only emergent species (*P. geminatum*, *P. hemitomom*, *E. cellulosa*, and *E. elongata*) and *N. odorata* were modeled as they are important habitat species or

indicators of habitat change.

Update to Multistate Model from Zweig and Kitchens (2014)

We used the model parameters from Zweig and Kitchens (2014) and current (2010-2013) hydrologic variables to update the ratio of *E. cellulosa*, *E. elongata*, and *N. odorata* dominated communities across the landscape of 3A South. For detailed methods, please see the attached paper (Zweig and Kitchens 2014).

Results

Multivariate Analysis

The cluster/ISA indicated 11 communities within the study area and they were named by the indicator species for that community or if there wasn't a strong indicator, they were named as a community by the species present (Figure S2): *Eleocharis cellulosa*, *Eleocharis elongata*, *Pontideria cordata*, *E. cellulosa*/sparse *C. jamaicense* prairie, *E. elongata*/sparse *C. jamaicense* prairie, ghost island emergent edge, *Cladium jamaicense*, shrub island (*Blechnum serrulatum*/*Osmunda regalis*), emergent slough, *Bacopa caroliniana* transitional, and a *Nymphaea odorata* slough. The three communities that are important to kite foraging in 3A South and 3B, *E. cellulosa*, *E. cellulosa*/Cladium prairie, and Bacopa transitional all occur in this analysis. *E. cellulosa* community only occurred in the western side of the study area and has disappeared 4 of 5 plots with continuous sampling between 2009-2011. The Bacopa transitional community also only occurs in the western portion of the study area and is stable in two of the continuous sampling plots. Between 2012 and 2013, it appeared in 4 other plots, one in the southwest and three in the northwest. The *E. cellulosa*/Cladium community has appeared intermittently across the landscape, but dominantly in the western section of 3A South. It is currently only present in one plot in the southwest quadrant.

Univariate Analysis

All modeled species densities were significantly affected by year (2002-2013) and community (prairie or slough) ($p < 0.05$). Stem density of all emergents decreased significantly over time (Figures S3-S6). Stem density of *N. odorata* increased significantly over time (Figure S7).

Update to Multistate Model from Zweig and Kitchens (2014)

The decreasing trend of the *E. cellulosa* community continues from 2010-2013, indicating the decline of important foraging habitat for the Snail Kite (Figure S8). Significant increases in the *N. odorata* community and decreases in *E. elongata* community also continue (Figures S9), with a slight decrease of *N. odorata* communities in 2011 due to the dry conditions.

Discussion

Previous multivariate analyses have suggested 10 communities (Zweig and Kitchens 2008), but increased amounts of data allow for more communities to be delineated. In 2011, there was a new sparse sawgrass community that was ubiquitous across the landscape and by including the 2012-2013 data, there is another change in communities. The *P. geminatum*/*P. hemitomom* community from previous multivariate analyses is missing and there is an additional sparse sawgrass community, but dominated by *E. cellulosa*. The other 9 communities from the cluster/ISA analysis correspond to previous communities from Zweig and Kitchens (2008). The presence of drier communities in the current analysis likely reflects the dry down in 2011 where the minimum water level at Site 65, in the impounded area, measured -14 cm. These environmental conditions favor the emergence of new sawgrass growth and invasion into prairies and sloughs.

The univariate and multistate models results suggest, even with a dry-down year, that the foraging habitat within WCA3AS is still 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 Zweig and Kitchens (2014) (included in report) 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 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, multistate, 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 S1: Study plots in Water Conservation Area 3A South. This analysis concentrated on plots 7, 8, and 9, but data is being collected landscape-wide.

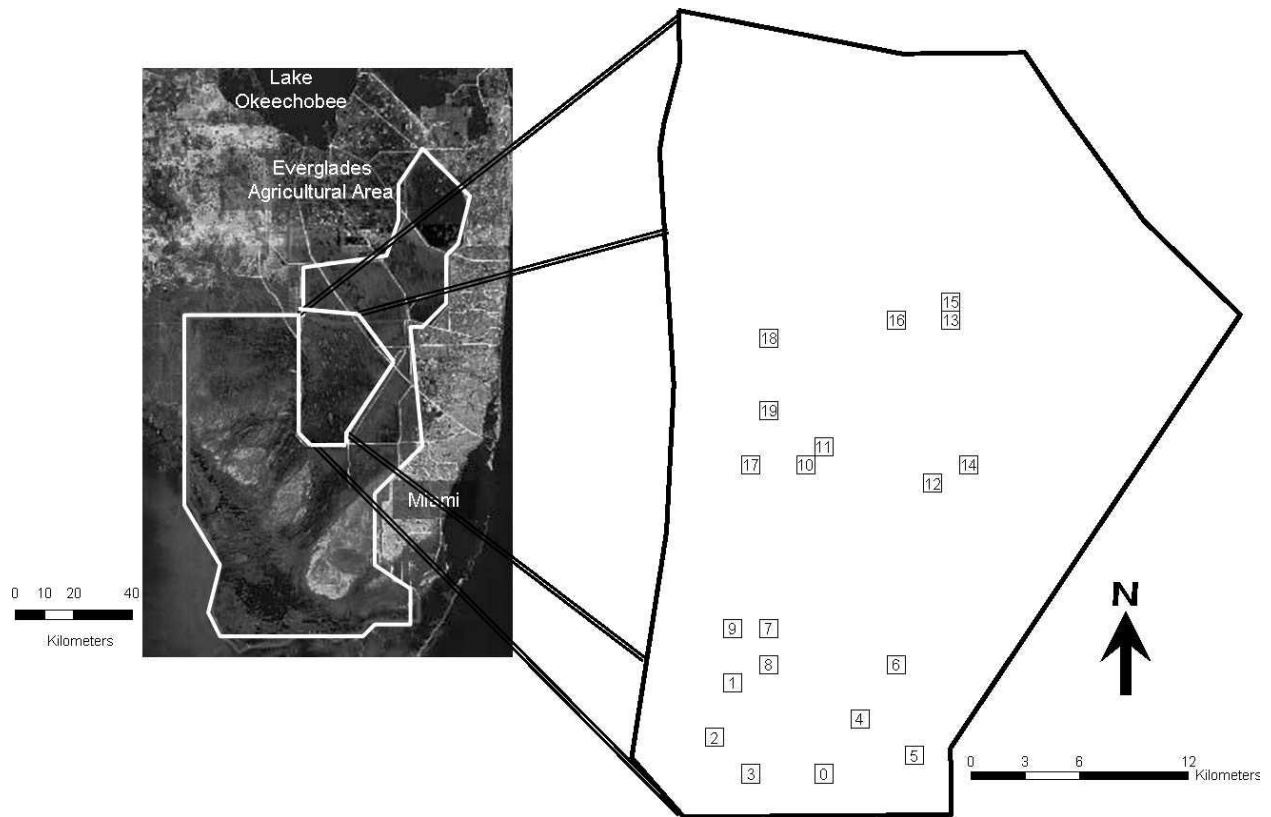


Figure S2: Non-metric multidimensional scaling ordination with vegetation data in Water Conservation Area 3A South. This graph demonstrates how vegetation communities are influenced by hydrologic and peat variables.

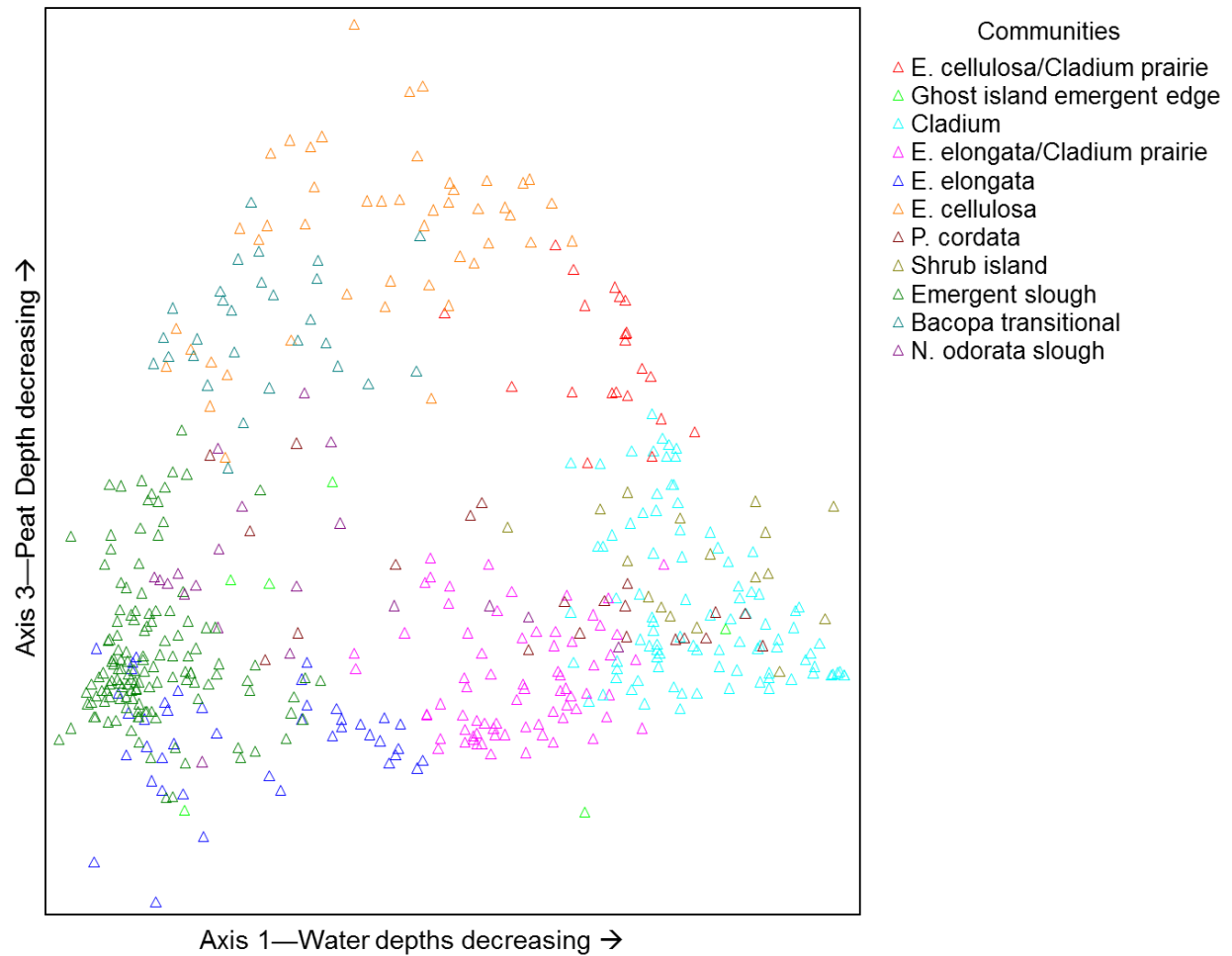


Figure S3: Modeled density estimates per 0.25-m², with standard errors, of *Panicum hemitomon* within our study area in Water Conservation Area 3A South.

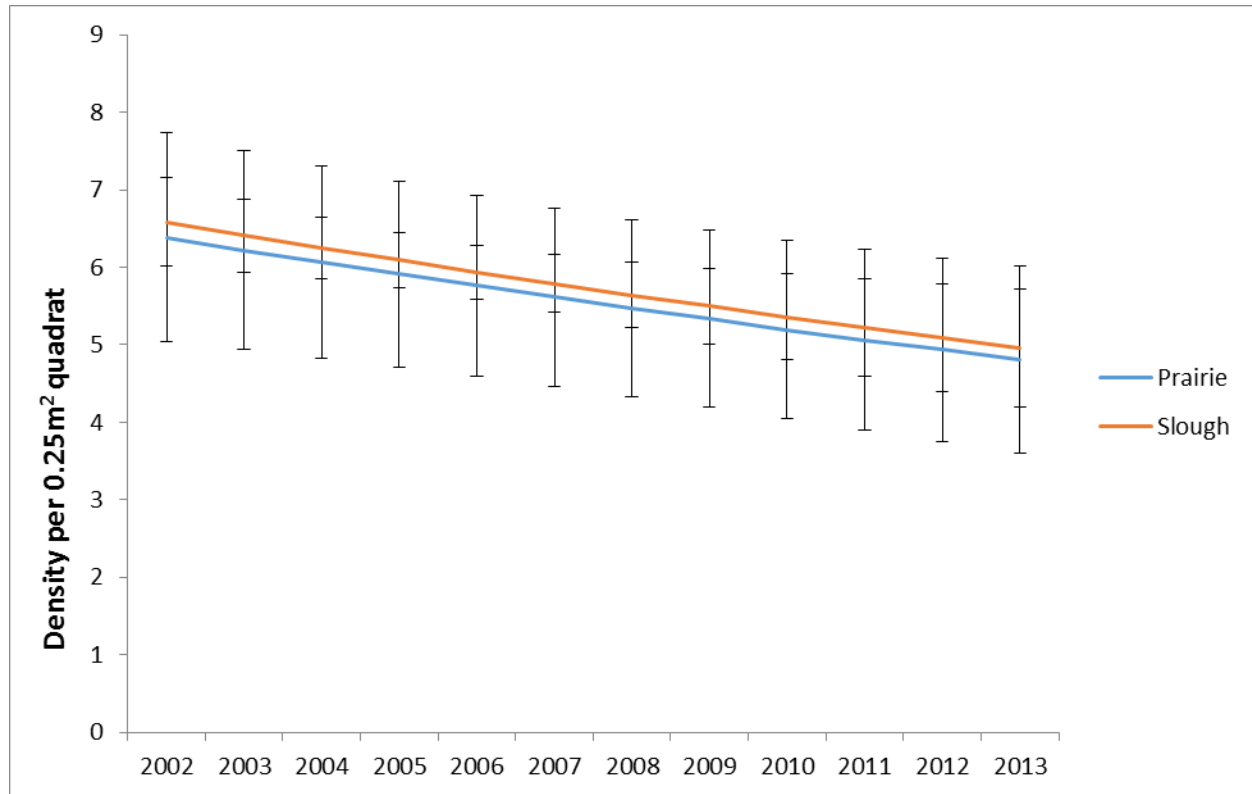


Figure S4: Modeled density estimates per 0.25-m², with standard errors, of *Paspalidum geminatum* within our study area in Water Conservation Area 3A South.

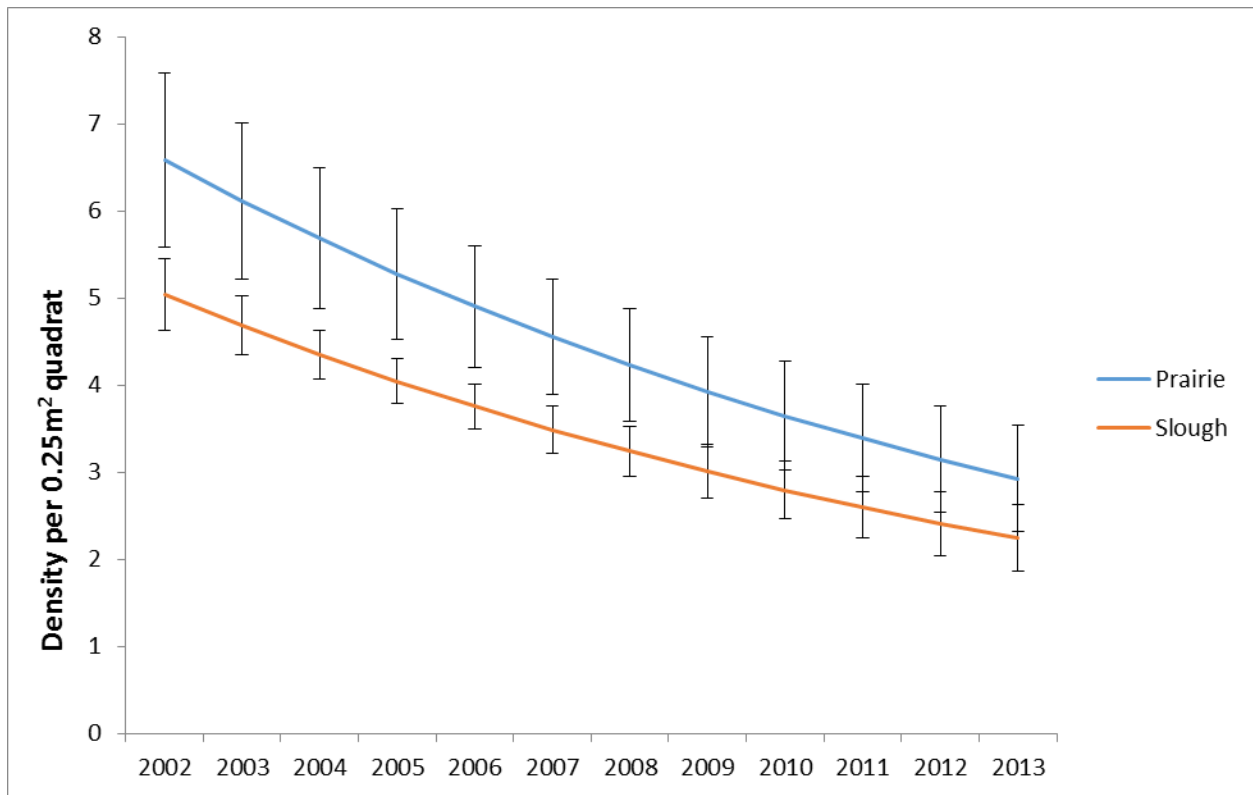


Figure S5: Modeled density estimates per 0.25-m², with standard errors, of *Eleocharis elongata* within our study area in Water Conservation Area 3A South.

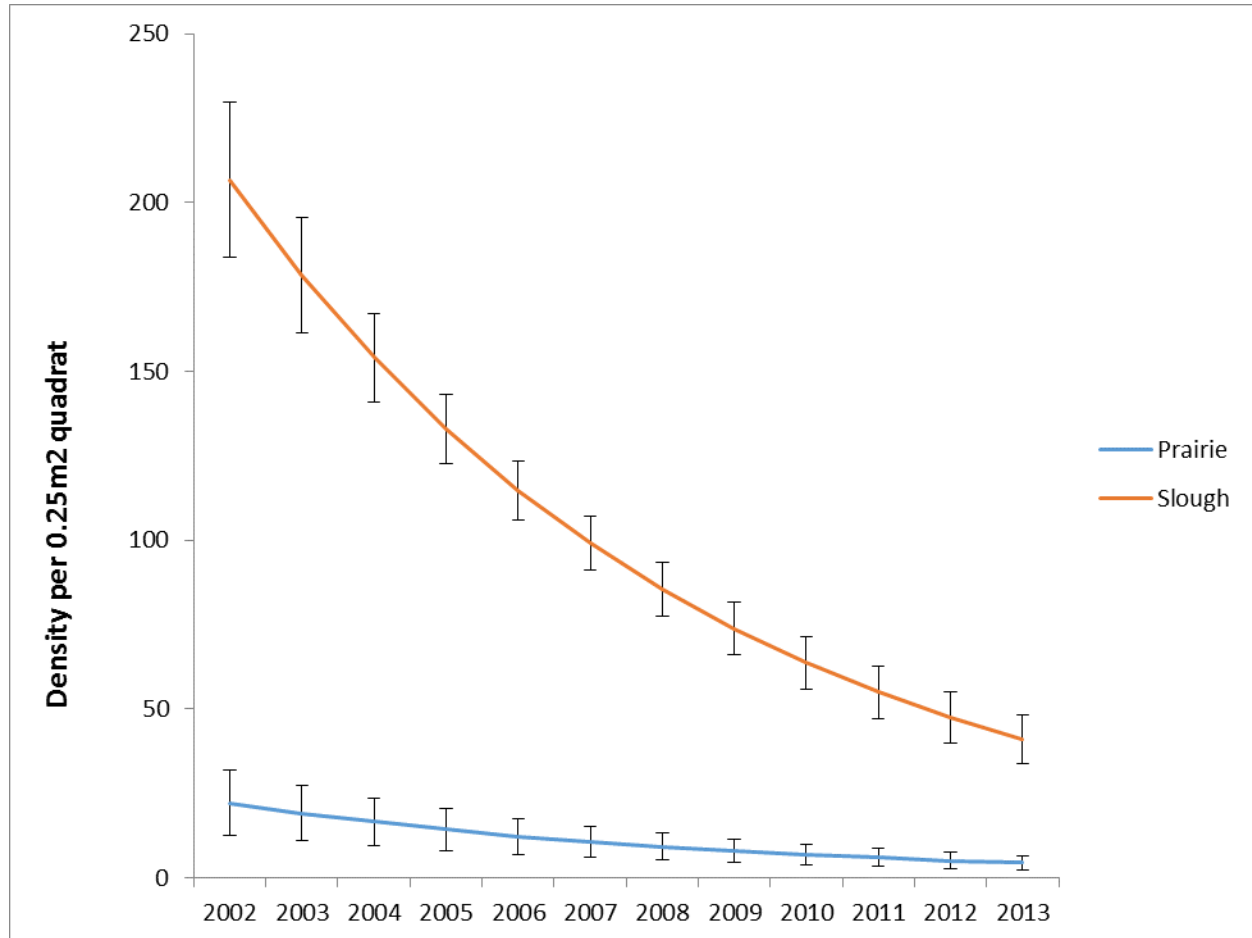


Figure S6: Modeled density estimates per 0.25-m², with standard errors, of *Eleocharis cellulosa* within our study area in Water Conservation Area 3A South.

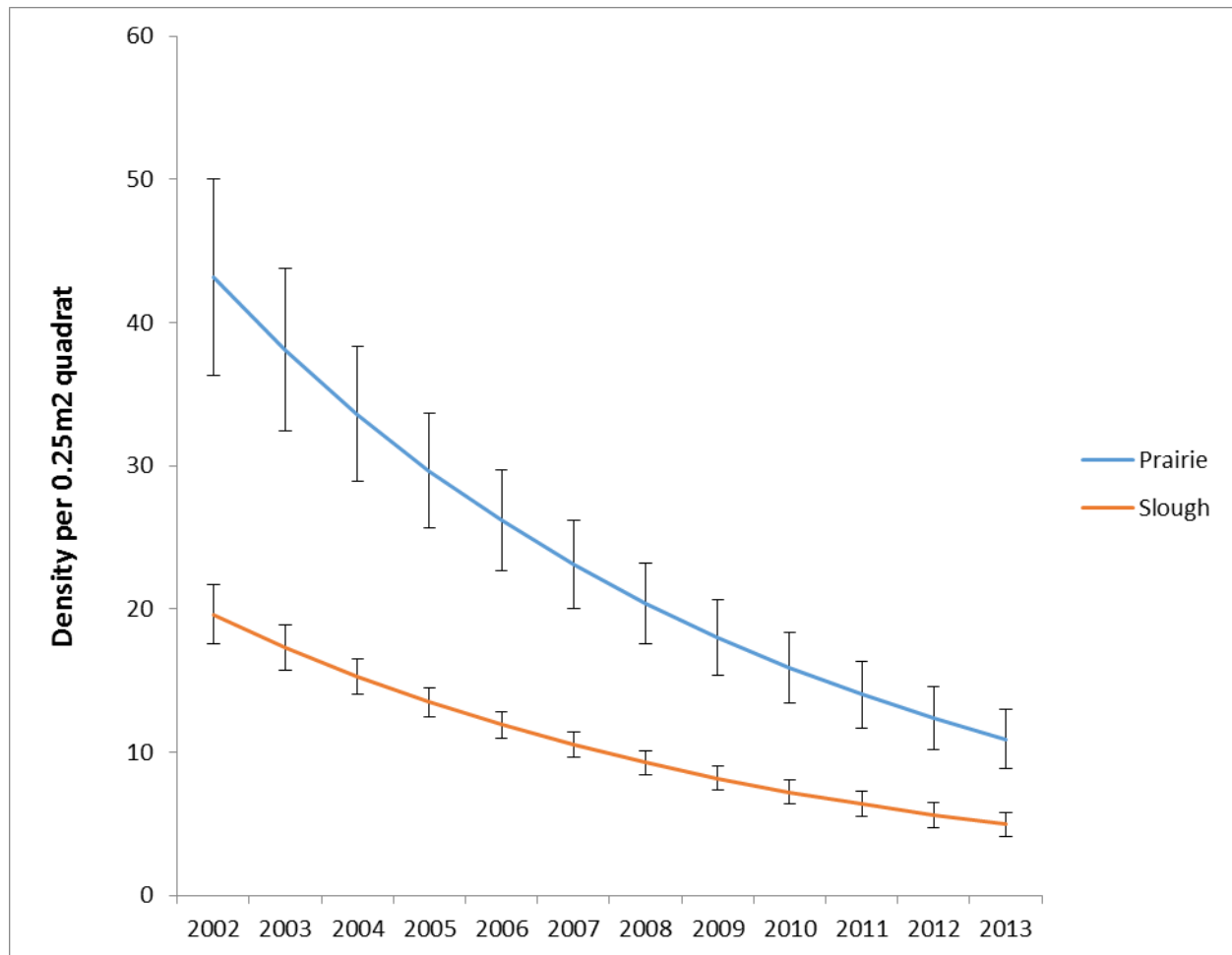


Figure S7: Modeled density estimates per 0.25-m², with standard errors, of *Nymphaea odorata* within our study area in Water Conservation Area 3A South.

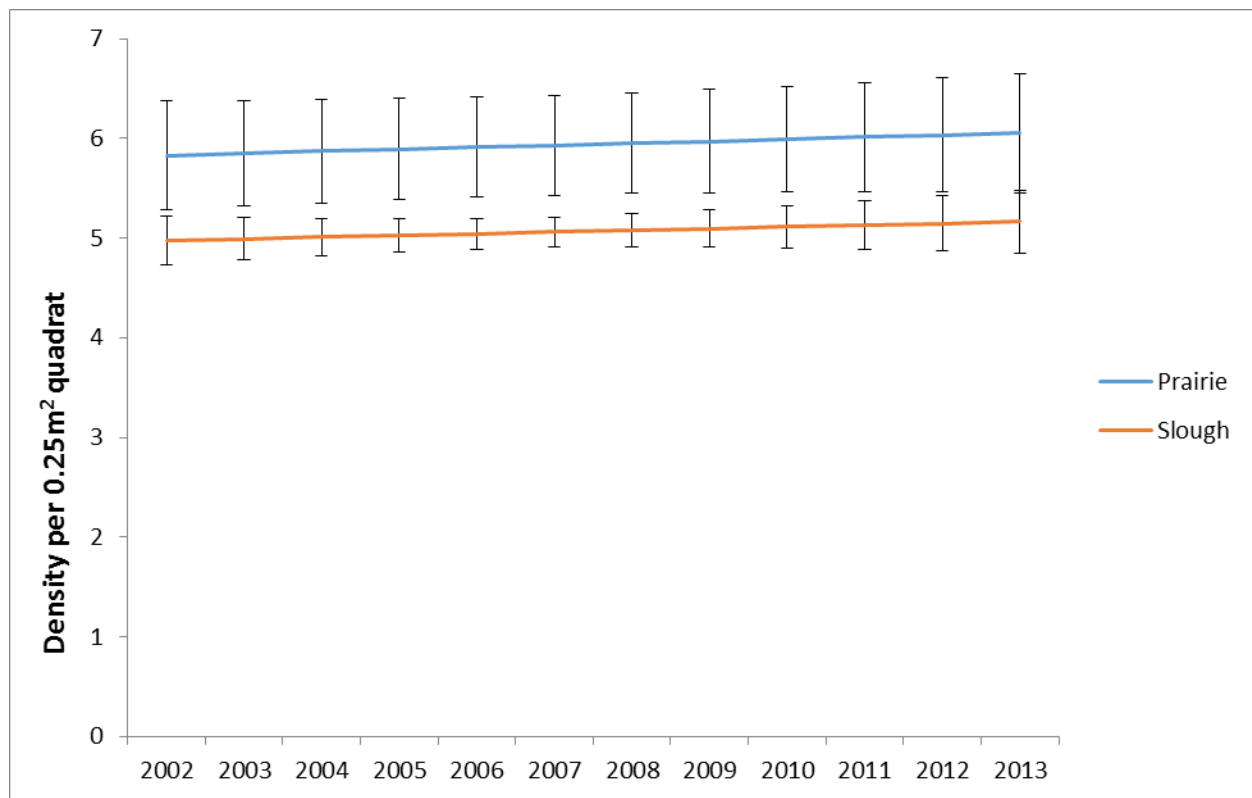


Figure S8: Update (2010-2013) to prairie/slough multistate model in Zweig and Kitchens (2014). Proportion of sites in each community state in Water Conservation Area 3A South from 1996-2013. Pre-2002 was modeled with an artificial neural network and 2006-2013 was forecast with a multistate model.

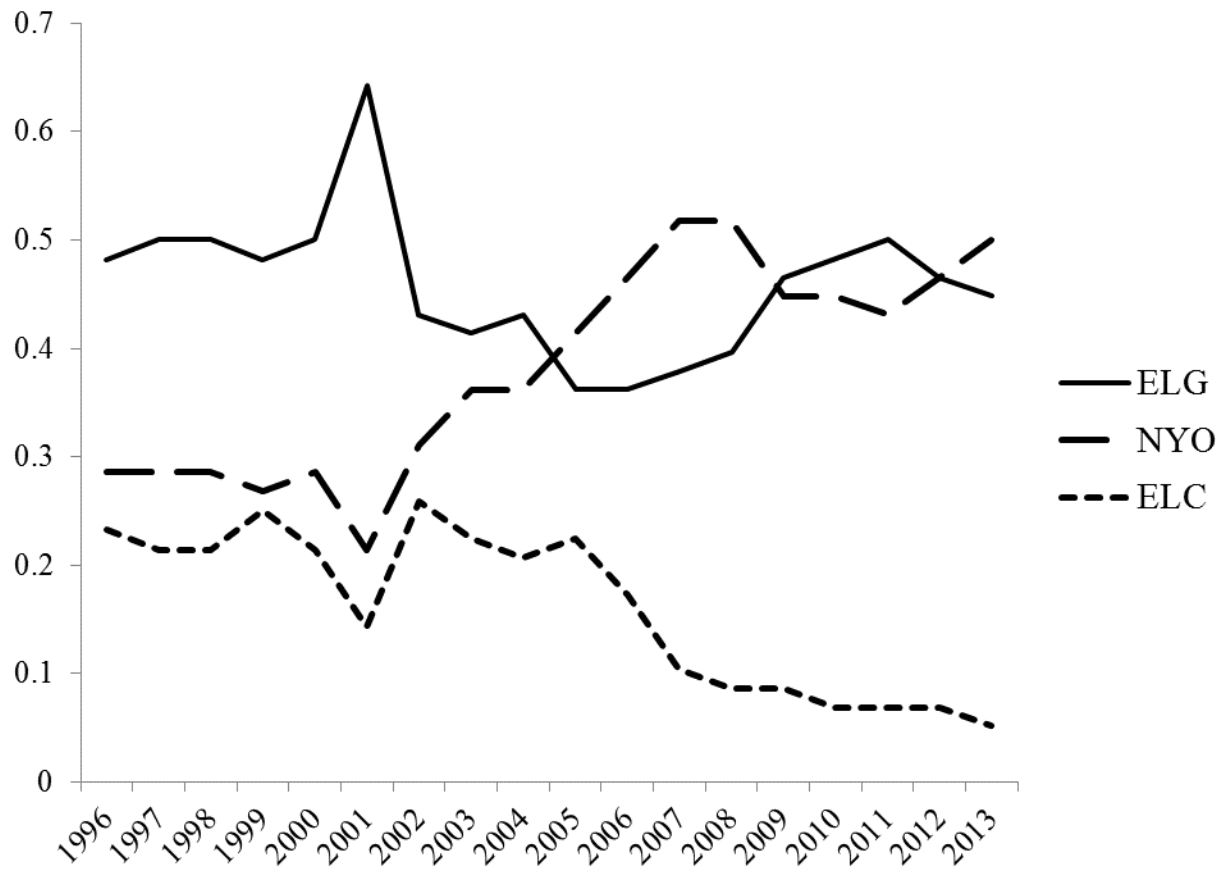
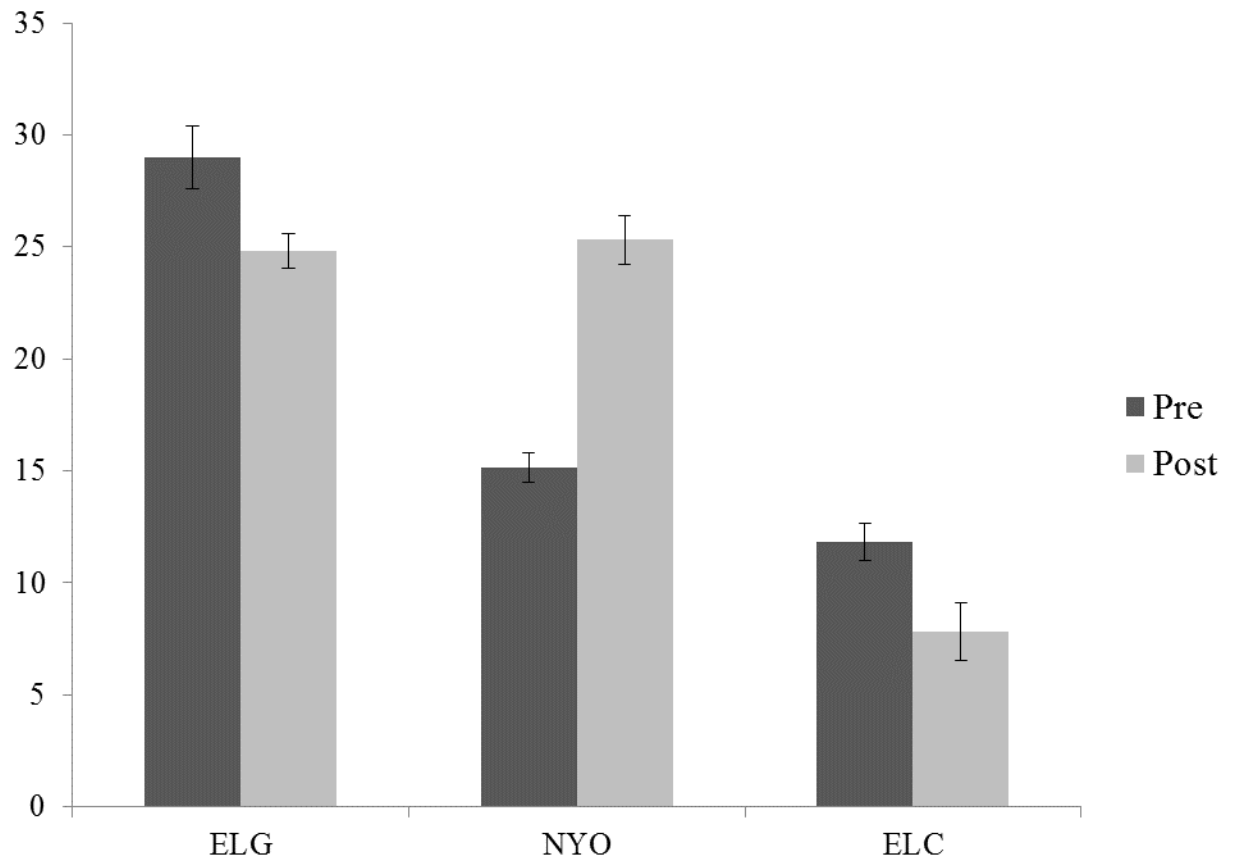


Figure S9: Update to comparison of number of sites in each community state in Water Conservation Area 3A South before (1996-2001) and after Snail Kite population decline (2002-2013). Significant differences only occur for the ELG and NYO community. ELG = *Eleocharis elongata*, ELC = *E. cellulosa*, NYO = *Nymphaea odorata*.



Appendix B

Publications and theses supported by this work from 2010-2014 (5-year period)

Publications:

- Bowling, A. C., J. Martin, and W. M. Kitchens. 2012. The effect of changes in habitat conditions on the movement of juvenile Snail Kites (*Rostrhamus sociabilis plumbeus*). *Ibis* 154(3):554-565.
- Cattau, C. E., P. Darby, R. J. Fletcher, and W. M. Kitchens. 2014. Reproductive responses of the endangered Snail Kite to variations in prey density. *Journal of Wildlife Management*. *In press*.
- Cattau, C. E., J. Martin, and W. M. Kitchens. 2010. Effects of an exotic prey species on a native specialist: Example of the Snail Kite. *Biological Conservation* 143 (2):513-520.
- Fletcher, R. J., M. A. Acevedo, B. E. Reichert, K. E. Pias, and W.M. Kitchens. 2011. Social network models predict movement and connectivity in ecological landscapes. *Proceedings of the National Academy of Science of the United States of America* 108(48):19282-19287.
- Fletcher, R. J., A. Revell, B. E. Reichert, W. M. Kitchens, J. D. Dixon, and J. D. Austin. 2013. Network modularity reveals critical scales for connectivity in ecology and evolution. *Nature Communications* 4:2572.
- Pias, K. E., Z. C. Welch, and W. M. Kitchens. 2012. An artificial perch to help Snail Kites handle an exotic apple snail. *Waterbirds* 35(2): 347-351.
- Reichert, B., C. Cattau, R. J. Fletcher, W. Kendall, and W. Kitchens. 2012. Extreme weather and experience influence reproduction in an endangered bird. *Ecology* 93:2580-2589.
- Reichert, B., J. Martin, W. Kendall, C. Cattau, and W. Kitchens. 2010. Interactive effects of senescence and natural disturbance on the survival probabilities of Snail Kites. *Oikos*. 119 (6): 972-979.
- Zweig, C. L. and W. M. Kitchens. 2014. Reconstructing historical habitat data with predictive models. *Ecological Applications* 24:196-203.
- Zweig, C. L., B. E. Reichert, and W. M. Kitchens. 2011. Implications of discontinuous elevation gradients on fragmentation and restoration in patterned wetlands. *Ecosphere* 2(8):art98.

Thesis/Dissertations

- Drzid, Lara. 2011. Aspects of the Abundance, Density, and Movement of Apple Snails Relative to the Invasive Submerged Aquatic Plants in a Central Florida Lake. Thesis. University of Florida
- Olbert, Jean. 2013. The breeding ecology of endangered Snail Kites (*Rostrhamus sociabilis plumbeus*) on primary nesting site in Central Florida, USA. Thesis. University of Florida.
- Pias, Kyle. 2012. Foraging ecology of breeding Snail Kites (*Rostrhamus sociabilis plumbeus*) on Lake Tohopekaliga, Florida, USA. Thesis. University of Florida.