
Importance of Well-Designed Monitoring Programs for the Conservation of Endangered Species: Case Study of the Snail Kite

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Abstract: *Monitoring natural populations is often a necessary step to establish the conservation status of species and to help improve management decisions. Nevertheless, many monitoring programs do not effectively address primary sources of variability in monitoring data, which ultimately may limit the utility of monitoring in identifying declines and improving management. To illustrate the importance of taking into account detectability and spatial variation, we used a recently proposed estimator of abundance (superpopulation estimator) to estimate population size of and number of young produced by the Snail Kite (*Rostrhamus sociabilis plumbeus*) in Florida. During the last decade, primary recovery targets set by the U.S. Fish and Wildlife Service for the Snail Kite that were based on deficient monitoring programs (i.e., uncorrected counts) were close to being met (by simply increasing search effort during count surveys). During that same period, the Snail Kite population declined dramatically (by 55% from 1997 to 2005) and the number of young decreased by 70% between 1992–1998 and 1999–2005. Our results provide a strong practical case in favor of the argument that investing a sufficient amount of time and resources into designing and implementing monitoring programs that carefully address detectability and spatial variation is critical for the conservation of endangered species.*

Keywords: capture-recapture models, population decline, *Rostrhamus sociabilis plumbeus*, superpopulation approach

Importancia de Programas de Monitoreo para la Conservación de Especies en Peligro Bien Diseñados: Caso de Estudio de *Rostrhamus sociabilis plumbeus*

Resumen: *A menudo, el monitoreo de poblaciones naturales es un paso necesario para establecer el estatus de conservación de las especies y ayudar a mejorar las decisiones de manejo. Sin embargo, muchos programas de monitoreo no atienden las principales fuentes de variabilidad en los datos de monitoreo, que a fin de cuentas pueden limitar la utilidad del monitoreo en la identificación de declinaciones y el mejoramiento del manejo. Para ilustrar la importancia de considerar la variabilidad espacial y de la detectabilidad, utilizamos un estimador propuesto recientemente (estimador de superpoblación) para estimar el tamaño poblacional y número de juveniles producidos por *Rostrhamus sociabilis plumbeus* en Florida. Durante la década pasada, los objetivos de recuperación primarios para *R. s. plumbeus* definidos por el Servicio de Pesca y Vida Silvestre de E.U.A., que fueron basados en programas de monitoreo deficientes (i.e., conteos no corregidos), casi fueron cumplidos (simplemente mediante el incremento del esfuerzo de búsqueda durante los conteos). Durante el mismo período, la población de *R. s. plumbeus* declinó dramáticamente (55% de 1997 a 2005) y el número de juveniles declinó 70% entre 1992–1998 y 1999–2005. Nuestros resultados son un fuerte caso práctico a favor del argumento de que la inversión de la suficiente cantidad de tiempo y recursos en el diseño e implementación*

de programas de monitoreo que consideran la variación espacial y de la detectabilidad es crítica para la conservación de especies en peligro.

Palabras Clave: declinación poblacional, método de superpoblación, modelos de captura-recaptura, *Rostrhamus sociabilis plumbeus*

Introduction

Monitoring natural populations is often a necessary step to establish the conservation status of species and to help improve management decisions (Yoccoz et al. 2001). Nevertheless, many monitoring programs do not effectively address two important components of variation in monitoring data: spatial variation and detectability, which ultimately may limit the utility of monitoring in identifying declines and improving management (Yoccoz et al. 2001). Detectability refers to the probability that an animal will be detected if it is present in the sampled area (Williams et al. 2002). Many sources of variation may affect detectability (e.g., observer effect, environmental conditions), and monitoring data that do not take detectability into account will typically lead to biased estimates (Williams et al. 2002). Spatial variation is another source of variability of monitoring data. It results from the inability to sample the entire area of interest (i.e., inference is drawn from selected spatial units that are only a fraction of the area of interest; this is a problem when the areas sampled are not representative of the entire area) (Williams et al. 2002).

The principle that monitoring programs should take detectability and spatial variation into consideration is gaining some support among wildlife biologists (Williams et al. 2002). Nevertheless, analyses based on uncorrected counts continue to be published in major journals (reviewed in Rosenstock et al. 2002; Conn et al. 2004). The continued controversy around the value of uncorrected count-based indexes results partly from the fact that monitoring programs that estimate detectability are often more labor intensive (Conn et al. 2004). Some authors have argued that when the focus is on population change rather than population size, uncorrected count-based indexes may be sufficient, but for this latter statement to be correct, detectability should remain constant over time, which is rarely the case when monitoring mobile organisms (Conn et al. 2004). Some managers have been slow to accept that it is crucial to estimate detection and to account for spatial variation when monitoring animal populations. This is problematic because recovery plans for many endangered species are still based on monitoring programs that ignore these primary sources of variations (e.g., Cape Sable Seaside Sparrow [*Ammodramus maritimus mirabilis*], Wood Stork [*Mycteria americana*]; USFWS 1999).

We used the monitoring of the Snail Kite (*Rostrhamus sociabilis plumbeus*) in Florida to illustrate the impor-

tance of considering detectability and spatial variation. The Snail Kite feeds almost exclusively on freshwater snails and, thus, is considered a wetland-dependent species (Beissinger 1988). In the United States the Snail Kite is restricted to the remaining wetlands of central and south Florida (Dreitz et al. 2002). Because the availability of snails to kites is strongly dependent on hydrological conditions, variations in water levels are likely to influence kite behavior and demography, especially droughts, which reduce snail availability (Beissinger & Takekawa 1983).

The drainage of the Everglades that began in the early 1930s and was followed by wetland destruction led to the collapse of kites in Florida (USFWS 1999). In 1967 the Snail Kite was first listed as endangered pursuant to the U.S. Endangered Species Conservation Act (ESA) (USFWS 1999). Three primary quantitative recovery criteria were set by the U.S. Fish and Wildlife Service (USFWS) in 1999 on which to base reclassification of the Snail Kite from endangered to threatened (USFWS 1999): (1) "the 10-year average for the total population size is estimated as ≥ 650 , with a coefficient of variation (CV) less than 20% for the pooled data" (USFWS 1999); (2) "no annual population estimate is less than 500"; and (3) "the rate of increase to be estimated annually or biannually and over the 10-year period will be greater than or equal to 1.0, sustained as a 3-year running average" (USFWS 1999). These criteria, however, were set in reference to data obtained from uncorrected counts (USFWS 1999). Since 1965 several agencies have been conducting kite surveys throughout the designated critical kite habitats (reviewed in Bennetts et al. 1999). One major weakness of most count surveys is that detectability is not considered (Bennetts et al. 1999). Hereafter we used the term *counts* to refer to uncorrected counts, which correspond to the number of animals counted during a survey. These counts represent an unknown fraction of the target population (i.e., detection probability is not taken into account; Williams et al. 2002). In contrast the phrases *estimate of population size* and *estimate of superpopulation size* correspond to population parameters of interest that take detectability into consideration.

Dreitz et al. (2002) provided the first estimates of population size that accounted for detection, and their estimates (from 1997 to 2000) were four to five times greater than the target set by the USFWS in 1999. Estimates of Dreitz et al. (2002) during the study period also indicated a fairly stable population. Under the objectives set by the recovery plan of 1999, these figures were encouraging.

Estimates presented in this study, however, indicate that criteria set by the recovery plan in 1999 need revision and that count surveys of populations that occupy large landscapes may be dangerously misleading.

Using a recent estimator of superpopulation size, we examined the implications of carefully considering detection probabilities and spatial variation when making inferences about changes in population size and number of young produced. In this study the superpopulation consisted of all kites that had a non-zero probability of being detected over the course of the sampling year (Dreitz et al. 2002). The superpopulation approach used in our study and in Dreitz et al. (2002) was based on capture-mark-resighting analyses. This approach allowed for the estimation of superpopulation size that takes into account detectability and spatial variation (Williams et al. 2002). This approach makes use of capture-mark-resight models such as Cormack-Jolly-Seber models (CJS) (Dreitz et al. 2002).

Methods

Primary Steps

We conducted six primary steps in this study. (1) We estimated superpopulation size for young and adults separately. (2) Based on these estimates, we examined changes in abundances by estimating population growth rates, and we looked for population decline. (3) We then compared count data with superpopulation estimates. Because detection probabilities for counts are typically <1.0 , counts will often underestimate the “true” population size (Williams et al. 2002). (4) We computed one type of detection probability (denoted P) as the ratio of the number of kites counted over the estimated superpopulation size and examined how detection varied over time for two types of count surveys. If this detection estimate varied substantially over time, then count-based indexes could be very misleading. For instance, if detection increases over time (e.g., because of additional field personnel), population growth rates derived from counts may suggest that the population is growing or is stable, when in fact, the population is decreasing. Detection probability based on the ratio statistic (P) is different from detection probability (denoted p) directly estimated with capture-mark-resight models such as the CJS (Williams et al. 2002). (5) We examined the recovery criteria that the USFWS set for kites to determine whether these criteria were met or were close to being met based on two types of monitoring data, one that considered detection and spatial variation (superpopulation approach) and another that did not (i.e., count data). (6) Finally, we examined the implications of addressing detectability and spatial sampling for the conservation of kites and other endangered species.

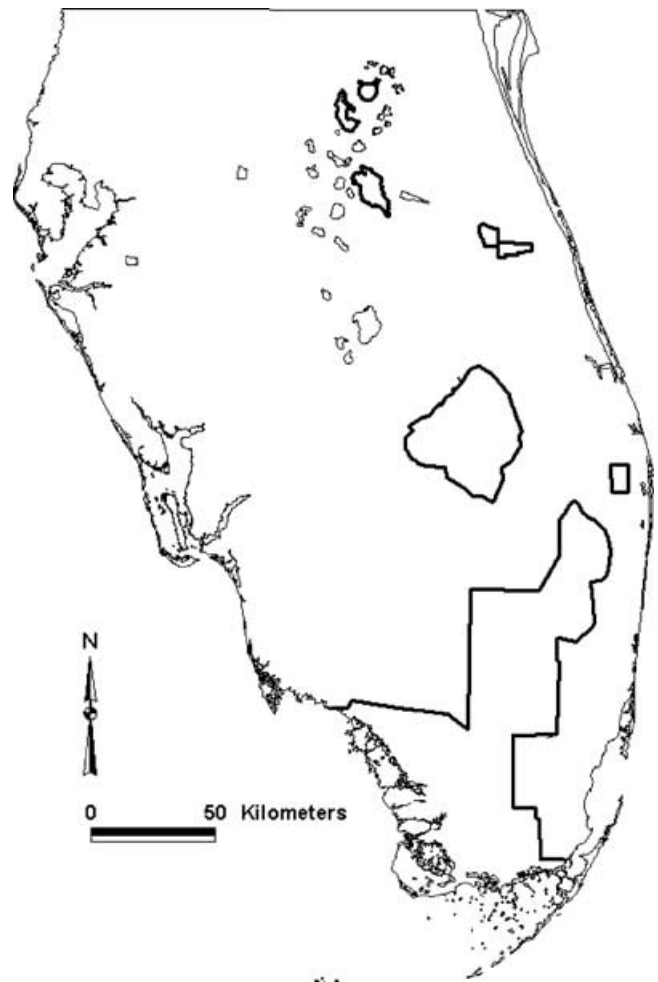


Figure 1. Map of the wetlands sampled to obtain both counts and capture-resighting information of Snail Kites for the estimation of population size. Thick black line delimits areas sampled by the Florida Cooperative Fish and Wildlife Research Unit.

Study Area

The Snail Kite population in Florida has been described as geographically isolated (Dreitz et al. 2002; Martin et al. 2006). Kites occupy the remaining wetlands of the Kissimmee-Okeechobee-Everglades freshwater watershed. The sampled units we used are identical to the units Dreitz et al. (2002) used and encompassed major kite habitats (Fig. 1). Although kites may temporarily emigrate to unsampled areas, it is unlikely that they will not return to the major wetlands included in the survey during some portion of the sampling period (Dreitz et al. 2002).

Sampling Methods

Multiple, consecutive surveys of Snail Kites from airboats were conducted during the peak of the breeding season (March through June) throughout the areas sampled from 1992 to 2005. During surveys workers located nests and

banded young kites when they were ready to fledge (approximately 25 days). A total of 1806 young were marked between 1992 and 2005. Prior to 1995, 134 kites were marked as adults (>1 year). In addition, 76 kites were marked as young prior to 1992. Kites were marked with alphanumeric bands.

Our protocol for population surveys from 2001 to 2005 followed that described by Dreitz et al. (2002) and was part of the same Florida Cooperative Fish and Wildlife Research Unit (FCFWRU) kite-monitoring program. Four to six consecutive surveys from airboats were conducted at 2- to 4-week intervals throughout the designated wetland units from 25 February to 30 June. From 2001 to 2003, the surveys started between 1 March and 8 March and ended between 15 June and 19 June. In 2004 and 2005, the surveys started between 25 February and 1 March and ended between 27 June and 30 June. During each survey we inspected with binoculars and spotting scopes every sighted kite. We categorized each observed individual as (1) marked if the kite carried a band that could be uniquely identified, (2) unmarked if the sighted kite did not carry an identifiable band, or (3) unknown if the banding status of the kite could not be determined.

Analysis

SUPERPOPULATION SIZE ESTIMATES OF ADULTS

We used the superpopulation approach (Schwarz & Arnason 1996) generalized by Schwarz and Stobo (1997) in a robust-design framework. This approach allows for movement during secondary occasions (i.e., between surveys within a year, see Dreitz et al. 2002). Dreitz et al. (2002) were the first to apply this method to the Snail Kite. All notations follow Dreitz et al. (2002). The superpopulation approach estimated the total number of kites present in the sampled area for at least some of the surveys during the sampling period (surveys within a year were denoted $i = 1, 2, \dots, n$). For any given year (denoted j), we referred to this estimate as the superpopulation estimate for each year (\hat{N}_j^*):

$$\hat{N}_j^* = \hat{N}_{1j} + \sum_{i=1}^{n-1} \hat{B}_{ij}, \quad (1)$$

where \hat{N}_{1j} is the estimated abundance of the first survey in year j . The \hat{N}_{ij} is estimated as

$$\hat{N}_{ij} = \frac{m_{ij} + u_{ij}}{\hat{p}_{ij}}, \quad (2)$$

where m_{ij} is the number of marked kites and u_{ij} the number of unmarked kites at each survey i in year j . Kites whose banding status was unknown were excluded from this analysis. The \hat{p}_{ij} (sighting probability) is the estimated probability of sighting a kite given that it was present in survey i of year j . Given the constraint, $p_{1j} = p_{2j}$, abundance can be estimated for all surveys within year j .

The \hat{B}_{ij} is the estimate of the number of new kites entering the sampled area (between survey i and $i + 1$) from areas not sampled on each survey:

$$\hat{B}_{ij} = \hat{N}_{i+1,j} - \hat{N}_{ij} \hat{\phi}_{ij}, \quad i > 0, \quad (3)$$

where $\hat{\phi}_{ij}$ (apparent survival) is the estimated probability of not dying and not permanently emigrating to an area not sampled between surveys i and $i + 1$ of year j . Given the constraint $p_{1j} = p_{2j}$, \hat{B}_{ij} can be estimated for $i = 1, \dots, (n - 1)$.

As in Dreitz et al. (2002), we used the CJS model implemented in program MARK to estimate $\hat{\phi}_{ij}$ and \hat{p}_{ij} (Cooch & White 2005). We analyzed each annual capture-recapture data set separately to obtain estimates of ϕ_i and p_i within a year. We preferred this approach to a multi-group approach (used in Dreitz et al. 2002) because the date for the within-year surveys did not match exactly from year to year. Unlike Dreitz et al. (2002) the number of sampling surveys in our study varied between four and six. (The number of sampling surveys should not affect the superpopulation estimate as long as the sampling periods remain similar.) Although we ended our surveys before 30 June in 2001, 2002, and 2003, our data indicate that all surveys conducted after the second week of June did not affect estimates of superpopulation size. Indeed, removing the last survey for years when surveys ended approximately on 30 June, did not affect the estimate of abundance. This can be explained by the fact that by the end of the sampling season, it is unlikely that many birds will enter the sampled areas for the first time (i.e., most birds have already done so before the last survey).

For model selection among CJS models, we used QAIC_c, which corresponds to the Akaike information criterion (AIC) corrected for small sample size and extra-binomial variation (Burnham & Anderson 2002). We ran four CJS models for each year. Models that assumed ϕ_i and p_i remained constant between surveys were denoted with a subscripted dot (.). Conversely, models that allowed ϕ_i and p_i to vary among surveys were denoted with a subscripted t . Thus, the four models were denoted: ϕ , p , ϕ , p , ϕ_t , $\phi_t p$, and $\phi_t p_t$. As recommended by Burnham and Anderson (2002), we used model-averaged estimates of ϕ and p . The purpose of estimating ϕ and p with CJS models was primarily to compute estimates of superpopulation size. Confidence intervals of estimates of superpopulation size were computed with the same parametric bootstrap procedure (500 simulations) described in Dreitz et al. (2002).

The assumptions for the superpopulation model are similar to the ones required for the more widely known Jolly-Seber model (Williams et al. 2002). In particular, homogeneity of rates among animals is assumed. The superpopulation model also assumes that all members of the superpopulation unavailable until t will exhibit similar probability of being available for capture at $t + 1$. The goodness-of-fit test (test 2 + test 3) implemented in

program RELEASE, which tests for homogeneity of ϕ_i and p_i and for lack of independence of survival and capture events (Burnham et al. 1987), is also applicable for the superpopulation model (Williams et al. 2002). There is no evidence of heterogeneity or lack of independence for ϕ_i and p_i when probability p is >0.05 (Cooch & White 2005). Burnham and Anderson (2002) indicate that model structure is acceptable for extrabinomial factor $\hat{c} < 4$, and they suggest adjustments be made for extrabinomial variation if $\hat{c} > 1$. We computed \hat{c} with program RELEASE (Burnham et al. 1987).

ESTIMATION OF POPULATION GROWTH RATE AND NUMBER OF YOUNG

Annual population growth rate (λ_j), was estimated as

$$\hat{\lambda}_j = \frac{\hat{N}_{j+1}^*}{\hat{N}_j^*}. \quad (4)$$

We then computed the arithmetic average of all the $\hat{\lambda}_j$ over the last 8 years (1998–2005) and a 3-year running average (denoted $\bar{\lambda}_{j-(j+2)}$).

We used the superpopulation approach described above to estimate the number of young produced in any given year (hereafter referred as young) for the entire superpopulation (denoted \hat{N}_{Yj}^*). For this analysis m_{ij} and u_{ij} (see Eq. 2) included exclusively kites that were born in year j . We used this approach only in 2004 and 2005 because we began recording m_{ij} and u_{ij} for the young in 2004. There was not enough band resight information of young in 2004 and 2005 to estimate ϕ_{ij} and p_{ij} that were specific to that particular age class. Therefore, we used ϕ_{ij} and p_{ij} computed for adults to estimate the number of young produced in 2004 and 2005.

DETECTION PROBABILITIES FOR NUMBER OF YOUNG PRODUCED EVERY YEAR

Only a proportion of the total number of young produced were detected and marked every year. To estimate the proportion of young marked during each year (i.e., detection probability of young), we used the following estimator (Williams et al. 2002):

$$\hat{P}_{Yj} = \frac{C_{Yj}}{\hat{N}_{Yj}^*}, \quad (5)$$

where \hat{P}_{Yj} is the detection probability of young in year j , and C_{Yj} is the number of young observed and marked in year j (hereafter referred as the number of young marked). The \hat{P}_{Yj} differed from survey-specific \hat{p}_{ij} (directly estimated with CJS models).

COUNT DATA

We used two types of count-survey data that we subsequently compared with estimates of superpopulation size: first count (FC) and maximum count (MC). For FC

we used the first FCFWRU annual count survey (total number of birds counted during the first survey) as an indicator of annual abundance. We used the first annual count survey because it was always conducted at the same date (1 March \pm 1 week). Many agencies, including the Florida Fish and Wildlife Conservation Commission (FFWCC), use this type of format for surveys in which a designated study area is sampled annually (typically at the same time of the year). The MC was annual count data of the maximum count obtained for any of the FCFWRU surveys within a sampling season. The MC and FC included marked, unmarked, and unknown kites. Nevertheless, because in 1997 unknown birds were not reported, all analyses related to FC and MC data focused on the period 1998–2005.

AVERAGE NUMBER OF KITES AND GROWTH RATE BASED ON COUNT DATA

We computed the arithmetic average for the three sets of count data (\bar{C}). We also used these count data sets to compute annual growth rate based on counts (λ_{cj}). The λ_{cj} were estimated as follows (Williams et al. 2002):

$$\hat{\lambda}_{cj} = \frac{C_{j+1}}{C_j}. \quad (6)$$

We then computed the 8-year arithmetic average of all the $\hat{\lambda}_{cj}$ and the 3-year running average: $\bar{\lambda}_{cj-(j+2)}$.

DETECTION PROBABILITIES FOR FC AND MC

Monitoring based on counts typically assumes detection probability to equal 1.0; however, in practice this assumption is rarely met (Williams et al. 2002). We determined the detection probability of Snail Kites with both FC and MC surveys by computing the ratio of the number of kites counted in a given year j (C_j) (with either FC or MC) over the estimated superpopulation size for that same year (\hat{N}_j^*):

$$\hat{P}_j(\text{FC}) = \frac{C_j(\text{FC})}{\hat{N}_j^*} \quad \text{and} \quad \hat{P}_j(\text{MC}) = \frac{C_j(\text{MC})}{\hat{N}_j^*}. \quad (7)$$

We emphasize that $\hat{P}_j(\text{FC})$ and $\hat{P}_j(\text{MC})$ differed from survey-specific \hat{p}_{ij} (directly estimated with CJS models). We also established, for each type of survey (FC and MC), the increase in detection probabilities (in percentage) necessary to obtain an average count of kites (\bar{C}) ≥ 650 .

Estimates of Precision and Magnitude of the Difference between Estimates

We used the delta method to compute the variances of derived estimates (Williams et al. 2002). The 95% confidence intervals (95% CI) of any parameter θ that were not strictly positive (e.g., estimate of magnitude of the difference, see below) were computed as follows: 95%

CI $[\hat{\theta}] = \hat{\theta} \pm t_{0.025,df} * \widehat{SE}[\hat{\theta}]$, where $\widehat{SE}[\hat{\theta}]$ is the estimated standard error of $\hat{\theta}$ and $t_{0.025,df}$ is the upper 97.5 percentile point of the t distribution on df (Burnham & Anderson 2002). As recommended by Burnham and Anderson (2002), for any parameter θ that is strictly positive (e.g., population size), we used an approximation for computing 95% CI $[\hat{\theta}]$ based on a lognormal distribution (Burnham & Anderson 2002). The magnitude of the difference between two estimates (\widehat{MD}) was estimated by computing the arithmetic difference between estimates (Cooch & White 2005).

Results

Population Size and Average Population Growth Rate

Estimates of $\hat{\phi}_{ij}$ and \hat{p}_{ij} were obtained with model averaging of models $\phi \cdot p \cdot$, $\phi \cdot p_t$, $\phi_t \cdot p \cdot$, and $\phi_t \cdot p_t$ for each year (estimates of $\hat{\phi}_{ij}$, \hat{p}_{ij} , and other survey-specific parameter estimates used to compute estimates of superpopulation size are available, see Supplementary Material). There was no need to adjust for lack of fit of the most general model in 2002, 2003, 2004, and 2005 because \hat{c} was ≤ 1 and test 2 + test 3 from RELEASE were not significant ($p > 0.05$). In 2001 the test 2 + test 3 was significant ($p = 0.02$). Therefore, we adjusted for lack of fit of the most general model in 2001 ($\hat{c} = 2.2$). Estimates of superpopulation size (\hat{N}_j^*) from 1997 to 2000 were obtained from Dreitz et al. (2002), whereas estimates from 2001 to 2005 are the results of the present study. Estimates of superpopulation size between 1997 and 2000 were fairly constant and relatively high (Dreitz et al. 2002; Fig. 2). Superpopulation size estimates decreased sharply during the interval 2000–2002, but there was an apparent stabilization, or even slight increase (but note the 95% CI overlap), in 2004 and 2005. The average superpopulation size for the last 9 years (1997–2005) was 2254 (95% CI = 2124–2392). Estimates of the 8-year average growth rate based on superpopulation estimates was 0.93 (95% CI = 0.84–1.03). Estimates of annual growth rate based on superpopulation estimates were > 1 in 1998, 2003, and 2004 (Table 1). Estimates of the 8-year average growth rate was 1.11 (95% CI = 0.91–1.37) based on FC and 0.99 (95% CI = 0.86–1.12) based on MC. Estimates of the 3-year running-average growth rate based on superpopulation estimates and on MC were > 1 for $\hat{\lambda}_{02-04}$ only (Table 1). Estimates of the 3-year running-average growth rate based on FC were < 1 for $\hat{\lambda}_{00-02}$ only (Table 1).

Average Number of Kites before and after Decline Based on Superpopulation Approach

Estimates of superpopulation size suggest three major periods: predecline (1997–2000); decline (2000–2002); and postdecline (2002–2005) (Fig. 2). We computed the average number of kites during the pre- and postdecline periods.

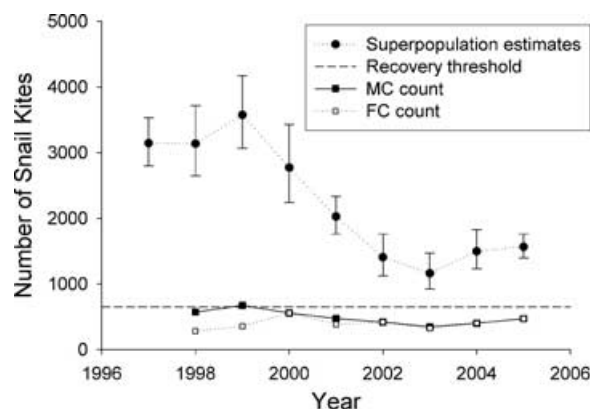


Figure 2. Comparison of the estimates of population size of Snail Kites (based on the superpopulation approach) with annual counts. Data for two count surveys are plotted in the figure: first count survey (FC); maximum count (MC) survey. Kite numbers and estimates of population size from 1997 to 2000 were obtained from Dreitz et al. (2002). Estimates from 2001 to 2005 are results of the present study. Error bars correspond to 95% confidence intervals. The recovery target for Snail Kites (650 birds), set by the U.S. Fish and Wildlife Service in 1999, is also presented.

Prior to decline (1997–2000), the average number of kites based on the superpopulation estimator was 3157 (95% CI = 2909–3426). After decline (2002–2005) the average number of kites was 1407 (95% CI = 1278–1550). There was a substantial decrease between before and after decline ($\widehat{MD} = 1750$; 95% CI = 1457–2041). This represented a 55% decrease (95% CI = 46–67%) when compared with predecline levels.

Table 1. Estimates of Snail Kites annual population growth rates and 3-year running average of growth rates.

Parameter ^a	Year (j)						
	1998	1999	2000	2001	2002	2003	2004
Annual rates							
$\hat{\lambda}_j(S)$	1.14 ^b	0.77 ^b	0.73 ^b	0.69	0.83	1.29	1.05
$\hat{\lambda}_{cj}(FC)$	1.26	1.57	0.69	1.09	0.77	1.25	1.17
$\hat{\lambda}_{cj}(MC)$	1.18	0.83	0.85	0.89	0.83	1.16	1.17
Average rates							
$\hat{\lambda}_{j-(j+2)}(S)$	0.88 ^b	0.73 ^b	0.75 ^b	0.94	1.05	—	—
$\hat{\lambda}_{cj-(j+2)}(FC)$	1.17	1.12	0.85	1.04	1.06	—	—
$\hat{\lambda}_{cj-(j+2)}(MC)$	0.95	0.85	0.86	0.96	1.05	—	—

^aParameter explanations: $\hat{\lambda}_j$, estimates of annual population growth rate based on superpopulation estimates (S); $\hat{\lambda}_{cj}$, estimates of annual population growth based on first-count surveys (FC) and maximum count survey (MC); $\hat{\lambda}_{j-(j+2)}$, 3-year running average growth rate based on $\hat{\lambda}_j(S)$; $\hat{\lambda}_{cj-(j+2)}$, 3-year running average growth rate based on $\hat{\lambda}_{cj}(FC)$ and $\hat{\lambda}_{cj}(MC)$.

^bComputed with data from Dreitz et al. (2002).

Average Number of Kites before and after Decline Based on Count Data

Average number of kites before decline (1998–2000) based on FC data was 397 kites (95% CI = 164–959) and 403 (95% CI = 316–514) after decline (2002–2005). Therefore, FC data showed a slight increase in kite numbers between the intervals 1998–2000 and 2002–2005; however, the difference was not biologically significant ($\widehat{MD} = 6$; 95% CI = -220 to 208). Based on MC data, average number of kites (\bar{C}) before decline (1998–2000) was 600 (95% CI = 462–779) and after decline (2002–2005) was 410 (95% CI = 337–499). Although 95% CI intervals of \bar{C} overlapped, MC data showed a substantial decrease in kite numbers between the intervals 1998–2000 and 2002–2005 ($\widehat{MD} = 190$; 95% CI = 82–298). This represented a 32% decrease (95% CI = 20–51%).

Average Number of Kites and Growth Rate Based on Count Data

The average number of kites counted with FC and MC for the last 8 years (1998–2005, but excluding 2001) was 401 kites (95% CI = 319–503) for FC and 491 kites (95% CI = 392–616) for MC. The CV was 0.09 for MC and FC. Estimates of annual growth rate based on FC were <1 in 2000 and 2002 (Table 1). Estimates of annual growth rate based on MC were <1 from 1999 to 2002 (Table 1).

Number of Young

There was a sharp decline in the number of young marked starting in 1999 (Fig. 3). The average number of young marked from 1992 to 1998 was 200 (95% CI = 145–277), whereas the average number of young marked between 1999 and 2005 was 61 (95% CI = 38–96). The difference was substantial ($\widehat{MD} = 139$; 95% CI = 75–204). This represented a 70% decrease (95% CI = 41–100%).

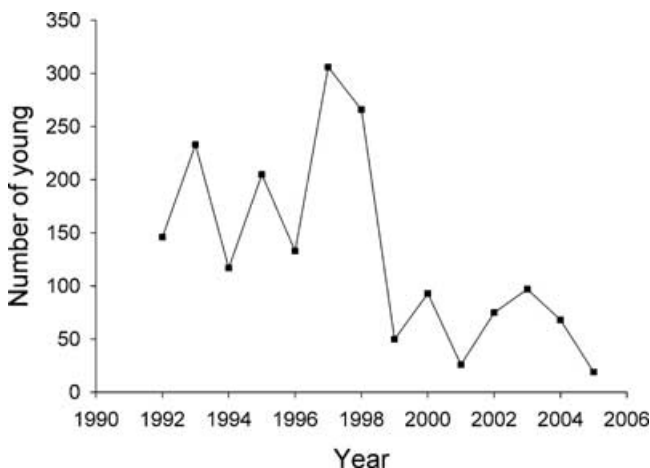


Figure 3. Number of young (i.e., nestlings close to fledging) Snail Kites marked every year from 1992 to 2005.

The number of young produced in 2004 and 2005 based on the superpopulation approach were $\hat{N}_{Y2004}^* = 414$ and $\hat{N}_{Y2005}^* = 55$. The detection probabilities in 2004 and 2005 were $\hat{P}_{Y2004} = 0.16$ and $\hat{P}_{Y2005} = 0.35$. Estimates of confidence intervals could not be computed because sample size of resighting of young kites was too small.

Detection Probabilities for FC and MC

An increase of 63% in detection probability was necessary to obtain an average count >650 for the FC survey; an increase of 33% in detection probability was necessary to reach a similar target based on the MC survey data (Table 2). Detection increased over the years for both type of surveys.

Discussion

Population Decline

Our results based on the superpopulation approach indicate that the population of Snail Kites in Florida declined sharply between 2000 and 2002 (Fig. 2). Although estimates were slightly higher for 2004 and 2005, there was no evidence of a substantial recovery. The reduction in the estimated average kite abundance before and after decline was substantial (56% reduction in abundance). The method we used to estimate the superpopulation size of kites was also useful in obtaining the number of young produced per breeding season. Although this parameter is difficult to estimate in the wild, it is often needed to evaluate the viability of threatened populations. For instance, the superpopulation approach is an appealing method to compute reliable estimates of fertility rates, which are critical to correct parameterization of many types of population viability analyses (Morris & Doak 2002).

We only had data to compute estimates of the number of young produced for 2 years (2004 and 2005). We also used these estimates to compute the proportion of young marked during these 2 years (i.e., detection of young). The fact that detection of young varied substantially in 2004 and 2005 suggests that one should be cautious in using the number of young marked as an indicator of the number of young produced. Nevertheless, we believe detection estimates for these 2 years corresponded to extreme values. We expected low detection probability for 2004 because birds bred unusually early, which meant a large proportion of birds fledged before they could be marked. Conversely, in 2005 we invested an unprecedented effort in nest searching and marking young, which led to higher detection. Unless detection has declined significantly between the intervals 1992–1998 and 1999–2005, we expect the observed number of young marked to be representative of an important decline in the number of young produced. We believe detection is likely to have increased in recent years because we invested more

Table 2. Estimates of detection probability of Snail Kites for first-count surveys (FC) and maximum count surveys (MC) for each year between 1998 and 2005^a.

Detection	Year (j)								\bar{C}
	1998 ^b	1999	2000 ^b	2001	2002	2003	2004	2005	
\hat{P}_j (FC)	0.09	0.10	0.20	0.19	0.30	0.28	0.27	0.30	399
\hat{P}_j (FC) + 63% * \hat{P}_j (FC)	0.15	0.16	0.33	0.31	0.49	0.45	0.44	0.49	651
\hat{P}_j (MC)	0.18	0.19	0.20	0.23	0.30	0.30	0.27	0.30	490
\hat{P}_j (MC) + 33% * \hat{P}_j (MC)	0.24	0.25	0.27	0.31	0.40	0.40	0.36	0.40	654

^aEstimates were obtained by computing the ratio count over superpopulation size for each year (j). The \bar{C} corresponds to the average number of kites counted using the estimated detection for FC (\hat{P}_j (FC)) and for MC (\hat{P}_j (MC)) and the detection probabilities that were increased by 63% for the FC surveys (\hat{P}_j (FC) + 63% * \hat{P}_j (FC)) and 33% for the MC surveys (\hat{P}_j (MC) + 33% * \hat{P}_j (MC)).

^bDetection probabilities computed with estimates of superpopulation size published in Dreitz et al. (2002).

effort in nest-searching activities than in earlier years. An increase in detection implies that the reduction in the number of young produced in recent years is even more severe than is apparent in Fig. 3. Models used to obtain the number of young produced assumed that estimates of ϕ_{ij} and p_{ij} for young and adults were similar. Appropriate sample size of resighting of young kites should be collected in the future to check the assumption that adult estimates of ϕ_{ij} and p_{ij} provide a reasonable approximation to estimate the number of young produced.

Problems Associated with Counts and Implications for Recovery Plans

Identifying population decline is critical to the process of species conservation. In practice it is often the documentation of population decline below a critical threshold that leads to the classification of species as endangered under the ESA. Additionally, identifying a reduction in population size may prevent unsubstantiated downlisting. The legal protection offered by the ESA is in many cases essential to the persistence of many species at risk of extinction (Doremus & Pagel 2001). Our results provide a compelling example of the risks associated with setting recovery targets based on deficient monitoring programs. Some of the current recovery targets presented in the Snail Kite recovery plan (USFWS 1999) could be met (even with a declining population) if monitoring does not account for major sources of errors such as detection.

One of the major recovery criteria listed in the plan states that the 10-year average population size should be >650. Even the most recent superpopulation estimates obtained during our study indicated that the actual Snail Kite population size may be twice this number. This suggests that although the 8-year-average counts obtained with FC and MC were all below the recovery target set by the USFWS (i.e., 650 kites), it is likely that by increasing the search effort (e.g., increase in the number of field personnel), more than 650 kites could have been counted during these surveys. In fact, an increase of 33% in detection probability (i.e., the proportion of kites counted

from the “true” population size) during MC counts and 63% during the FC counts would have boosted the average number of kites counted over 8 years to above the 650 target (Table 2). In both cases CV was <0.2 (i.e., <20%). The second recovery criteria states that kite numbers should not fall below 500 for any given year. This condition would not have been met for the FC count with detection increased by 63%, because the count in 1998 would be 460 (but increasing detection from 0.09 to 0.16 in 1998 would have brought the count for that year to 501). Similarly, this condition would not have been met for the MC count with detection increased by 33%, because the count in 2003 would be 463 (but increasing detection from 0.30 to 0.43 in 2003 would have brought the count for that year to 501).

The third recovery criterion stipulates that the 3-year running average should not be <1.0 over a period of 10 years. Out of the five averages that could be computed for the last 8 years of data for the FC count, only one value fell below 1.0. Reducing the proportion of birds that were observed in 2001 from 0.19 to 0.11 (Table 2), would have pushed all values of the running average for the FC count above 1.0 (although the lower CI of these values may have fallen below 1.0, nothing is mentioned in the recovery plan about parameter uncertainty of growth rate estimates). One can think of scenarios that would cause such a reduction in the proportion of kites counted. For example, dry conditions could reduce airboat access to wetlands used by kites.

Finally, when computing the 8-year average of growth rate for the superpopulation, the FC count and the MC count, we found that the growth rate was <1.0 for the superpopulation and the MC count but was >1.0 for the FC count (in all cases lower 95% CI were <1.0). The fact that the estimate of the average growth based on FC data was >1.0, even though the population was declining, is most likely due to the increase in detection probabilities over the years (Table 2). This increase in detection probability was also observed for the MC data, and resulted in an inflated 8-year average growth rate for the MC survey as well. The observed increase in detection probabilities

over time could be due to an increase in the number of field personnel in recent years (since 2002).

A particularly disturbing fact regarding count data is that, despite the drop in kite abundance (56% based on the superpopulation approach), the FC count did not indicate a reduction in kite numbers, though the MC count did indicate a reduction. The MC count may be less biased than FC because for every sampling year the maximum count will be closer to the true abundance than any other single count. This is because all counts underestimate true abundance; therefore, the maximum count should be the closer to the true abundance than any other count. Nevertheless, because both types of counts ignored detection and did not deal effectively with sampling variation, they were biased. The FC count (i.e., single annual count) is by far the most common type of count survey. The format of the FC count was very similar to the surveys conducted by the FFWCC between 1995 and 2004, except that the FFWCC annual count took place during midwinter (December to February) and was restricted to fewer wetlands (FFWCC, unpublished data). This spatial restriction also increases potential for errors associated with spatial variation. Thus, by simply varying the proportion of kites observed during counts (i.e., detection), three major recovery criteria in the Snail Kite recovery plan were close to being met based on a monitoring that relied on counts (e.g., FC) in spite of an alarming decrease in estimated population size and reduced reproduction.

Our primary purpose was not to set new recovery criteria for the Snail Kite (although our study strongly suggests that existing criteria are in need of revisions), and several authors have proposed promising approaches to set more appropriate criteria (e.g., Gerber & DeMaster 1999; Morris & Doak 2002). Instead, we emphasize the critical importance of designing monitoring programs that address major, common sources of errors because reliability of the recovery criteria will depend strongly on the quality of the monitoring data.>

Importance of Monitoring to Diagnose Causes of Decline

Although identification of population decline is an important step, it is evidently only part of the process of protecting a species. A next step should be to diagnose the cause(s) of decline, or alternatively, factors limiting growth. In the case of kites the drought that occurred in 2001 appears to coincide with the population decline and strongly affected adult and juvenile survival (Martin et al. 2006). The drought affected kite survival, however, only temporarily (1–2 years, Martin et al. 2006). The lack of evident recovery 4 years after this natural disturbance suggests that factors affecting reproduction and recruitment may prevent growth. The drastic reduction in the number of young kites marked (70% decrease), suggests that factors limiting reproduction may deserve more attention than they have in the past. Nevertheless, rigorous evalu-

ation of the causes of decline and factors limiting growth need to be performed. Hypotheses related to disease, predation, food availability, and nest substrate should probably be the focus of future investigations (Perry et al. 2004; J.M. et al. unpublished). The multiple competing hypotheses approach (MCH) provides an appealing framework to disentangle the factors that could potentially affect population growth of threatened species (Williams et al. 2002; Perry et al. 2004). Ideally, monitoring programs designed to tease apart ecological hypotheses with MCH will incorporate both spatial variation and detectability. Addressing spatial variation is particularly important to assess hypotheses related to spatial dynamics effectively (Yoccoz et al. 2001). This may be of particular relevance to the management of spatially structured populations of species that occupy large landscapes (Yoccoz et al. 2001).

Conclusion

A growing number of ecologists are recognizing the value of using designs that incorporate both detectability and spatial sampling because they allow for better parameter estimates and they favor more effective evaluation of ecological hypotheses (reviewed in Yoccoz et al. 2001). As illustrated by our results, these sampling-design issues are extremely relevant to the protection of endangered species. Indeed, ignoring detectability and spatial variation may lead to dangerously inappropriate management decisions (e.g., unsubstantiated downlisting). Nonetheless, considerable resources continue to be invested in monitoring programs that ignore these sources of variability, and many recovery plans continue to rely on these flawed programs. Given the immediate risks of extinction faced by an increasing number of species, it is urgent for managers and conservation biologists to rigorously revisit recovery plans and monitoring programs that do not effectively address spatial sampling and detectability.

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Supplementary Material

The following supplementary material is available for this article:

Table S1. Survey-specific parameter estimates used to compute estimates of superpopulation for Snail Kites between 2001 and 2005.

This material is available as part of the online article from: <http://www.blackwell-synergy.com/doi/abs/10.1111/j.1523-1739.2006.00613.x>

Literature Cited

- Beissinger, S. R. 1988. The Snail Kite. Pages 148–165 in R. S. Palmer, editor. *Handbook of North American birds*. Yale University Press, New Haven, Connecticut.
- Beissinger, S. R., and J. E. Takekawa. 1983. Habitat use by and dispersal of Snail Kites in Florida during drought conditions. *Florida Field Naturalist* **11**:89–106.
- Bennetts, R. E., W. A. Link, J. R. Sauer, and P. W. Sykes Jr. 1999. Factors influencing counts in an annual survey of Snail Kites in Florida. *Auk* **116**:316–323.
- Burnham, K. P., D. R. Anderson, G. C. White, C. Brownie, and K. H. Pollock. 1987. Design and analysis of methods for fish survival experiments based on release-recapture. *American Fisheries Society Monograph* **5**:1–437.
- Burnham, K. P., and D. R. Anderson. 2002. *Model selection and multimodel inference: a practical information-theoretic approach*. Springer-Verlag, New York.
- Conn, P. B., L. L. Bailey, and J. R. Sauer. 2004. Indexes as surrogates to abundance for low-abundance species. Pages 59–74 in W. L. Thompson, editor. *Sampling rare or elusive species*. Island Press, Washington, D.C.
- Cooch, E. G., and G. C. White. 2005. MARK—an introductory guide. Available from <http://www.phidot.org/software/mark/docs/book/> (accessed November 2006).
- Doremus, H., and J. E. Pagel. 2001. Why listing may be forever: perspectives on delisting under the U.S. Endangered Species Act. *Conservation Biology* **15**:1258–1268.
- Dreitz, V. J., J. D. Nichols, J. E. Hines, R. E. Bennetts, W. M. Kitchens, and D. L. DeAngelis. 2002. The use of resighting data to estimate the rate of population growth of the Snail Kite in Florida. *Journal of Applied Statistics* **29**:609–623.
- Gerber, L. R., and D. P. DeMaster. 1999. A quantitative approach to endangered species act classification of long lived vertebrates: application to the North Pacific humpback whale. *Conservation Biology* **13**:1203–1214.
- Martin, J., J. D. Nichols, W. M. Kitchens, and J. E. Hines. 2006. Multiscale patterns of movement in fragmented landscapes and consequences on demography of the Snail Kite in Florida. *Journal of Animal Ecology* **75**:527–539.
- Morris, W. F., and D. F. Doak. 2002. *Quantitative conservation biology: theory and practice of population viability analysis*. Sinauer Associates, Sunderland, Massachusetts.
- Perry, Z. M., S. R. Beissinger, S. H. Newman, E. B. Burkett, and T. D. Williams. 2004. Applying the declining population paradigm: diagnosing causes of poor reproduction in the Marbled Murrelet. *Conservation Biology* **18**:1088–1098.
- Rosenstock, S. S., D. R. Anderson, K. M. Giesen, T. Leukering, and M. F. Carter. 2002. Landbird counting techniques: current practices and an alternative. *Auk* **119**:46–53.
- Schwarz, C. J., and A. N. Arnason. 1996. A general methodology for the analysis of capture-recapture experiments in open populations. *Biometrics* **52**:860–873.
- Schwarz, C. J., and W. T. Stobo. 1997. Estimating temporary emigration using the robust design. *Biometrics* **53**:178–194.
- USFWS (U.S. Fish and Wildlife Service). 1999. South Florida multi-species recovery plan. USFWS, Vero Beach, Florida. Available from http://ecos.fws.gov/docs/recovery_plans/1999/990518.pdf (accessed November 2006).
- Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. *Analysis and management of animal populations*. Academic Press, San Diego, California.
- Yoccoz, N. G., J. D. Nichols, and T. Boulinier. 2001. Monitoring of biological diversity in space and time. *Trends in Ecology & Evolution* **16**:446–453.