

**FINAL REPORT:
AVIAN RESTORATION IN EVERGLADES NATIONAL PARK:
PHASE IV**

AVIAN MONITORING, EFFECTS OF FIRE, HYDROLOGY, AND VEGETATION, AND LONG-TERM
RECOMMENDATIONS FOR MANAGEMENT OF THE PINE ROCKLAND BIRD COMMUNITY



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HOW TO READ THIS REPORT

The following report represents research conducted under grants from Everglades National Park through the Critical Ecosystems Science Initiative and the Land Water Conservation Fund. The research addresses three independent, but interrelated goals. In the executive summary, bulleted highlights are listed under individual headings for objectives within each goal. In the body text of the report, each goal is associated with its own section. Some of the research has been published in peer-reviewed manuscripts or other final reports. In those cases, we noted this in the text of the section and attached those documents as appendices at the end of the report.

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EXECUTIVE SUMMARY

This reports summarizes the results of a comprehensive 3-year project (2004-2007) investigating questions related to the long-term management and conservation of the pine rockland avifauna of Everglades National Park. The project had three primary goals, each with specific objectives; key results of each objective are listed below.

Goal 1: Evaluate the success of the Brown-headed Nuthatch, Eastern Bluebird, and Wild Turkey reintroduction programs in Long Pine Key.

Objective 1. Evaluate the success of Brown-headed Nuthatch and Eastern Bluebird reintroductions through population and demographic monitoring.

- Brown-headed Nuthatch population size increased from 31 to 87 adults from the cessation of translocations in 2001 to 2005, reaching about 30% of the estimated carrying capacity. The population declined during the following two years to approximately 52 individuals, but the population remained widely distributed across Long Pine Key.
- Pradel reverse-time, capture-recapture models indicated that the nuthatch's realized population growth rate during the period from 2003-2007, on average, was > 1.0 . Realized population growth rate varied annually, increasing from 2002-2005 (2002: $\lambda = 1.15$, SE = 0.13; 2003: $\lambda = 1.28$, SE = 0.12; 2005: $\lambda = 1.32$, SE = 0.20) and decreasing thereafter (2006: $\lambda = 0.67$, SE = 0.10; 2007: $\lambda = 0.77$, SE = 0.13).
- We suspect that nuthatch declines in 2006 and 2007 may be related to Hurricanes Katrina and Wilma, which struck Long Pine Key following the 2005 breeding season. Although hurricanes may have caused direct mortality, they more likely produced indirect effects on survival and reproduction via changes in food availability (e.g., by stripping pine trees of their cones).
- The Eastern Bluebird population in Long Pine Key decreased in size from 38 to 35 adults from the conclusion of translocations in 2001 to 2007. Pradel reverse-time, capture-recapture models indicated the population was slowly declining with no annual variation ($\lambda = 0.92$, 95% CI = 0.83–1.00).
- Factors limiting the bluebird population remain elusive. Bluebird productivity declined over the course of the study, with most nest failures due to predation. We found no evidence for an increase in avian predators and have no information on the population sizes of other terrestrial predators. Declining reproduction may reflect deteriorating habitat conditions, perhaps through the availability or quality of nesting cavities and snags. We also observed numerous instances of mortality, especially for juveniles, due to vehicle collisions.
- The re-establishment of Brown-headed Nuthatch and Eastern Bluebird populations in Long Pine Key indicates progress in restoration of the pine rockland ecosystem within Everglades National Park. However, neither population was demonstrably secure at the end of the study.

- We recommend that population size and demographic monitoring of Eastern Bluebirds continue because of the population's small size and negative growth rate. We recommend that management actions aimed at directly increasing population size be implemented; for example, placing aluminum flashing around nesting cavities to increase productivity and actions to reduce vehicle speeds, particularly along Research Road, to increase juvenile survival. Research to address limiting factors for bluebirds, such as cavity availability, should be investigated.
- Habitat management for both species should focus on using prescribed fire to provide open forest conditions with abundant snags. Short-interval (1–3 years) fires are suitable in areas with dense hardwood understories, but longer return intervals (4-7 years) should be considered in other areas so as to optimize the balance between snag creation and snag mortality.

Objective 2. Investigate the relationships between environmental factors (i.e., fire, hydrology), vegetation, and demographics in nuthatches and bluebirds.

- We examined how variation in vegetation structure at the nest site, fire history, and hydrological conditions affected productivity of Brown-headed Nuthatches and Eastern Bluebirds in the pine rocklands of southern Florida. We located and monitored the fate of 141 Brown-headed Nuthatch nests and 248 Eastern Bluebird nests in Long Pine Key and Raccoon Point from 1998-2003.
- The most important predictor of productivity for nuthatches was the date on which a nest attempt began, with earlier nests producing more fledglings. The number of large (>15 cm dbh) pine snags and, to a lesser extent, the number of small pine trees surrounding a nest site were positively associated with productivity.
- The best predictors of bluebird production were snag dbh, snag height, and distance from cavity to a branch. These variables likely reflect vulnerability to predation. In general though, these variables alone have a relatively weak effect on productivity.
- None of the fire or hydrological variables were directly associated with productivity. However, fire is an important factor in creating and consuming snags and hydrology is often a modulator of fire intensity.
- We recommend that land managers in Long Pine Key focus on providing abundant large pine snags because doing so will increase productivity and also may increase nest-site availability and the percentage of individuals that breed each year. Prescribed burning is likely the most effective way to increase the abundance of large pine snags. However, caution should be exercised when doing so because of the trade-off between snag recruitment and snag consumption that accompanies the use of fire.

Objective 3. Evaluate the Wild Turkey reintroduction program through population monitoring.

- We monitored the reintroduced Florida Wild Turkey in Long Pine Key during the period from October 2004 to July 2007 using standardized camera surveys following FFWCC protocols.

- Following initial camera surveys that indicated a population size index of ≤ 15 individuals, 31 turkeys (20 females; 11 males), of which 10 were radio-tagged, were released south of the Long Pine Key Campground in January 2006, ,
- Subsequent camera surveys indicated the population size index increased, reaching 21 individuals during the winter of 2007, and personal observations detected successful breeding in 2006 and 2007, with > 39 juveniles produced.
- Turkey home range sizes (95%KHR) varied from 5.7 - 33.7 km² (mean =16.9, SE = 4.0) and were generally higher than estimates from other studies. Low habitat quality at the southernmost limit of the species' range may require larger home ranges to support individuals in Long Pine Key.
- Turkeys almost exclusively used pine and prairie/marsh habitats, especially in the core of their home range. Continued restoration in the hole-in-the-donut should create additional prairie habitats and increase turkey carrying capacity in Long Pine Key.
- We recommend that camera surveys continue for two more years, as recommended by FFWCC.

Goal 2: Implement a monitoring program for breeding and non-breeding birds in pine rocklands of South Florida.

Objective 1. Investigate the relationships between environmental factors (e.g., fire, hydrology), vegetation, and avian populations in south Florida pine rocklands.

- We established 104 bird survey points in Long Pine Key and 95 in Raccoon Point, and an additional 238 points in Big Cypress National Preserve, Miami-Dade County and Florida Panther National Wildlife Refuge. We conducted avian surveys during the non-breeding (15 Dec. – 15 Feb.) and breeding (15 Apr. – 1 June) seasons, and at each point, we measured vegetation attributes, determined the time since last fire and the number of burns in the past 10 years, and estimated the average water stage for each sampling period.
- We found variation in hydrology was the primary driver of large-scale variation in vegetation structure and the distribution and abundance of breeding and wintering birds across the range of south Florida slash pine. Fire history was of secondary importance. However, models explained low levels of variation, suggesting that other factors such as biotic interactions, other past and ongoing sources of disturbance, and biogeography may be important in explaining large-scale variation in avian composition.
- Variation in hydrology was the primary driver of within-site variation in the distribution and abundance of breeding and wintering birds. Fire history was of secondary importance. In Long Pine Key and three other sites, variation in breeding-bird density was most strongly related to variation in water-table elevation. Water-table elevation was always the strongest predictor of bird densities during the non-breeding season.

- Five species (Northern Cardinal, Pine Warbler, Red-bellied Woodpecker, Downy Woodpecker, Great-crested Flycatcher) showed a positive association with fire across all study sites. For all species, abundance remained constant for up to 5 years after fire, then declined from 5-7 years post-fire, and finally leveled out at > 7 years post-fire. Three of these species are cavity nesters – Downy Woodpecker, Red-bellied Woodpecker, and Great-crested Flycatcher – and their increased abundance at shorter fire return intervals may be due to the increase in number of snags associated with recent fires.
- We modeled long-term snag population dynamics of an old-growth forest in Big Cypress National Preserve. We found that: 1) dry-season burns were more intense than wet-season burns and resulted in a higher probability of snag creation than wet-season burns; 2) prescribed burns conducted at longer intervals increased the probability of snag creation; 3) the probability of mortality of existing snags (snag loss) was greater with wet season burns, and 4) burning in the dry season and with longer return intervals is predicted to yield greater densities of snags.
- Several areas of research need were highlighted included research into factors that drive variation in snag recruitment and mortality, factors that drive variation in the abundance and distribution of cavity-nesting birds, and factors that drive variation in wintering birds.

Goal 3: Provide recommendations for long-term monitoring recommendations for the breeding and non-breeding avian communities and for Brown-headed Nuthatches and Eastern Bluebirds.

Objective 1. Provide long-term monitoring recommendations for the breeding and non-breeding avian communities and for Brown-headed Nuthatches and Eastern Bluebirds.

- We conducted a power analysis via simulation for a rare species, Brown-headed Nuthatch, and a common species, Pine Warbler, to provide estimates of the required sample sizes needed for long-term trend monitoring for a broad range of species.
- Few sampling points (< 40) were needed to effectively monitor (power > 0.80) Pine Warblers at any of the three modeled effect sizes. For nuthatches, 95 points were needed to detect a 50% decline over 10 years and 45 points were needed to detect a 50% decline over 25 years. For our weakest trend, 25% decline over 10 years, nuthatches can not be monitored without substantially more than 104 survey points.
- The study design that was established in 2005 effectively monitors 15 of 19 (79%) breeding season species and 18 of 23 (78%) non-breeding season for declines at levels considered appropriate for avian monitoring programs (i.e., annual declines of 5% in 10 years [50% decline] or annual declines of 3% for 25 years [50% decline]). The study design will be less effective at monitoring birds at the weaker effect size, a 25% decline in 10 years (3% annual rate), but even under this conservative trend, many species from numerous guilds in both sampling periods were effectively monitored.
- We recommend that the established study design for monitoring pinelands birds be incorporated as a component of a long-term monitoring program in the pine rocklands of

ENP. A long-term monitoring program for pineland birds would be valuable to evaluate the long-term restoration (i.e., CERP) and management (e.g., prescribed fire) actions in this critically endangered ecosystem, where few long-term monitoring projects have been initiated. A long-term monitoring program would also provide an efficient means to monitor the status of one of the reintroduced species (Brown-headed Nuthatch), and cavity-nesting birds in general. Cavity-nesting birds have been considered as potential performance measures in the pine rocklands.

- We used distance sampling to generate breeding and non-breeding seasons population estimates for Brown-headed Nuthatches and Eastern Bluebirds in Long Pine Key conducted from 2005-2008 and compared results to ground based population surveys.
- Annual population size estimates calculated using distance sampling were not in accord with ground-based estimates, with distance sampling estimates generally 2-4 times higher than ground-based estimates.
- Both techniques contain potential sources of bias, which likely contributed to differences in population size estimates. Distance sampling relies on an accurate model of detectability, which is determined, in part, by the numbers of detections of focal species. Detections of nuthatches and bluebirds were low due to their small population sizes. For the ground based surveys to be accurate, every individual that was present within the 3,940 ha of pine forest that served as our sampling area must be counted, an assumption surely violated to some unknown extent.
- Population estimates trends were similar and of the same magnitude between techniques, indicating that both methods likely provide valuable information for monitoring. The advantage of distance sampling is that it estimates a parameter and its variance via a defined statistical model, whereas the ground-based survey is an index that is related to population size in an unknown fashion. Distance sampling is also substantially more cost-effective because one can monitor abundance and trends of many species. Ground based surveys are most valuable when populations are smaller, such as in the case of Eastern Bluebirds.
- We recommend that distance sampling be employed for long-term monitoring of Brown-headed Nuthatches as long as population sizes remains fairly large (>50 individuals) and there is no indication of consistent declines over time or a reduction in their distribution across Long Pine Key. For Eastern Bluebirds, we recommend that ground-based population estimates be continued until their population size and distribution increases and population models reveal sustained population growth. When species are rare, accurate population size estimates are important so that management actions can be quickly implemented if population levels dip to critical levels.

INTRODUCTION

The pine rocklands are a globally unique subtropical forest ecosystem considered "critically imperiled" by the U.S. Fish and Wildlife Service (USFWS 1999) and the network of Natural Heritage programs (NatureServe 2004). Found only in southern Florida, the Bahamas, and Cuba, pine rocklands have been severely altered by human development, logging, and altered fire and hydrological regimes. In southern Florida, pine rocklands occur in the Florida Keys, Big Cypress Swamp, mostly in Big Cypress National Preserve (BCNP), and along the Atlantic coastal ridge, which terminates in Everglades National Park (ENP). The most glaring impacts to pine rocklands have occurred along the Atlantic coastal ridge. Once covering over 70,000 ha, the pine rocklands of the Atlantic coastal ridge have been reduced in size by more than 90% (Doren et al. 1993). The largest remaining tract is the 8,100 ha upland area in Long Pine Key, ENP.

While the majority of remaining pine rocklands, outside of the Florida Keys, have been protected and are no longer threatened by development, the restoration and long-term management of its plant and animal communities present significant challenges. The pine rocklands are fire maintained, requiring fire to deter succession to hardwood hammock. Prescribed fire is the primary management action applied in this ecosystem. South Florida is also the target of a large-scale hydrological restoration program, the Comprehensive Everglades Restoration Plan (CERP), which aims to restore a more natural pattern of hydrology, with respect to quantity and timing, through the Everglades System. However, little is known about the effects of manipulating fire or hydrological regimes on the terrestrial fauna, particularly birds, and their habitats in pine rocklands. As such, managers lack guidelines for integrating the needs of the pine rockland avifauna into management and restoration plans. Addressing this information gap is critical if restoration and long-term management is to proceed effectively and efficiently in pine rocklands, and studies on the effects of fire and hydrology on avian communities has been identified as a significant information need in the Department of Interior's Science Plan for South Florida (DOI 2005), the USFWS Multi-species Recovery Plan (USFWS 1999), and the Avian Conservation Implementation Plans for ENP and BCNP (Watson 2003a, b).

Birds are of particular interest as a subject of research in the pine rocklands, as they appear particularly vulnerable to the degradation and loss of habitat. The most notable effects have occurred in ENP where seven species of breeding bird have been extirpated: Wild Turkey (*Meleagris gallopavo*), Summer Tanager (*Piranga rubra*), Brown-headed Nuthatch (*Sitta pusilla*), Eastern Bluebird (*Sialia sialis*), Red-cockaded Woodpecker (*Picoides borealis*), Southeastern American Kestrel (*Falco sparverius paulus*), and Hairy Woodpecker (*Picoides villosus*). That five of the seven extirpated species are cavity nesters (Nuthatch, Bluebird, Woodpeckers, and Kestrel) suggests that this guild may be especially vulnerable to habitat changes. Pine rocklands also provide important wintering habitat for neotropical migrants (Emlen 1980), but the effects of habitat degradation on wintering birds are undocumented. However, many of the neotropical migrants that winter in the pine rockland ecosystem have shown long-term population declines (e.g., Palm Warbler, *Dendroica palmarum*, a Partners In Flight species of concern for subtropical Florida) and recent evidence links population declines of neotropical migrants with habitat quality on the wintering grounds (Norris et al. 2004).

In 1997, an avian restoration program was initiated to develop and implement translocation techniques aimed at restoring viable populations of Brown-headed Nuthatches and Eastern Bluebirds to ENP (Slater 2001). Following this effort, an attempt to reintroduce Wild Turkey was initiated in 2000. Both studies were initiated as tests of the progress made in restoring the rare pineland ecosystem (e.g., restoration of natural fire regimes, protection and recovery of the area from logging) represented by Long Pine Key, ENP. Post-release monitoring of nuthatch and bluebird populations indicated the reintroductions were a success, but additional monitoring was recommended to fully evaluate the success of the program (Slater 2004). Monitoring of Wild Turkeys indicated that reproduction had occurred but few subsequent observations of individuals suggested that the population may require additional augmentation.

In 2004, we initiated a comprehensive 3-year project of the pine rockland avifauna to address three goals, each with specific objectives.

Goal 1: Evaluate the success of the nuthatch, bluebird, and wild turkey reintroduction programs in the uplands of Long Pine Key.

Specific objectives were to:

- (1) evaluate the success of Brown-headed Nuthatch and Eastern Bluebird reintroductions through population and demographic monitoring;

- (2) investigate the relationships between environmental factors (i.e., fire, hydrology), vegetation, and demographics in nuthatches and bluebirds; and
- (3) evaluate the Wild Turkey reintroduction program through population monitoring.

Goal 2: Implement a monitoring program for breeding and non-breeding birds in pine rocklands of South Florida.

Specific objectives were to:

- (1) investigate the relationships between environmental factors (e.g., fire, hydrology), vegetation, and avian populations in south Florida pine rocklands. Although this component was initiated in Long Pine Key, ENP and Raccoon Point, BCNP, we leveraged these funds to expand the project to include five additional sites and an extra year of surveys (2008) with a grant from the Joint Fire Science Program.

Goal 3: Provide recommendations for long-term monitoring of the pineland avifauna.

Specific objectives were to:

- (1) provide long term monitoring recommendations for the breeding and non-breeding avian communities and for the two reintroduced species, nuthatches and bluebirds, in pine rocklands of ENP.

SECTION 1.0

EVALUATE THE SUCCESS OF THE NUTHATCH, BLUEBIRD, AND WILD TURKEY REINTRODUCTION PROGRAMS IN THE UPLANDS OF LONG PINE KEY.

Objective 1. Evaluate the success of Brown-headed Nuthatch and Eastern Bluebird reintroductions through population and demographic monitoring.

Results from this objective have been published in the Journal of Wildlife Management (Lloyd, J.D., G.L. Slater, and S. Snow (2009)). A copy of this paper is attached in [Appendix 1](#).

Objective 2. Investigate the relationships between environmental factors fire, hydrology, vegetation, and demographics in nuthatches and bluebirds.

A manuscript detailing the relationships between Brown-headed Nuthatch productivity, fire, hydrology, and vegetation has been published in the Journal of Wildlife Management (Lloyd and Slater 2007). A copy of this paper is attached in [Appendix 2](#).

A report detailing the relationships between Eastern Bluebird productivity, fire, hydrology, and vegetation was completed in 2008 and submitted to ENP. We aim to submit this manuscript for publication in Fall 2010. A copy of this report is provided in [Appendix 3](#).

Objective 3. Evaluating the Wild Turkey reintroduction program.

Introduction

The Florida Wild Turkey (*Meleagris gallopavo osceola*) was extirpated from the Long Pine Key region of ENP in the mid- to late 1950's. Factors contributing to their extirpation likely included unregulated hunting on private in-holdings within the park boundaries (O. Bass, W. B. Robertson, Jr., G. Simmons, L. E. Williams, Jr.; pers. comm. to Skip Snow 1999), habitat loss and fragmentation of pine forests, and altered fire regimes (Snyder et al. 1990).

Historical records and verbal accounts suggest that several attempts were made to reintroduce turkeys to Long Pine Key. The first reintroduction attempt was believed to have been initiated by landowners to private in-holdings sometime during the late 1940's. A second attempt may have taken place in the early 1960s. Verbal accounts suggest that approximately 20 to 30 birds were released and that successful reproduction occurred. However, these reintroduction efforts appear to have failed, perhaps due to continued hunting pressure on private farm in-holdings in ENP. ENP wildlife observation records indicate a third reintroduction occurred in the spring of 1971, around the Pine Island area, based on reports of individuals killed by vehicle collisions. The last ENP record of a Wild Turkey was near Pine Glades Lake in the spring of 1975, the year Hole-in-the-donut (HID) in-holdings were transferred to ENP ownership.

A renewed interest in establishing a turkey population in ENP emerged in 1999 for several reasons. First, significant progress in restoration of the endangered pine rockland ecosystem in Long Pine Key had occurred. For example, the pine rockland forest had matured to approximately 60 years of age and the application of a putative natural fire regimes had been implemented. Secondly, unregulated hunting was no longer a threat to the persistence of a reintroduced population. Thirdly, reintroduction efforts of other pineland bird species, the Eastern Bluebird and Brown-headed Nuthatch, were ongoing and appeared to be successful. Finally, natural reestablishment of turkeys was considered unlikely due to the large distance between ENP and the closest source population in Big Cypress National Preserve and the absence of suitable habitat corridors for dispersal.

In 2000, a reintroduction program was initiated by ENP, the Florida Fish and Wildlife Conservation Commission (FFWCC), the Homestead Chapter of the National Wild Turkey Federation (NWTFF), and Ecostudies Institute. The reintroduction served as another test of the

progress made in restoring pine rockland habitat, both through the protection of these forests and through appropriate fire and hydrological management. In January 2000, twenty-nine individuals were released into Long Pine Key.

Success of the turkey reintroduction was difficult to assess because monitoring following the translocation was inconsistent. Radio-telemetry indicated turkeys established territories, while intermittent camera surveys in following years indicated released individuals bred successfully. However, as time passed fewer turkey observations were reported, suggesting that the size of the turkey population in Long Pine Key was declining. To formally assess the turkey reintroduction in Long Pine Key, we initiated a 3-year program of standardized camera survey to 1) determine if additional translocations of Wild Turkeys should be conducted, and 2) evaluate the success of the turkey reintroduction.

Methods

Study area

We studied Florida Wild Turkey in the Long Pine Key region of ENP during the period from October 2004 to July 2007. Long Pine Key is an 8,100 ha upland area that contains approximately 4,600 ha of pine forest (Snyder et al. 1990, Doren et al. 1993). Within the relatively continuous pine forest, which is mostly even-aged, embedded habitats include *Muhlenbergia* prairie, hardwood hammocks, and cypress forest (Olmstead et al. 1983). During the past decade, the fire management program has implemented a prescribed fire regime characterized by fire return intervals of 2-4 years and primary ignition periods in the early rainy season (May – June).

Camera Surveys

We followed the FFWCC Wild Turkey Management Section's protocol for establishing a camera monitoring program. We established ten permanent camera survey stations in Long Pine Key, configured a minimum of 1.6 km apart along the fire roads in Long Pine Key ([Fig. 1-1](#)). We conducted the first survey in October/November 2004. Subsequent surveys were conducted twice a year during two standardized periods recommended by FFWCC: late winter (January/February) and summer (June - September). At each camera site, cracked corn was placed in a single bait pile, spread thinly in front of the camera, and along two thin stringers in both directions away from the camera up to ¼ mile. Sites were pre-baited a minimum of one

week prior to the start date of the camera survey. Each camera unit (CamTrakker, Watkinsville, GA) was placed on a tree adjacent to the fire road at a height where the infra-red sensor was approximately 15 inches above ground ([Fig. 1-2](#)). Cameras were programmed to be activate from sunrise until sunset, with a delay of 5 minutes to avoid repeated pictures of a single feeding event, and the date and time were printed on all pictures. Pictures were downloaded every 2-3 days and bait sites were maintained as needed. We determined a population size index for sex and age classes from the identification of unique individuals, verifiable from tags, bands, or other unique markings, or determined using the criteria that a turkey could move 1 mile/1 hour.

Second Release

Results from the first two camera surveys indicated a turkey population index of ≤ 15 individuals, the criteria established by FFWCC to trigger a second release. In January 2006, thirty-one turkeys (20 females; 11 males), of which 10 were radio-tagged, were released south of the Long Pine Key Campground ([Table 1-1](#)).

Survival and home range analysis. We monitored radio-tagged individuals daily for the first week following release because we assumed turkeys were most vulnerable to mortality and made their largest exploratory movements during this period. After this adjustment period, we located each turkey two times per week until July 2006. We located radio-tagged turkeys in the Long Pine Key area using a series of fixed and random telemetry stations distributed throughout Long Pine Key. For each location, we tried to obtain \geq two compass bearings approximately perpendicular to each other and no greater than 20 minutes apart.

We used LOAS (2005 Ecological Software Solutions) software to calculate locations from multiple bearings. In most cases, we only used a set of bearings that were obtained within a 20 minute period. However, in some cases we used bearings taken within a 40 minute period, if locations were collected midday, when we expected birds to be less active. Each location was considered independent because sequential locations were determined > 24 hours apart and the majority of points were > 48 hours apart. Using the Animal Movement extension in ArcView 3.2 (Hooge and Eichenlaub 1997), we calculated home range size at two scales: a 95% fixed kernel density utilization distribution or kernel home range (KHR) and a 50% KHR, corresponding to the areas in which there was a 95% or 50% probability of an individual being located. We consider the 95% KHR to represent the home range and the 50% KHR to represent the core area within the individual's home range.

Habitat use. We characterized habitat use by radio-tagged turkeys by determining composition of vegetation within 95% and 50% KHRs. We used the South Florida Mapping Project's GIS vegetation database (see Welch et al. 1999) as a base habitat map in determining the percentage of each habitat category.

Personal Observation Summary

In addition to the camera surveys, records of personal observations by park visitors and personnel from NPS and other agency cooperators were maintained to provide supplementary information on the population's distribution and reproductive success.

Evaluating Success

To evaluate the reintroduction effort, we proposed two criteria to serve as benchmarks for success: 1) annual increases in turkey abundance as measured through the population size index using camera surveys, and 2) evidence of successful reproduction.

Results

Second Wild Turkey Release

Survival and home range analysis. Four of the ten (40%) radio-tagged birds, all adult females, died within 2 months of their release. All four individuals were found in moderate to severe decay states. We assumed all birds were dead for > 2 weeks. The cause of death was unknown for three individuals. The fourth individual was found cached in a solution hole in an area of dense shrubs with bite marks on the carcass and transmitter, all evidence pointing to predation by bobcat (*Lynx rufus*).

The remaining six individuals were monitored until 1 July. We obtained 29 – 46 independent locations for each individual. Seaman et al. (1999) recommends greater than 30 independent locations to determine home range estimates. We reached this benchmark for all but one individual. Turkey home range varied from 5.7 - 33.7 km² (mean \pm S.E., 16.9 \pm 4.0) for the 95% KHR and 0.83 – 2.99 km² (1.8 \pm 0.3) for the 50% KHR ([Table 1-2](#))

There was no evidence that any of the radio-tagged females reproduced. All of the radio-tagged females showed signs of frequent movements over fairly large areas throughout the study period.

Habitat use. Turkeys primarily used prairie/marsh and pine forest habitats. These broad categories made up >90% of the vegetation composition in both the 95% and 50% KHRs. The five most important specific habitat categories: Muhly prairie, pine savannah, Brazilian pepper

(*Schinus terebinthifolius*), Slash Pine with hardwoods, and mixed graminoids, were identical in both the 95% and 50% KHR scale ([Table 1-3](#), [1-4](#)). Within the 50% KHR, Muhly prairie and pine forest comprised over 60% of the area, suggesting these habitats are particularly important to turkeys. For all six radio-tagged turkeys their core home range centered on the area south and west of the Long Pine Key campground. It is unclear if this is a result of the area having the highest habitat quality or that individuals simply did not wander from their release site.

Brazilian pepper was an important habitat category within both home range scales. This reflects the decision by many of the turkeys to use the HID area, which is categorized as Brazilian pepper in the base vegetation map. However, the base map does not include changes over the last decade due to restoration, and today much of this area has been restored to prairie habitats. Thus, we believe turkeys are choosing the prairie habitats within the Brazilian pepper, as opposed to the Brazilian pepper.

Camera Surveys

Seven camera surveys were conducted during the study period ([Table 1-5](#)). Both surveys prior to the second turkey release yielded a population size index of 4 individuals. One of the individuals detected was from the original release in 2000. Camera surveys conducted after the second release of turkeys in January 2006 indicated the population size index increased. The exception was the final camera survey in Summer 2007, when the population size index dropped to six individuals. However, no females were detected on that survey.

Personal Observations

After the turkey release in January, turkeys were regularly observed by visitors and NPS and other agency personnel. Most observations were of individuals seen along Research Road, which is not surprising as it is heavily traveled by NPS and other agency employees. The other location where birds were regularly observed was in the HID area, particularly on the spoil mounds and in other grassy openings. Personal observations provide important information on reproduction not acquired on camera surveys because poults are rarely detected on camera surveys. During the 2006 breeding season, 1-3 tagged adult females were observed with 10-14 poults along research road on five different occasions. The highest juvenile count, observed by ENP biologist Helen Fitting, was 23 juveniles on 24 August 2006. The high number of poults suggest that at least 2 females successfully bred, as average clutch size in Florida is 10.3 (range =

5-17; Williams and Austin 1988). In 2007, personal observations indicated a minimum of four successful breeding events and 39 juveniles.

Discussion

Survival and home range

Forty percent mortality of radio-tagged birds is similar to mortality estimates in other turkey reintroduction efforts. All individual died shortly after their release, indicating that the cause of death was likely influenced by factors related to the release or risks associated with acclimating to a new area. Turkey mortality in other reintroductions varied from 28% to 76% (Little and Varland 1981, Clark 1985, Hollis 1985), and in the first turkey release in ENP, 60% of individuals died in the first 6 months (Slater 2000).

Home range estimates in this study (mean = 16.9 km²) were higher than estimates from other studies (Eaton 1992). In general, variation in turkey home range size is high in response to sex, variability of annual food supplies, age and social position (Eaton 1992). Olsen (2006) found home range sizes of female turkeys ranged from approximately 1.5 – 3.8 km² at two sites in north-central FL that varied in habitat quality. However, Thogmartin (2001) found mean home range of 18.9 km² in Arkansas. Given the oligotrophic conditions of habitats in the Everglades ecosystem, we suspect that home range sizes need to be large for individuals to acquire all the resources they need for survival and reproduction.

Habitat use

Radio-tagged turkeys in Long Pine Key primarily used pine and prairie/marsh habitats, especially in the core of their home range. These data are congruent with other habitat use studies conducted during the nesting and summer periods (Eaton 1992), and show their preference for edge habitats and habitat complexity. Turkeys in the southeast US require mature forest habitats with scattered openings, such as prairies, pasture, and grassy glades. Grassy openings are critical as foraging areas, especially for poults, but these areas must be adjacent to escape cover either in the form of dense understory vegetation or forest cover. All of the radio-tagged turkey's core home range centered on the area south and west of the Long Pine Key campground, where there is a substantial amount of prairie/forest ecotone. Turkeys released to Long Pine Key likely use the Muhly, and other graminoid, prairies as primary foraging sites. The understory vegetation density in both prairie and forest upland habitats in Long Pine Key are determined by fire frequency and intensity. Thus, fire effects that incorporate spatial

heterogeneity in fire intensity and fire return interval are likely critical factors in creating the habitat conditions that benefit turkeys and many other wildlife species (e.g., butterflies, bluebirds). Current fire management in ENP seeks to incorporate this heterogeneity in their prescribed fire applications (pers. comm., Rick Anderson, ENP Fire Ecologist). However, the effectiveness of this relatively new practice has not been evaluated.

Surprisingly, turkeys did not use cypress habitats, as reported in other central Florida studies (Williams and Austin 1988). Cypress habitats are found adjacent to the Long Pine Key area. However, the population may not be large enough to disperse and colonize into those habitats. As was observed from the 2000 release, several turkeys spent a large amount of time in the HID area. However, the area has changed significantly since 2000 with the restoration of > 1,600 ha of prairie. Another 1,000 ha of Brazilian pepper is scheduled to be restored over the next several years, and the creation of additional prairie habitats should increase the carrying capacity for turkeys in the Long Pine Key area.

Evaluating Success

Evidence from the camera surveys and personal observations suggest that the second reintroduction of turkeys into Long Pine Key in January 2006 has, thus far, been successful. Two criteria were proposed to serve as benchmarks for success: 1) annual increases in turkey abundance as measured by camera surveys, and 2) evidence of successful reproduction. Camera surveys yielded population size indexes that were similar between the winter camera surveys in 2006 (22 adults) and 2007 (21 adults). However, we detected 3 more females in 2007 than 2006, and the abundance of females may be a better measure to monitor than total abundance considering the turkeys' polygynous mating system.

Evidence of success was also based on observations of breeding in 2006 and 2007. Although turkeys typically do not reproduce in the spring following translocation, we know that at least two females bred in 2006. In 2007, we found evidence of 5 successful breeding efforts and 39 juveniles. Because most personal observations occur near research road and the HID, we likely miss breeding events in other parts of Long Pine Key. As this area becomes saturated, we may not necessarily observe increases in the number of poults detected. However, breeding in this region of Long Pine Key will likely serve as an annual indicator of successful reproduction in other occupied areas of Long Pine Key. Furthermore, successful reproduction confirms the

availability of suitable breeding habitat in Long Pine Key. Given the degree of reproduction observed, we should expect the turkey population to increase in size in 2008.

Visitation rates to bait stations by turkeys during the summer surveys appear more variable than during the winter surveys, making their effectiveness for monitoring turkeys unclear. During the Summer 2007 survey, only 6 turkeys, all males, were detected, compared to 17 in 2006. Variation in visitation rates may be due to several factors. First, females may avoid bait stations because their offspring require a diet dominated by invertebrates. Females without young often join other females with young and thus also do not visit bait stations. Local food availability may also influence visitation rates, although it is unclear how food availability varies seasonally for turkeys in ENP. If food availability is higher during the summer sampling period, turkeys may be less attracted to bait stations sites. Water levels may also play a role in visitation rates by turkeys. Again, it is unclear how water levels and seasonality influences turkey movements or food availability in ENP. High water levels make baiting camera stations problematic, as many of the roads are flooded in or near bait stations and bait stringers. Bait stringers are often shorter than the recommended length or are covered with water. This may result in a longer period needed for turkeys to find the bait stations or may push turkeys into higher elevation interior pinelands away from camera locations. Overall, infrequent captures of turkey images in the summer suggest this period may not be effective for monitoring the turkey reintroduction.

Recommendations

FFWCC recommended camera survey monitoring for five years following the second release of Wild Turkeys and that level of effort was agreed upon in the MOU between ENP and FFWCC. Consequently, we recommend that camera surveys continue for two more years through 2009. Funding has been made available for that level of effort. At the end of 2009, final recommendations regarding long-term monitoring of the turkey population will be determined.

Although summer surveys appear to be less effective than winter surveys, we recommend continuing the summer survey during the next two years to evaluate differences between winter and summer survey results.

Table 1-1. List of released Wild Turkeys in Everglades National Park in January 2006.

Date (capture)	Date (release)	Frequency	Band #	Wing Tag #	Color	Sex	Age	Capture Location
1/5/2006	1/5/2006	151.640	329	2	red	F	Juv	Lykes
1/5/2006	1/5/2006	151.780	483	1	red	F	Adult	Lykes
1/5/2006	1/5/2006	151.860	405	14	red	F	Adult	Lykes
1/5/2006	1/5/2006	151.920	404	4	red	F	Adult	Lykes
1/5/2006	1/5/2006	151.990	445	3	red	F	Adult	Lykes
1/7/2006	1/8/2006	151.600	406	22	red	F	Juv	Sharps
1/7/2006	1/8/2006	151.720	404	20	red	F	Adult	Sharps
1/7/2006	1/8/2006	151.830	405	21	red	F	Adult	Sharps
1/7/2006	1/8/2006	151.890	402	18	red	F	Adult	Sharps
1/7/2006	1/8/2006	151.940	403	19	red	F	Adult	Sharps
1/5/2006	1/5/2006		332	5	red	F	Adult	Lykes
1/5/2006	1/5/2006		326	6	red	F	Adult	Lykes
1/5/2006	1/5/2006		334	7	red	F	Adult	Lykes
1/5/2006	1/5/2006		61	8	red	F	Adult	Lykes
1/5/2006	1/5/2006		310	9	red	F	Adult	Lykes
1/5/2006	1/5/2006		320	10	red	F	Adult	Lykes
1/5/2006	1/5/2006		365	17	red	F	Adult	Lykes
1/5/2006	1/5/2006		418	15	red	F	Adult	Lykes
1/5/2006	1/5/2006		366	16	red	F	Juv	Lykes
1/5/2006	1/5/2006		778	64	white	M	Adult	Lykes
1/5/2006	1/5/2006		688	57	white	M	Adult	Lykes
1/5/2006	1/5/2006		669	56	white	M	Adult	Lykes
1/5/2006	1/5/2006		776	66	white	M	Adult	Lykes
1/5/2006	1/5/2006		777	65	white	M	Adult	Lykes
1/6/2006	1/7/2006		670	55	white	M	Juv	Sharps
1/6/2006	1/7/2006		771	70	white	M	Juv	Sharps
1/6/2006	1/7/2006		773	69	white	M	Juv	Sharps
1/6/2006	1/7/2006		772	71	white	M	Juv	Sharps
1/6/2006	1/7/2006		774	68	white	M	Juv	Sharps
1/6/2006	1/7/2006		775	67	white	M	Juv	Sharps
1/7/2006	1/8/2006		311	11	red	F	Juv	Sharps

Table 1-2. Home range summary for radio-tagged Wild Turkeys.

**Size
(Km²)**

Table 1-4. Percentage of vegetation types within 50% KHR for 6 radio-tagged turkeys in Long Pine Key, ENP.

Broad Vegetation Type	Specific vegetation Type	Turkey 60	Turkey 64	Turkey 72	Turkey 83	Turkey 86	Turkey 99	Mean	SD	Cumulative %
Prairie/Marsh	Muhly	40.8	20.1	36.8	42.8	27.4	26.2	32.4	9.1	32.4
Pine Forest	Pine Savanna	32.4	23.7	29.2	39.5	24.4	30.7	30.0	5.8	62.4
Prairie/Marsh/Exotic	Brazilian Pepper	8.4	33.9	10.2	9.9	38.1	20.9	20.2	13.1	82.6
Pine Forest	Slash Pine with Hardwood	17.2	7.9	19.7	7.3	6.1	7.0	10.9	5.9	93.4
Prairie/Marsh	Mixed Graminoids	0.9	11.1	1.3	0.1	3.0	14.2	5.1	6.0	98.5
Human	Road	0.0	0.8	2.1	0.0	0.8	0.6	0.7	0.8	99.2
Scrub	Saw Palmetto Scrub	0.3	0.1	0.2	0.4	0.2	0.2	0.2	0.1	99.4
Scrub	Hardwood Scrub	0.0	0.8	0.3	0.0	0.0	0.0	0.2	0.3	99.6
Scrub	Groundsel Bush	0.0	0.6	0.0	0.0	0.0	0.0	0.1	0.3	99.7
Prairie/Marsh	Sawgrass	0.0	0.4	0.0	0.0	0.0	0.2	0.1	0.2	99.8
Water	Water	0.0	0.2	0.3	0.0	0.0	0.0	0.1	0.1	99.9
Human	Human Influence	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.1	100.0
Shrub wetland	Willow (<i>Salix caroliniana</i>)	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.0	100.0

Table 1-5. Summary of camera survey results for the period from Fall 2004 to Summer 2007 in Long Pine Key, ENP.

Survey Period	Survey Dates	Number camera days	Number images	Sites detected	Images per camera day	Estimated population size	Total ♂	Total ♀	Total unknown	Tagged ♂	Tagged ♀	Comments
Fall 2004	5 - 12 Nov.	80	12	3	0.15	4	2	2		1		
Winter 2005	2 Feb. - 4 Mar.	178	95	5	0.53	4	3	1		1		
Summer 2005	Cancelled due to weather											
January 2006 -----Turkey Released												
Winter 2006	12 Feb. - 17 Mar.	185	261	8	1.4	22	10	12		8	11	
Summer 2006	4-19 July	144	292	4	2.0	17	9	8		8	8	
Winter 2007	2 - 18 Feb.		64	6		21	6	15		2	9	
Summer 2007	2-10 July	81	34	4	0.40	6	5		1	2		

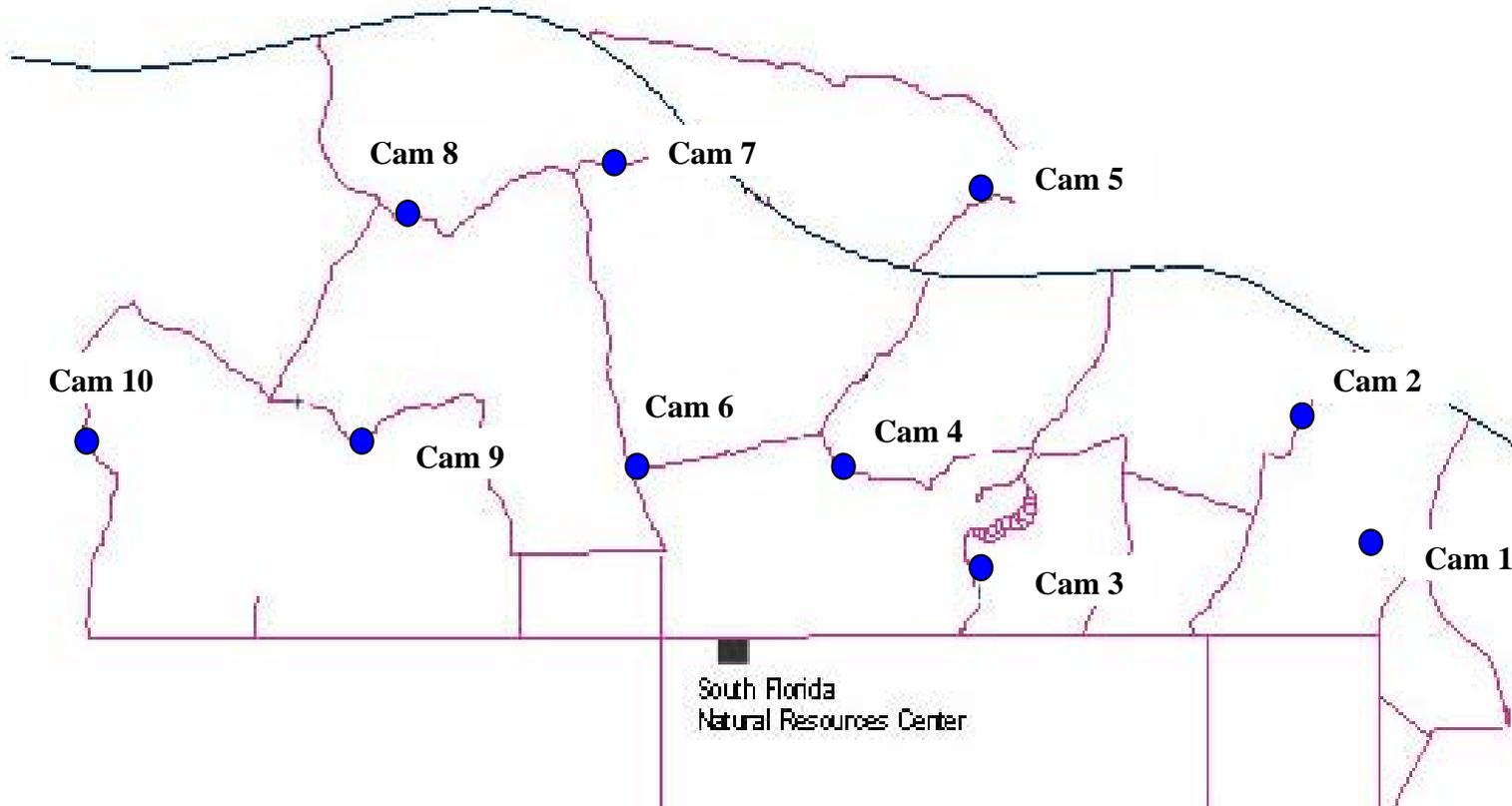


Figure 1-1. Location of camera sites for Wild Turkey surveys.



Figure 1-2. Image of camera unit on pine tree at survey station.

SECTION 2.0

IMPLEMENT A MONITORING PROGRAM FOR BREEDING AND NON-BREEDING BIRDS IN PINE ROCKLANDS OF SOUTH FLORIDA.

Objective 1. Investigate the relationships between environmental factors (e.g., fire, hydrology), vegetation, and avian populations in south Florida pine rocklands.

With funding from Everglades National Park, we initiated a study to investigate the relationships between environmental factors and bird communities in pine rockland habitats. This project was initiated in Long Pine Key, ENP and Raccoon Point, BCNP. However, we obtained a grant for the Joint Fire Science Program, allowing us to broaden the scope of the project, add five additional sites, and include an extra year of surveys (2008). The report for this project was completed in August 2009 and is attached to this document as [APPENDIX 4](#).

SECTION 3.0

PROVIDE RECOMMENDATIONS FOR LONG-TERM MONITORING OF THE PINELAND AVIFAUNA IN EVERGLADES NATIONAL PARK.

Objective 1. Provide long-term monitoring recommendations for the breeding and non-breeding avian communities and for Brown-headed Nuthatches and Eastern Bluebirds.

Introduction

Monitoring of wildlife populations is important to provide an early warning of impending threats, evaluate large-scale disturbance process (e.g., hurricanes), and assess the effects of management practices and restoration efforts. Most avian monitoring activities collect data on: 1) population size or abundance (as an index of population size) and their trends, and 2) population demography (e.g., reproduction and survival). Linking parameters of population size and demographics to habitat characteristics can be a key component of a comprehensive monitoring program.

In previous sections of this report, we described the monitoring of reintroduced species to estimate their population size and demographic parameters (Section 1) and the monitoring of the overall pine rockland bird community during the breeding and non-breeding seasons to characterize their composition and abundance (Section 2). The final goal of this project is to evaluate these monitoring efforts and develop recommendations for long-term monitoring of pineland birds in ENP.

In this section, we addressed two questions related to long-term monitoring of pineland birds. First, we asked if our study design for monitoring birds in the pine rocklands of Long Pine Key, ENP provides adequate statistical power to detect a trend in species abundance if one exists. Second, we asked if population size estimates for two reintroduced species, Brown-headed Nuthatches and Eastern Bluebirds, derived using distance sampling methods via the network of point counts described in Section 2 were similar to ground-based estimates determined in Section 1.

Methods

Power analysis of community wide species

Statistical power ($1-\beta$) is an estimate of the probability of rejecting the statistical null hypothesis at a given level of alpha (α), often $\alpha = 0.05$, when, in fact, the null hypothesis is false. In the context of trend analysis, statistical power reflects the probability of detecting a trend

when one exists (i.e., the slope of the count vs. time regression is not equal to zero). Evaluating the power to detect trends of a given magnitude is a useful, albeit not comprehensive (Seavy and Reynolds 2007), means by which to evaluate population monitoring, as monitoring programs with low statistical power are inefficient and may not yield useful information for managers.

For some analyses, statistical power can be estimated analytically. For more complicated analyses, such as those used to estimate population trends, there is no analytical solution for calculating power. In these cases, power can only be estimated via simulation. We began by formulating a linear model ($y_i = a + b*t$, where y_i is the number of individuals at time i , a is the intercept or number of individuals at time 0, b is the slope of the trend, and t is time. For this study, we evaluated three trends of interest (i.e., effect sizes): 1) a 50% decline over 25 years (i.e., 3% annual decline), 2) a 50% decline over 10 years (i.e., 5% annual decline), and a 25% decline over 10 years (i.e., 3% annual decline). The first two effect sizes represent strong trends, and are commonly applied standards for trend analysis by Partners in Flight and other conservation groups. They describe a small decline over a long time period and a large decline over a short time period. However, under an ideal scenario, managers would like to be able to detect a small decline over a short time period (trend 3), so that remedial actions could be implemented prior to large-scale declines. The objective of the simulations was to determine the number of survey points necessary to detect the proposed trends of interest with power > 0.8 , a level which is a common benchmark for avian monitoring.

We created a simulated time series by drawing a value for each y_i from a Poisson distribution with a mean value equal to the value of y_i estimated from the linear model. The simulated time series therefore had a mean trend as defined by the linear model, but with individual values for each year that reflected random variation (e.g., as might be due to sampling error or random, inter-annual changes in the number of individuals present at a point). We considered the effect of changing sample size by incrementally changing the number of time series (i.e., the number of simulated survey points) in each simulated data set. We then analyzed each simulated data set using generalized linear mixed models (GLMM), where time was considered as a fixed effect and sampling point was analyzed as a random effect. We assumed a first-order autoregressive correlation structure, such that the within-group errors of adjacent points in the time series were correlated. We repeated the process of simulation and analysis 500 times (in preliminary analyses, we ran up to 2,000 simulations and found that estimated power

remained constant after 500 simulations), storing the P-value from each analysis and then estimating statistical power as the proportion of simulations that yielded $P \leq 0.10$. In this context we believe a higher alpha (i.e., higher probability of type I error), compared to the traditional $\alpha = 0.05$, is justified as the goal of most monitoring programs is to alert managers to potential problems and missing a problem poses a greater risk than mistakenly identifying a problem when one does not exist.

Counts of plants and animals often exhibit more variation than expected under the Poisson distribution, and alternative distributions such as the negative binomial may be more appropriate (e.g., Bliss 1953, Purcell 2005). Unlike the Poisson distribution, under which the variance is equal to the mean, the expected variance of the negative binomial distribution increases faster than the mean. As such, simulated time series based on the Poisson distribution may result in overestimates of true statistical power if the biological populations of interest are distributed in a negative binomial fashion. The negative binomial models adds another parameter (θ) that influences the rate at which the variance increases relative to the mean (variance = mean + mean²/ θ). We evaluated the applicability of the Poisson distribution by comparing observed distributions to expected distributions under the Poisson and zero-inflated negative binomial (ZINB). The ZINB is a special case of the negative binomial that accommodates the excessive zeros often found in count data.

All of the simulations described above were repeated for two starting values (i.e., intercepts in the linear model), 0.29 and 2.27. We chose these values to reflect the average number of individuals counted per point for a rare species (Brown-headed Nuthatch [BHNU], 0.29) and one of the most common species (Pine Warbler [PIWA], 2.4). Starting values are estimates of abundance within 150 m corrected for detectability in Program Distance. The starting values in a trend analysis influence the statistical power of the analysis, and so by considering two different values we can provide a conservative estimate of the required sample size for a broad range of species. We report density and estimates of abundance for other pine rockland bird species detected on breeding and non-breeding season surveys and assess their ability to be monitored effectively based on simulation outcomes for PIWA and BHNU. All simulations were conducted in R (R Development Core Team 2008).

Population size estimates of reintroduced species

See Section 1 for methods of estimating population size from ground-based surveys.

We used distance sampling and data from 104 point counts in Long Pine Key conducted from 2005-2008 during the breeding and non-breeding seasons (see Section 2 for detailed methods) to generate population estimates of Brown-headed Nuthatches and Eastern Bluebirds. The breeding season surveys were conducted from 15 April – 1 June and thus provide a direct temporal comparison to the breeding season ground-based surveys. The non-breeding season surveys were conducted just prior to the breeding season (15 December – 15 February).

Distance sampling is one of several methods that can be used to adjust counts of birds to address imperfect detectability due to observers, vegetation, etc. Analyses generate density estimates and 95% confidence intervals, which in turn can be used to calculate population size if the size of the sampling area has been determined. We estimated density of nuthatches and bluebirds using distance sampling as implemented by Program Distance (Thomas et al. 2006). Data were not truncated because models indicated good fit without truncation and because there were few detections of each species. We considered only vocal detections in the final set of models. We considered three models of the detection function: uniform key function, with a cosine expansion; half-normal key function with a cosine expansion; and hazard-rate key function with a simple polynomial expansion term. Because of the low number of detections, we modeled a detection function based on all year's data. We selected the detection function model with the lowest AIC value and then generated separate density estimates for each year. Although density estimates are calculated for each year, the estimates are not independent due to the use of the pooled detection function.

When we established our network of avian survey stations in Long Pine Key, we utilized the UGA vegetation cover map to identify all pine polygons within Long Pine Key. Prior to establishing points, we excluded polygons that were logistically unfeasible to access because of their distance from roads. We then randomly placed points throughout the “available” polygons under the criteria that 1) stations are ≥ 350 m apart, and 2) stations are surrounded by at least 50 m of contiguous pine forest. We calculated the effective area sampled by our 104 sampling points by summing the area of those polygons which were considered available for the establishment of a survey point. The summed value of pine habitat sampled in Long Pine Key was 3,940 hectares and this value was used to generate population size estimates from the density estimates. This area approximately corresponds with the sampling area covered by the ground-based surveys conducted in Section 1.

Results

Power analysis of community wide species

The observed distribution of PIWA counts best followed the theoretical distribution for the Poisson ([Fig. 3-1](#)), and thus simulations were only run with the Poisson distribution. Few sampling points were needed to effectively monitor PIWA at any of the three effect sizes modeled ([Fig. 3-2](#)). Even under the weakest trend, a 3% annual decline over 10 years, only about 40 points, sampled annually, were required to achieve a level of power $(1-\beta) > 0.8$ for PIWA.

For BHNU, the observed distribution of counts did not follow the expected Poisson distribution, due to the excess number of zeros BHNU ([Fig. 3-3](#)). The distribution that most closely matched the observed distribution of counts was a zero-inflated negative binomial distribution with an estimated shape parameter = 0.20. Simulations indicated that approximately 95 points were needed to have adequate power (> 0.80) to detect a 50% decline over 10 years and 45 points were needed to detect a 50% decline over 25 years ([Fig. 3-4](#)). However, for our weakest trend (25% decline over 10 years), simulations indicated that many more points than were modeled in this exercise, perhaps 300-400, would be needed to achieve power levels equaling 0.80 ([Fig. 3-4](#)).

Based on the results of the simulations, 15 species detected during the breeding season have mean abundance equal to or greater than BHNU and thus would likely be monitored with sufficient power to detect 50% decline over 10 years or 25 years ([Table 3-1](#)). Two species have mean abundance values within 0.10 of BHNU and have less likelihood of being monitored effectively. Mean abundance of two species, including the Eastern Bluebird, is substantially lower than the BHNU, and these species are unlikely to be monitored with sufficient power to detect any of the three trends.

During the non-breeding season, both species abundance and species diversity are higher than in the breeding season. Six species, all wintering migrants, have mean abundance greater than our starting value for the PIWA and several species are only slightly lower in abundance, indicating that many should be effectively monitored at all three trend levels. Overall, eighteen species recorded during the non-breeding season have greater mean abundance than our starting value for BHNU, and thus are likely to be monitored effectively at levels similar to or better than the BHNU ([Table 3-2](#)). Four species are unlikely to be monitored at any of the trend levels.

Population size estimates of reintroduced species

Brown-headed Nuthatch. Ground-based surveys resulted in nuthatch population size estimates of 87, 66, and 52 adults from 2005-2007, respectively (see Section 1, [Fig. 3-5](#)).

Sixty-three Brown-headed Nuthatches were detected on breeding season surveys and 51 nuthatches were detected on nonbreeding season surveys during the period from 2005-2008. Brown-headed Nuthatches share a vocal repertoire between sexes, and thus density estimates apply to pooled densities of males and females (individuals/ha). For survey data collected in the breeding season, the hazard rate function with a simple polynomial expansion model resulted in the lowest AIC value. However, other models were not >2 AIC values from the best model, indicating they could be considered in the range of plausible models that best fit the observed data ([Table 3-3](#)). The best model yielded annual nuthatch densities ranging from 0.041 – 0.086, which multiplied by the area sampled (3,940 ha) produced annual population size estimates from 158 – 355 individuals for the period from 2005-2008, all values substantially higher than ground-based estimates ([Table 3-4](#), [Fig. 3-5](#)). For the non-breeding season survey data, the hazard rate function with a simple polynomial expansion was also selected as the best model; no other models were within 2 AIC values ([Table 3-5](#)). Annual density estimates ranged from 0.021 – 0.07, translating to population size estimates of the sampled area of 83, 177, 276, 165 individuals for the period from 2005-2008, respectively ([Table 3-6](#), [Fig. 3-5](#)).

Eastern Bluebird. Eastern Bluebirds were detected 24 times on breeding season surveys and 51 times on non-breeding season surveys from 2005-2008. Eastern Bluebirds have sex-specific vocalizations, with males singing more frequently during the breeding season than females. During the breeding season, density estimates apply presumably to singing males only, because we assumed they are the primary source of detections. During the non-breeding season, density estimates presumably apply to both sexes, since vocalizations out of the breeding season are not likely to be sex-specific. During the breeding season, the uniform function with a cosine expansion was the best model, although the two other models were within 2 AIC values ([Table 3-7](#)). Density of singing males ranged from 0.010 – 0.025, translating to a population size of singing males of 39, 99, 59, and 39 during the period from 2005-2008, respectively ([Table 3-8](#)). Ground based surveys yielded lower population size estimates of males of 19, 17, 17, and 18 during the same period (See Section 1, [Figure 3-6](#)). For the Eastern Bluebird detection function during the non-breeding season, the hazard rate function with a simple polynomial expansion

model resulted in the lowest AIC value; other models were < 2 AIC values from the best model (Table 3-9). The density of total individuals ranged from 0.007 – 0.032, resulting in population size estimates of 24, 59, 28, 126 individuals within the sampled area (Table 3-10, Fig. 3-6). Ground-based estimates of population size were 39, 34, 35, 37 individuals during the breeding seasons from 2005-2008 (Fig. 3-6).

Discussion

Power analysis of community wide species

Results indicate that the majority of breeding and non-breeding bird species in the pine rocklands of Long Pine Key can be effectively monitored by the study design that was established in 2005. That study design uses distance sampling methodology to survey 104 points distributed through Long Pine Key once a year during the breeding and non-breeding season. We found that this design was sufficient for detecting annual declines of 5% in 10 years (50% decline) or annual declines of 3% for 25 years (50% decline) with power > 0.80 for all but the rarest species. The two larger effect sizes simulated in our models are commonly set as monitoring targets to assess species risk in avian monitoring programs (e.g., Rich et al. 2004). Overall, based on our simulations, the study design effectively monitors 15 of 19 (79%) species found during the breeding season and 18 of 23 (78%) species in the non-breeding season.

The study design will be less effective at monitoring birds at the weaker effect size, a 25% decline in 10 years (3% annual rate), with simulations indicating that only the most abundant species were effectively monitored. However, even under this conservative trend, quite a few species from numerous guilds in both the breeding and non-breeding season were effectively monitored. The Pine Warbler with a starting average abundance value of 2.27 required 40 survey points to effectively monitor its population at the weaker trend level, suggesting that even less abundant species such as Great-crested Flycatcher (1.75) and White-eyed Vireo (1.37) would likely be effectively monitored with a program of 104 points. During the non-breeding season, when many wintering migrants are quite abundant, as many as 8-10 species are likely to be monitored with sufficient power to detect a 3% annual declining trend in ten years. Improving our power to detect a decreasing trend for less abundant species would require significantly increasing the number of survey points. However, that option is unfeasible in Long Pine Key because few areas of pine rockland remain that lack points and are logistically

feasible to visit. Another option that could improve power would be to increase the number of visits during the sampling period. We did not investigate this option.

The number of survey points required to achieve sufficient power (>0.80) for the various trend levels in the simulation depended on the starting abundance and the distribution. The species with the larger starting abundance, PIWA, was more easily monitored than the BHNU. This suggests that the ability to monitor trends for some species whose populations we expect to increase with time, such as the reintroduced populations of nuthatches and bluebirds, should become more effective. As was evident in the BHNU simulations and expected under theoretical considerations, a greater number of survey points were needed to detect trends under the zero-inflated negative binomial distribution than the Poisson distribution. This is due to the fact that the variance is much larger under the negative binomial than the Poisson.

We only considered one-tailed tests for decreasing trends in the simulations. This approach is reasonable, as a decrease in population trends is generally of most concern to land managers. However, in some cases, increasing trends are of interests, such as in the case of reintroduced species or introduced species (e.g. starlings) that may compete with native species. In general, one-tailed tests for an increasing trend result in slightly lower power, while two-tailed tests yield substantially lower power (Purcell et al. 2005).

While analyses indicated that the existing study design effectively monitors most pine rockland birds, several caveats deserve noting. First, the simulations may overestimate power if populations show greater variation than expected under the Poisson or zero-inflated negative binomial distributions. However, for the two species modeled in the simulations, the theoretical distributions appeared to be in relative accord with observed distributions. Secondly, the simulations also may overestimate power if some points are never used by a species because of habitat conditions. We suspect that this is not a significant problem in our sampling design because our sampling area, pine rocklands, should be inhabitable by all the target species, even though vegetation varies due to disturbance and succession. Finally, it is also important to note that our power to detect trends may improve in the future. If variables explaining sources of variation in bird abundance (e.g., date, time, observer, habitat) that are not related to the “signal” of the trend are identified, these can be incorporated as additional explanatory variables to improve models of trend analyses, and thus increase power.

Long-term recommendations for monitoring pine rockland bird

We recommend that the established study design for monitoring pinelands birds be incorporated as a component of a long-term monitoring program in the pine rocklands of ENP. Pine rocklands are a critically endangered ecosystem (USFWS 1999), yet few long-term monitoring projects have been initiated in the pine rocklands. Birds are often considered excellent indicators of ecosystem health and a long-term monitoring program for pineland birds would be valuable to evaluate the long-term restoration (i.e., CERP) and management (e.g., prescribed fire) actions in this critically endangered ecosystem. Indeed, such studies have been identified in the DOI Science Plan (2005) as an important research need. Short-term studies of avian populations in the pine rocklands have already yielded important information on the influence of hydrology and fire in determining avian abundance and distribution (Appendix 4, Lloyd and Slater in prep). In addition to overall community monitoring, a long-term monitoring program would also provide an efficient means to monitor the status of one of the reintroduced species (Brown-headed Nuthatch), and cavity-nesting birds in general. Cavity-nesting birds have been considered as potential performance measures in the pine rocklands based on their apparent vulnerability to habitat change as indicated by the numerous extirpations in the Long Pine Key region of ENP. The monitoring program would not be effective in monitoring the reintroduced cavity-nester Eastern Bluebird, at least until the population becomes more widely established and increases in size.

We believe that pineland bird monitoring would be an efficient use of the limited amount of resources available for monitoring. In general, birds are an efficient group for monitoring compared to other, harder to detect taxa, such as mammals. In our experience, an observer can survey 6 points per morning, on average. Completing the 104 survey would therefore require 18 days of work. Including another week for training in bird identification and distance estimation and several days lost to bad weather or equipment failure, completing the surveys and data entry would require approximately 5 weeks of work per sampling period (breeding and non-breeding season), assuming a 5-day work week. An additional amount of time of about 1-2 months for summary analysis and completing an annual report would also need to be incorporated. As the monitoring data set grew, opportunities for additional analyses would increase. For long-term trend analyses, monitoring would need to occur for at least 10 years, although 25 years would be necessary for identifying smaller annual declines for less abundant species.

Population size estimates of reintroduced species

Annual population size estimates of nuthatches and bluebirds calculated using distance sampling in either the breeding or non-breeding season were not in accord with estimates derived from ground-based surveys. In general, population size estimates generated from distance sampling were 2-4 times higher than ground-based surveys and confidence intervals did not envelop ground-based estimates. The single exception was the distance sampling estimates of Eastern Bluebird population size in the non-breeding season, which were relatively similar, except for one year (2008), to the ground-based population estimates.

Both distance sampling and ground-based censuses contain potential sources of bias that likely contributed to differences in population size estimates. Distance sampling relies on an accurate model of the detection function to calculate density, and better models are achieved with larger numbers of detections; the recommended number of detections to model the detection function adequately is > 60 detections (Buckland et al. 2001). Over four years of surveys, we detected nuthatches 63 times (breeding season) and bluebirds 51 times (non-breeding season), and thus for any single year the number of detections was exceedingly small. In response, we pooled detections among year to calculate the detection function at the expense of incorporating annual variation in detectability. Detectability does likely vary annually, due to breeding phenology, weather conditions, or observers. In this study, distance sampling appears to be overestimating detectability, but for reasons that are unknown to us.

Improving detection function models presents substantial challenges at current population levels of the reintroduced species. Detection functions should improve if the number of annual detections increases, as would be the case if the reintroduced population was growing. However, evidence suggests that only the Brown-headed Nuthatch population is exhibiting population growth > 1.0 (Lloyd et al. 2009). Other options for increasing the number of detections is through increasing the number of survey stations or revisiting stations multiple times during a season. Only the latter choice seems practical as increasing the number of survey stations to the existing network is limited by the amount of pineland habitat that is available for additional points. Even with multiple visits, achieving 60-80 detections per survey may be difficult until population sizes increase substantially.

The ground-based surveys do not account for detectability, and thus contain a source of bias. The ground based surveys assume that every individual bird that was present within the

3,940 ha of pine forest that served as our sampling area was also counted. That assumption was surely violated to some unknown extent and thus our surveys are likely underestimates of true population size. Detectability of individuals on the ground surveys are related to how elusive the species is, which may be influenced by abundance and how well the habitat is sampled. Given the size of our sampling area, both factors likely influenced detectability. Ground-based surveys were more intensive in the core area of the reintroduced populations where density was highest. This included the eastern portions of Long Pine Key bounded by Research road and the Main park road. Monitoring in the western fire blocks of A, B, C and areas further west were logistically more difficult to access and less time was spent surveying these areas. Although bluebirds appeared fairly restricted to our core area, nuthatches were found at low densities throughout Long Pine Key and we undoubtedly missed some portion of individuals. However, although some individuals may have been missed, we do not believe that ground-based estimates were 2-4 times lower than the actual population, which would be the case if the estimates from distance sampling were correct.

Distance sampling and ground-based population estimates were not in accord, but the trends of population estimates from distance sampling and ground based surveys were similar and of the same magnitude, indicating that both methods likely provide valuable information for monitoring. Results suggest distance sampling should not be considered for documenting population size, but that it is likely acceptable for trend monitoring. The advantage to distance sampling is that it estimates a parameter and its variance via a defined statistical model, whereas the ground-based survey is an index that is related to population size in an unknown fashion. Distance sampling is also substantially more cost-effective because one can monitor abundance and trends of many species. Ground based surveys are most valuable when populations are smaller, such as in the case of Eastern Bluebirds.

Long-term recommendations for monitoring reintroduced populations

Both distance sampling and ground-based surveys provide valuable information for population monitoring of the two reintroduced species in Long Pine Key. We recommend that distance sampling be employed for long-term monitoring of Brown-headed Nuthatches as long as population sizes remains fairly large (>50 individuals) and there is no indication of consistent declines over time or a reduction in their distribution across Long Pine Key. Although we currently can not use the method to estimate population size, evidence indicates that density

estimates determined from breeding season surveys track population levels. Further research to improve modeling the detection function for nuthatches would be helpful to long-term monitoring of nuthatches. In particular, information on understanding how factors, such as breeding phenology, influence detectability would be helpful.

For Eastern Bluebirds, we recommend that ground-based population estimates be continued until their population size and distribution increases and population models reveal sustained population growth. When species are rare, accurate population size estimates are important so that management actions can be quickly implemented in situations where population levels dip to critical levels. Curiously, ground-based surveys appeared to be more consistent with distance sampling estimates of population size in the winter, a time when many more detections of bluebirds are recorded. Further research investigating the use of non-breeding season counts to track population trends of Eastern Bluebirds is warranted.

Table 3-1. List of breeding-season pine rockland bird species and their potential to be monitored effectively based on the outcome of simulations for PIWA and BHNU. Table includes estimated density (95% confidence interval) and mean abundance (within 150 m) of breeding birds in pine rocklands at Long Pine Key, Everglades National Park, Florida, Densities were estimated using distance sampling, and data were collected at 104 points in Long Pine Key. Surveys were conducted between April and June during 2005-2008.

Species	Density (individuals ha ⁻¹)	Mean Abundance
Species likely to be monitored with desired precision if sampling is designed around BHNU		
Northern Cardinal	0.57 (0.51 – 0.65)	2.70
Red-bellied Woodpecker	0.24 (0.16 – 0.37)	2.55
Pine Warbler	0.21 (0.13 – 0.34)	2.27
Eastern Towhee	0.21 (0.14 – 0.33)	1.80
Great-crested Flycatcher	0.19 (0.11 – 0.30)	1.75
White-eyed Vireo	0.18 (0.11 – 0.29)	1.37
Downy Woodpecker	0.09 (0.05 – 0.17)	0.71
Carolina Wren	0.04 (0.03 – 0.06)	0.55
Northern Flicker	0.05 (0.04 – 0.07)	0.54
Northern Bobwhite	**	0.43
Eastern Meadowlark	0.03 (0.03 – 0.04)	0.38
Mourning Dove	0.03 (0.02 – 0.04)	0.35
American Crow	**	0.30
Common Yellowthroat	**	0.30
Brown-headed Nuthatch	0.06 (0.04 – 0.10)	0.29
Species that may be monitored with desired precision if sampling is designed around BHNU.		
Blue Jay	0.02 (0.01 – 0.03)	0.24
Common Grackle	0.01 (0.01 – 0.03)	0.19
Species that are unlikely to be monitored with desired precision if sampling is designed around BHNU.		
Northern Mockingbird	0.01 (0.01 – 0.03)	0.11
Eastern Bluebird	0.02 (0.01 – 0.09)	0.09

** Density estimate not calculated due to too few detections to model detection function.

Table 3-2. Estimated density (95% confidence interval) and mean abundance (within 150 m) of non-breeding birds in pine rocklands at Long Pine Key, Everglades National Park, Florida, Densities were estimated using distance sampling, and data were collected at 104 points in Long Pine Key. Surveys were conducted between April and June during 2005-2008. Species detected at fewer than 10% of points are not included.

Species	Density (individuals ha ⁻¹)	Mean Abundance
Species likely to be monitored with desired precision if sampling is designed around BHNU		
Yellow-rumped Warbler	3.04 (2.89 – 3.20)	28.87
Pine Warbler	0.74 (0.67 – 0.81)	8.21
Gray Catbird	1.07 (1.00 – 1.14)	8.04
Common Yellowthroat	0.66 (0.62 – 0.69)	4.18
American Robin	0.21 (0.18 – 0.25)	4.02
House Wren	0.42 (0.39 – 0.46)	3.13
Palm Warbler	0.26 (0.23 – 0.30)	2.17
Blue-gray Gnatcatcher	0.25 (0.24 – 0.28)	1.78
Red-bellied Woodpecker	0.20 (0.19 – 0.22)	1.62
Northern Cardinal	0.17 (0.15 – 0.19)	1.22
Eastern Phoebe	0.08 (0.07 – 0.10)	0.96
Eastern Towhee	0.11 (0.09 – 0.12)	0.75
Downy Woodpecker*	0.10 (0.09 – 0.11)	0.75
Northern Mockingbird	0.09 (0.08 – 0.11)	0.71
Brown-headed Nuthatch	0.05 (0.03 – 0.09)	0.56
White-eyed Vireo	0.06 (0.06 – 0.07)	0.47
Northern Flicker	0.06 (0.05 – 0.07)	0.43
Carolina Wren*	0.04 (0.03 – 0.05)	0.29
Species that may be monitored with desired precision if sampling is designed around BHNU.		
American Crow	**	0.25
Species that are unlikely to be monitored with desired precision if sampling is designed around BHNU.		
Blue Jay	0.02 (0.02 – 0.03)	0.19
Great-crested Flycatcher	0.02 (0.02 – 0.03)	0.17
Eastern Meadowlark	**	0.14
Eastern Bluebird	0.02 (0.01 – 0.05)	0.13

** Density estimate not calculated due to too few detections to model detection function.

Table 3-3. Summary of models of detection functions evaluated for Brown-headed Nuthatches at Long Pine Key, 2005-2008. Models are based 63 auditory detections during the breeding season, April - June.

Model	Pooled detection function?	Δ AIC	Average annual density ^a	Lower 95% CI	Upper 95% CI
Hazard rate/simple polynomial	Yes	0	0.064	0.042	0.099
Uniform/cosine	Yes	0.12	0.072	0.035	0.150
Half-normal/cosine	Yes	1.62	0.066	0.019	0.234

^a Average annual density is the mean of annual estimates of density, weighted by survey effort in each year.

Table 3-4. Estimated density (individuals ha⁻¹) and population size of Brown-headed Nuthatches during the breeding season (April – June) at Long Pine Key, 2005 – 2008.

Year	Density	Lower 95% CI	Upper 95% CI	Population size	Lower 95% CI	Upper 95% CI
2005	0.086	0.066	0.112	339	260	441
2006	0.065	0.050	0.085	256	197	335
2007	0.041	0.031	0.053	162	122	209
2008	0.065	0.050	0.085	256	197	335

* No ground-based surveys were conducted in 2008.

Table 3-5. Summary of models of detection functions evaluated for Brown-headed Nuthatches at Long Pine Key, 2005-2008. Models are based on 51 auditory detections during the non-breeding season, December - February.

Model	Pooled detection function?	Δ AIC	Average annual density ^a	Lower 95% CI	Upper 95% CI
Hazard rate/simple polynomial	Yes	0	0.044	0.023	0.085
Half-normal/cosine	Yes	4.73	0.044	0.009	0.215
Uniform/cosine	Yes	42.66	0.023	0.012	0.043

^a Average annual density is the mean of annual estimates of density, weighted by survey effort in each year.

Table 3-6. Estimated density (individuals ha⁻¹) of Brown-headed Nuthatches during the non-breeding season (December – February) 2005 – 2008.

Year	Density	Lower 95% CI	Upper 95% CI	Population size	Lower 95% CI	Upper 95% CI
2005	0.021	0.016	0.027	83	63	106
2006	0.045	0.035	0.059	177	138	232
2007	0.070	0.054	0.090	276	213	355
2008	0.042	0.032	0.054	165	126	213

* No ground-based surveys were conducted in 2008.

Table 3-7. Summary of models of detection functions evaluated for Eastern Bluebirds at Long Pine Key, 2005-2008. Models are based 24 auditory detections during the breeding season, April - June.

Model	Pooled detection function?	Δ AIC	Average annual density ^a	Lower 95% CI	Upper 95% CI
Uniform/cosine	Yes	0	0.015	0.008	0.031
Half-normal/cosine	Yes	0.38	0.018	0.009	0.037
Hazard rate/simple polynomial	Yes	0.49	0.013	0.006	0.027

^a Average annual density is the mean of annual estimates of density, weighted by survey effort in each year

Table 3-8. Estimated density (individuals ha⁻¹) of male Eastern Bluebirds and population size during the breeding season (April – June) at Long Pine Key, 2005 – 2008.

Year	Density	Lower 95% CI	Upper 95% CI	Population size	Lower 95% CI	Upper 95% CI
2005	0.010	0.008	0.013	39	32	51
2006	0.025	0.021	0.031	99	83	122
2007	0.015	0.012	0.019	59	47	75
2008	0.010	0.008	0.013	39	32	51

Table 3-9. Summary of models of detection functions evaluated for Eastern Bluebirds at Long Pine Key, 2005-2008. Models are based 51 auditory detections during the non-breeding season, December - February.

Model	Pooled detection function?	ΔAIC	Average annual density ^a	Lower 95% CI	Upper 95% CI
Hazard rate/simple polynomial	Yes	0	0.015	0.005	0.046
Uniform/cosine	Yes	0.58	0.016	0.005	0.052
Half-normal/cosine	Yes	0.74	0.024	0.008	0.074

^a Average annual density is the mean of annual estimates of density, weighted by survey effort in each year

Table 3-10. Estimated density (individuals ha⁻¹) of Eastern Bluebirds during the non-breeding season (December – February) 2005 – 2008.

Year	Density	Lower 95% CI	Upper 95% CI	Population size	Lower 95% CI	Upper 95% CI
2005	0.007	0.005	0.010	24	20	39
2006	0.013	0.010	0.020	59	39	87
2007	0.006	0.005	0.010	28	20	39
2008	0.032	0.022	0.046	126	83	185

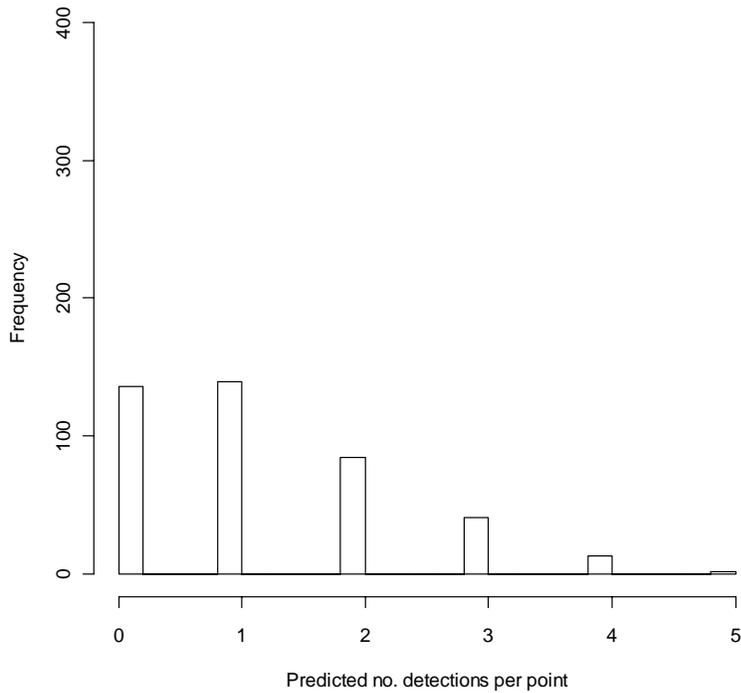
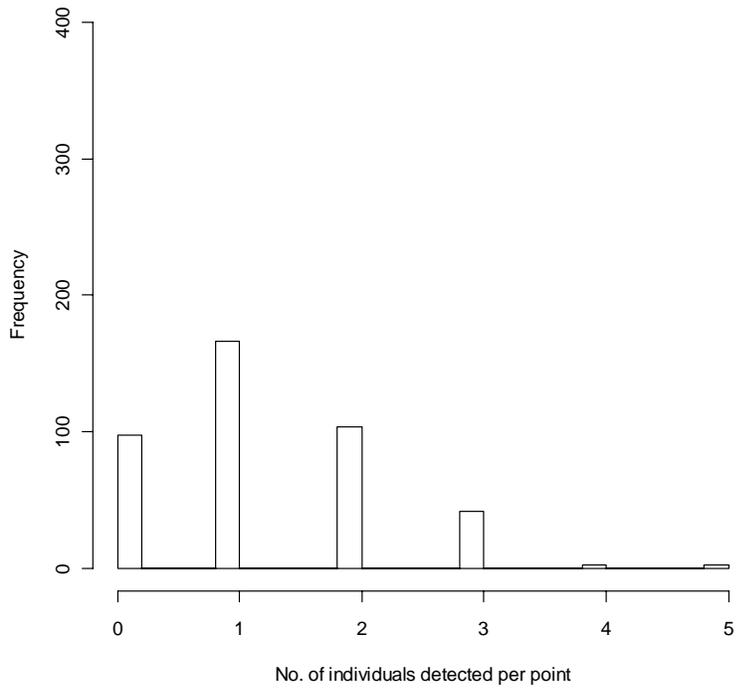


Figure 3-1. Observed and predicted distributions of Pine Warbler breeding season counts. Upper graph is observed counts for surveys at 104 points in Long Pine Key, ENP conducted in 2005-2008. Lower graph is predicted distribution assuming a Poisson distribution.

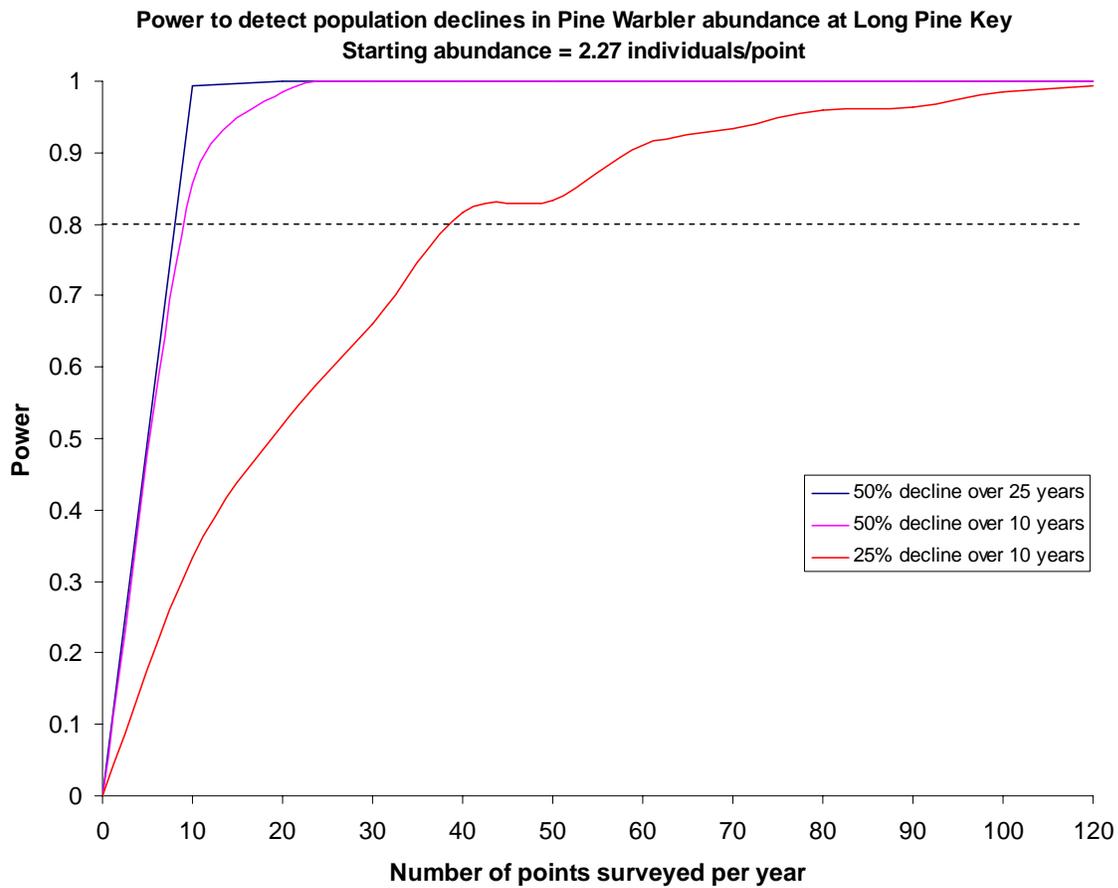


Figure 3-2. The influence of the number of survey points on power to detect a declining trend at three different effect sizes for the Pine Warbler. Results based on one-tailed tests with $\alpha = 0.10$.

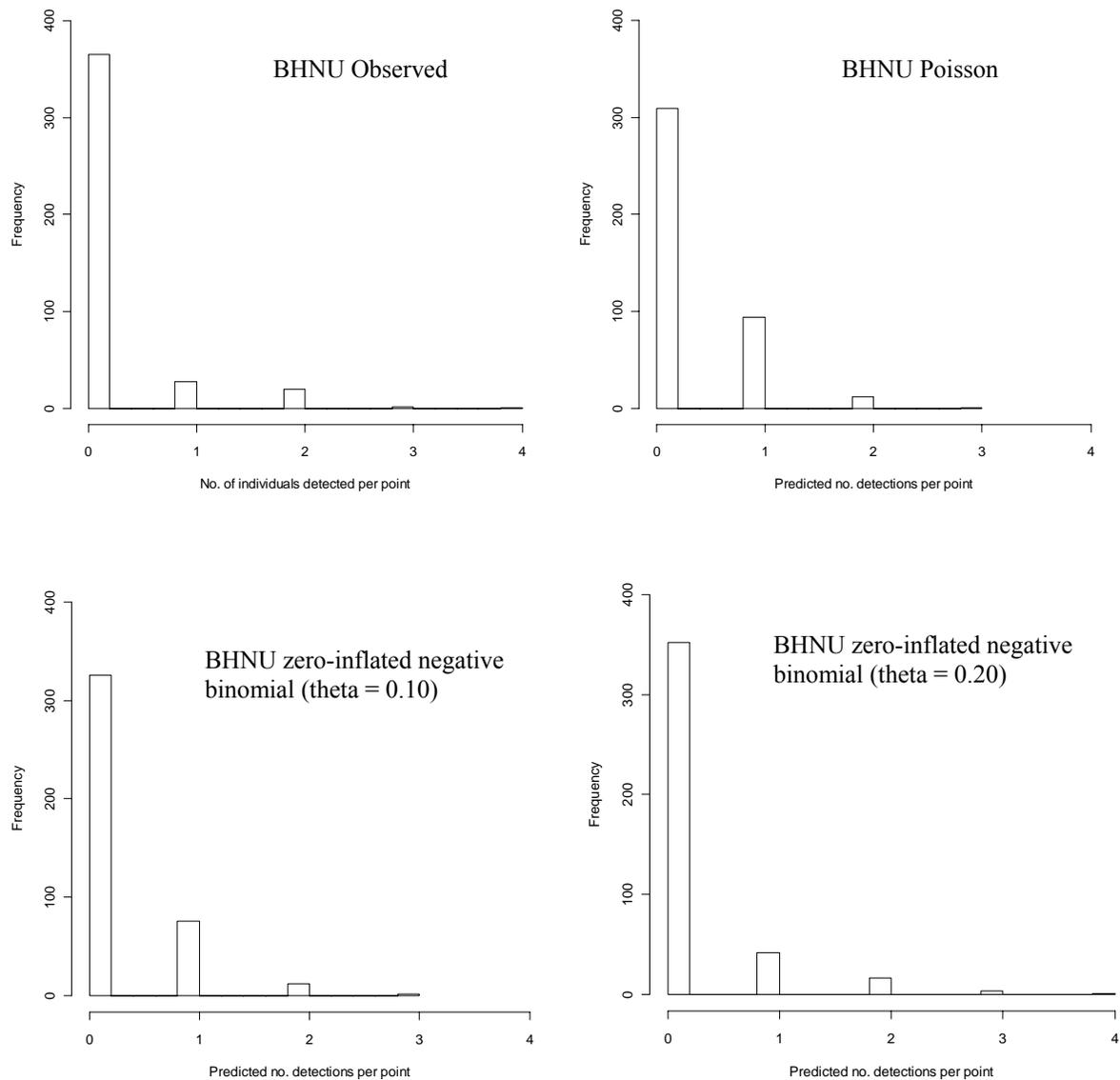


Figure 3-3. Observed and predicted distribution of Brown-headed Nuthatch breeding season counts. Upper left graph is observed counts for surveys at 104 points in Long Pine Key, ENP conducted in 2005-2008. Remaining graphs indicate a predicted distribution of Brown-headed Nuthatch breeding season counts assuming a Poisson (upper right), zero-inflated negative binomial distribution with a shape parameter (θ) = 0.10 (lower left), and a zero-inflated negative binomial distribution with a shape parameter (θ) = 0.20 (lower right).

**Power to detect population declines in Brown-headed Nuthatch abundance at Long Pine Key
Starting abundance = 0.29 individuals/point**

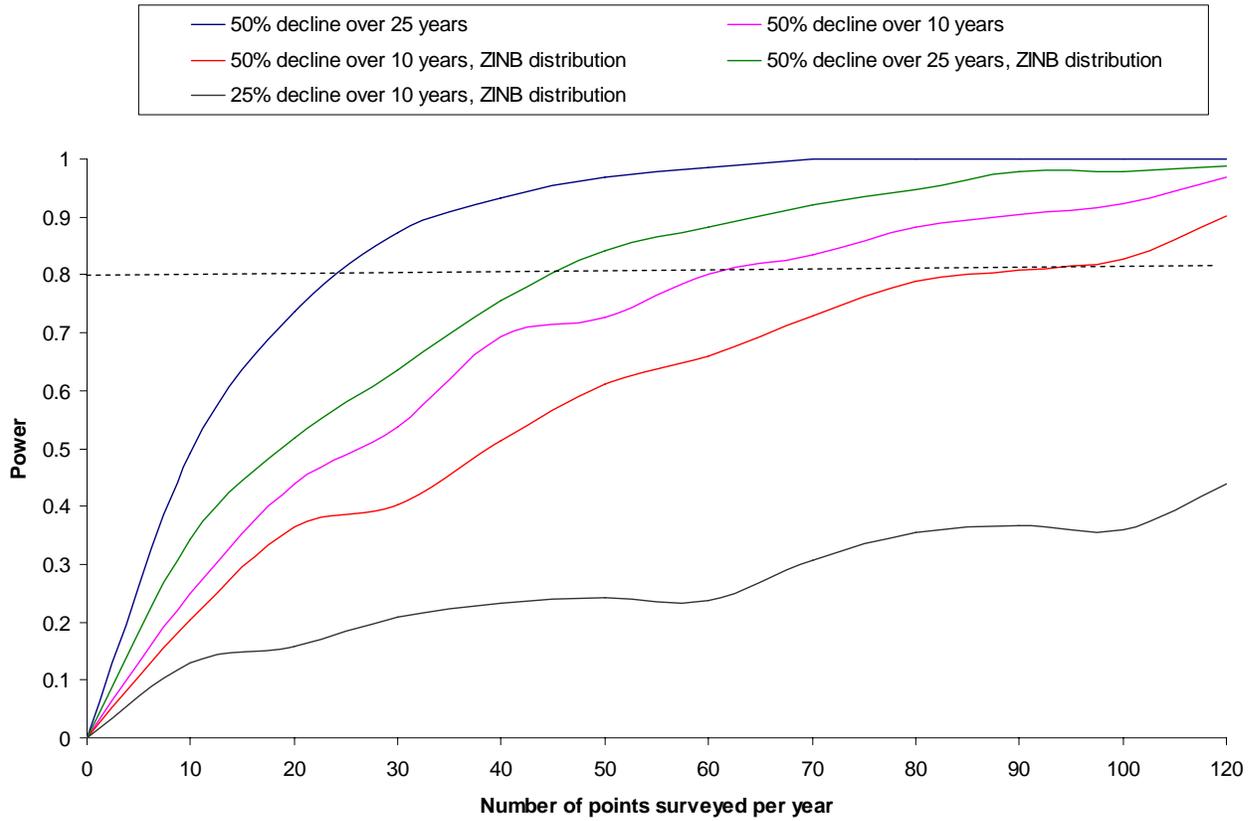


Figure 3-4. The influence of the number of survey points on power to detect a declining trend at three different effect sizes for the Pine Warbler. Results based on one-tailed tests with $\alpha = 0.10$.

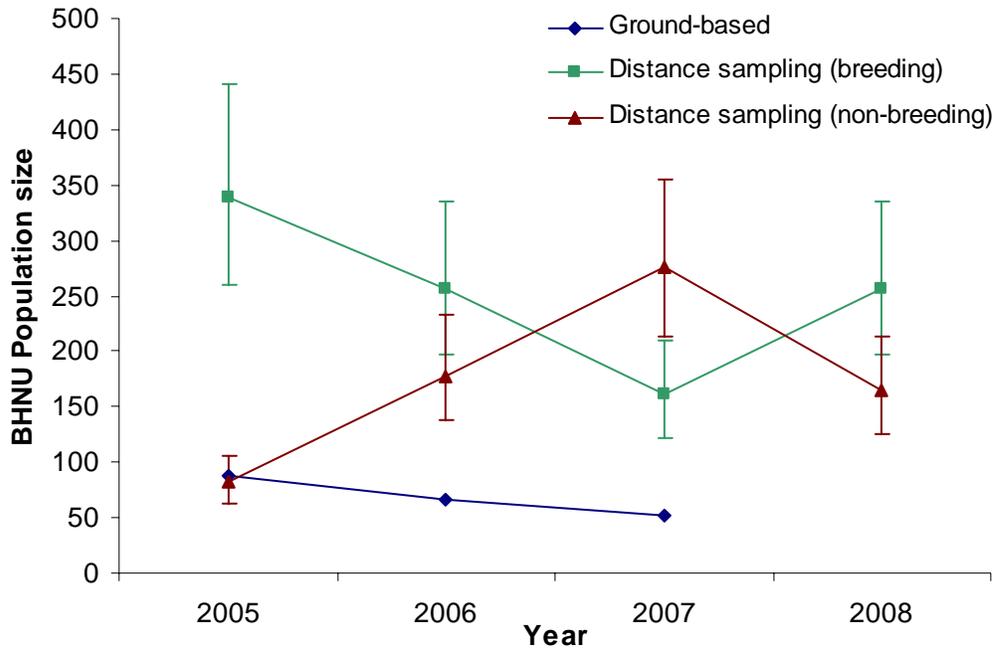


Figure 3-5. Comparison of Brown-headed Nuthatch population size estimated from breeding season ground-based surveys and distance sampling using breeding season and non-breeding season point count surveys conducted from 2005-2008. No ground-based surveys were conducted in 2008.

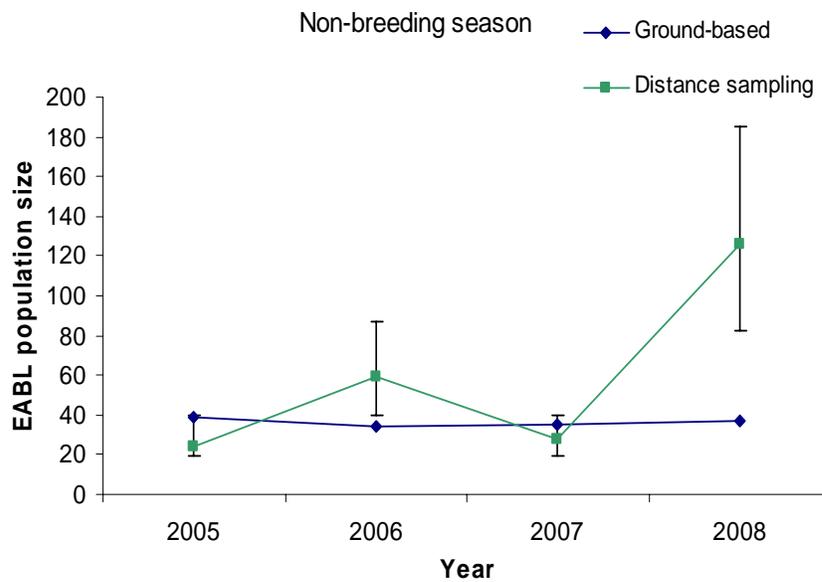
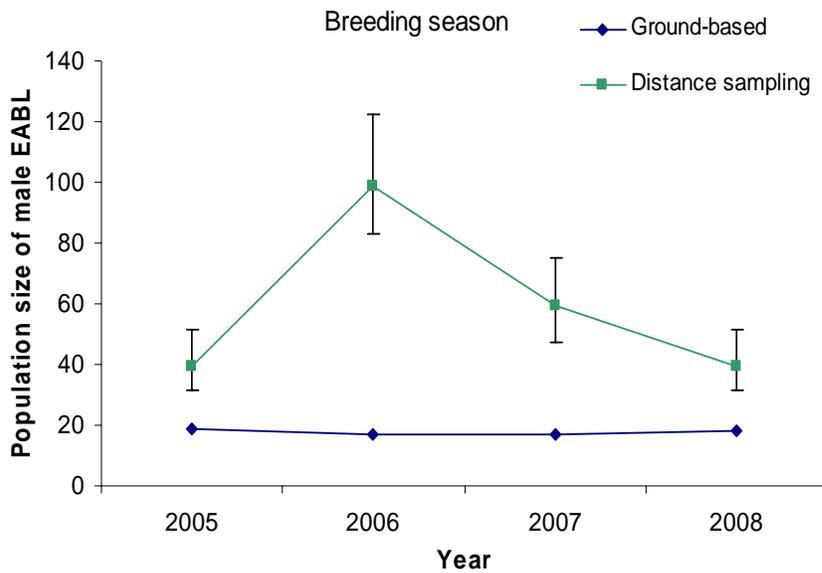


Figure 3-6. Comparison of Eastern Bluebird population size estimated from breeding season ground-based surveys and distance sampling using breeding season (upper) and non-breeding season (lower) point count surveys conducted from 2005-2008. Upper graph is population size of only males.

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Demography of Reintroduced Eastern Bluebirds and Brown-Headed Nuthatches

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ABSTRACT Species reintroductions are used commonly as a tool for conservation, but rigorous, quantitative assessments of their outcome rarely occur. Such assessments are critical for determining success of the reintroduction and for identifying management actions needed to ensure persistence of reintroduced populations. We collected 9 years of demographic data on populations of brown-headed nuthatches (*Sitta pusilla*) and Eastern bluebirds (*Sialia sialis*) reintroduced via translocation into Long Pine Key, Everglades National Park, Florida, USA. Realized population growth of brown-headed nuthatches was positive in the first 3 years after cessation of translocations ($\lambda_{2002} = 1.15$, SE = 0.13; $\lambda_{2003} = 1.28$, SE = 0.12; $\lambda_{2005} = 1.32$, SE = 0.20) but became negative thereafter ($\lambda_{2006} = 0.67$, SE = 0.10; $\lambda_{2007} = 0.77$, SE = 0.13). Realized growth rate for the Eastern bluebird population did not vary among years and indicated either a stable or a slowly declining population ($\lambda = 0.92$, SE = 0.04). Reintroductions were a qualified success; they resulted in the re-establishment of populations of both species, but neither population grew to the extent expected and both remained at risk of extinction. (JOURNAL OF WILDLIFE MANAGEMENT 73(6):955–964; 2009)

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KEY WORDS brown-headed nuthatch, Eastern bluebird, Everglades, population growth, populations, Pradel model, reintroduction, *Sialia sialis*, *Sitta pusilla*.

Despite the widespread use of reintroductions to re-establish populations of native species extirpated by habitat degradation or overexploitation (Tear et al. 1993, Wolf et al. 1996), rigorous, well-documented assessments of postreintroduction demography remain scarce (Fischer and Lindenmayer 2000). The failure to monitor demography of reintroduced populations has hindered identification of factors associated with reintroduction success and retarded progress in improving the success rate of species reintroductions (Scott and Carpenter 1987, Minckley 1995, Sarrazin and Barbault 1996, Fischer and Lindenmayer 2000). Detailed demographic studies can provide insight into basic and applied questions of population biology, including which management actions increase the likelihood of successful reintroductions.

In this study, conducted over 9 years, we assessed demography of populations of 2 bird species, Eastern bluebird (*Sialia sialis*) and brown-headed nuthatch (*Sitta pusilla*), that were reintroduced to Everglades National Park, Florida, USA. Both species were extirpated from Everglades National Park by the mid-1950s, part of a larger wave of local bird extinctions that was triggered by the widespread elimination and degradation of south Florida's pine (slash pine; *Pinus elliottii* var. *densa*) rockland ecosystem (Snyder et al. 1990). The reintroduction of Eastern bluebirds and brown-headed nuthatches was viewed as a test of the progress made in restoring this unique, fire-dependent ecosystem, with one measure of success being the ability to re-establish self-sustaining populations of extirpated species. Our objectives were to determine whether the reintroductions resulted in self-sustaining populations and to identify

management actions needed to ensure persistence of both populations.

STUDY AREA

We collected data on a population of each species reintroduced to Long Pine Key, Everglades National Park (25.3°N, 80.7°W). Long Pine Key is an 8,100-ha upland area, of which 4,600 ha is covered by pine rockland, a fire-dependent forest ecosystem restricted to limestone outcroppings in southern Florida and portions of Cuba and the Bahamas (Snyder et al. 1990). Long Pine Key is the largest remaining patch of pine rockland on the Atlantic coastal ridge. The dominant canopy species in Long Pine Key was south Florida slash pine. Other plant communities embedded within the pine forest included prairie, hardwood hammock, and cypress (*Taxodium* spp.) forest. The pine forest was even-aged as a result of extensive logging in the 1930s and 1940s, and snags were abundant due to widespread tree mortality associated with Hurricane Andrew in 1992. Beginning in the mid-1990s, Everglades National Park instituted an aggressive fire management program, with a 1–3-year fire-return interval, to reduce an overdeveloped shrub and palm understory, as well as high fuel loads that accumulated after years of fire suppression and Hurricane Andrew. In 2001, the goals of the fire management program shifted from restoration to maintenance, and the target fire-return interval was lengthened to 2–4 years.

METHODS

We first translocated brown-headed nuthatches and Eastern bluebirds to Long Pine Key during December 1997–February 1998. A preintroduction assessment estimated that Long Pine Key, which had been the focus of intensive

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efforts at ecosystem restoration, could support approximately 200 breeding pairs of both species (Slater 1997). This estimate of carrying capacity was based on mean nest densities (brown-headed nuthatches: 0.04 nests/ha; Eastern bluebirds: 0.04 nests/ha) at 2 sites in Big Cypress National Preserve (25.9°N, 80.9°W), including one from which we captured individuals for translocation, and the estimated amount of suitable habitat (4,600 ha) in Long Pine Key. The estimate of carrying capacity assumed that all pine forest in Long Pine Key was suitable for both species and that carrying capacity per unit area was the same as we observed in Big Cypress National Preserve. We obtained all brown-headed nuthatches and most (76%, $n = 47$) Eastern bluebirds used in translocations from the nearest source populations, which were located in Big Cypress National Preserve approximately 40 km from the reintroduction site. We captured the remaining Eastern bluebirds at golf courses in Naples, Florida (26.1°N, 81.8°W), approximately 140 km from the reintroduction site. Despite the proximity of the source populations, we found no evidence of natural recolonization in the 4 decades between extirpation of brown-headed nuthatches and Eastern bluebirds from Everglades National Park and the start of the reintroduction program. We captured most translocated birds on their territories and moved them as pairs (78% of brown-headed nuthatches and 76% of Eastern bluebirds), although we moved some cooperatively breeding brown-headed nuthatches as groups and some bluebird pairs with their nestlings. After capture, we transported pairs or groups to the reintroduction site, placed them in outdoor aviaries constructed in appropriate habitat, and provided them with ad libitum access to food and water. We kept Eastern bluebirds in aviaries for 1–3 weeks, except for 2 pairs that nested in an aviary, which we allowed to remain inside until their young left the nest. We released Eastern bluebird pairs with nestlings after the young had fledged and were capable of sustained flight. We kept brown-headed nuthatches in aviaries for 1–7 days. We conducted additional translocations each year during December–March (brown-headed nuthatch) and February–April (Eastern bluebird) until 2001.

We collected demographic data from the reintroduced populations in each of the breeding seasons from 1998 to 2007, excluding 2004, during which we collected no data. Thus, these data cover 4 years during which we translocated individuals to Long Pine Key and 5 years posttranslocation. We collected information on reproduction and population size by locating breeding territories through a combination of randomly located point-transect surveys, systematic playback surveys, and targeted playbacks of vocalizations in areas previously used by breeding pairs and in unoccupied habitat deemed suitable. The size of the area surveyed remained constant throughout the course of the study. We expended similar levels of survey effort in areas occupied by Eastern bluebirds and brown-headed nuthatches and in areas with no previous record of occupancy by either of the 2 study species to ensure that all individuals within the study area had a nonzero probability of detection. We conducted

point-transect surveys at 100 randomly located stations, with each station visited once between December and February and again between April and June of each year. We conducted systematic playback surveys by walking transects that were spaced at approximately 300-m intervals throughout Long Pine Key. An observer stopped every 100 m along each transect, broadcast a recorded vocalization of each species, and listened for responses. We did not survey portions of transects that crossed hardwood hammocks, because neither brown-headed nuthatches nor Eastern bluebirds use this forest type. We surveyed line transects twice each year. Finally, we also broadcast recorded vocalizations in areas where territories had been located in previous years and at the ecotone between glades and pine forest, along which bluebirds frequently nested. We conducted systematic playback surveys and targeted surveys during March–June of each year. Based on the estimated effective detection radius of the point-transect surveys, and assuming the effective detection radius for systematic and targeted surveys was the same as for point-transect surveys, we calculated that our effective survey area for both species was approximately 3,940 ha, or approximately 86% of the estimated extent of pine forest in Long Pine Key (M. S. Faherty, Ecostudies Institute, unpublished data).

We indexed the size of the adult population in each year by spot-mapping territories and counting the adults associated with the territory. Brown-headed nuthatches, which breed either as pairs or in cooperative groups of up to 5 individuals, maintain group territories year-round. Eastern bluebirds, although they remain in Long Pine Key year-round, did not maintain territories outside of the breeding season. Using counts of individuals on each territory that we located may have underestimated true population size because nonterritorial individuals may have gone undetected. In addition, an unknown amount of annual variation in our counts was due to variation in our ability to detect individuals in different years. For both of these reasons, our annual index of population size is best viewed as a minimum estimate of number of birds present in each year, and as a consequence we chose to base most of our inferences on results of the demographic analyses detailed below.

Once located, we monitored each territory consistently beginning in mid-February for evidence of breeding activity. Once we noted excavation and nest-building behaviors, we checked nest sites regularly until egg-laying began. During observations on brown-headed nuthatch territories, we counted number of adults present on the territory. We indexed size of the breeding group on each territory by observing (on >2 occasions) the number of adults participating in breeding activities such as cavity excavation, nest building, incubation feeding, or feeding of the nestlings. The extent of cooperative breeding by brown-headed nuthatches seems to be density-dependent, at least in south Florida, with cooperative breeding becoming more common as populations approach carrying capacity (Cox and Slater 2008). We postulated that average size of nuthatch breeding groups might reflect density-dependent pressures, with group size increasing as the population grew

and approached carrying capacity. We used linear regression to examine the relationship between our annual index of population size and our annual index of mean group size for brown-headed nuthatches. For both brown-headed nuthatches and Eastern bluebirds, we determined nest status every 3–5 days until nestlings fledged or the nest failed by using a pole-mounted video camera (Tree Top Peeper™ or Nuthatch Peeper System, Sandpiper Technologies, Inc., Manteca, CA) or by conducting behavioral observations at nests, usually for <30 minutes. To minimize disturbing the birds, we only used pole-mounted cameras on nests when a change of status was imminent (i.e., clutch completion, hatching, or fledging), and we observed activity at nests through binoculars from a distance of approximately 40 m. At this distance, we found that presence of an observer had no obvious effect on adult breeding behavior (e.g., adults did not abort incubation feeding or nestling feeding attempts).

We considered a nest successful if it fledged ≥ 1 nestling, and we calculated breeding productivity as number of young fledged per pair per year. We determined number of young fledged by conducting 2 visits to each territory after the date on which young left the nest. Young of both species remain with their parents for up to 1 month after departing the nest (Gowaty and Plissner 1998, Withgott and Smith 1998); however, our index of productivity reflected the minimum number fledged because some individuals that were alive may have gone undetected. As a consequence, our index of breeding productivity may underestimate actual number of young fledged per pair per year. We continued to monitor territories until mid-July to determine whether renesting occurred. We calculated an index of percentage of birds breeding each year by dividing total number of birds observed breeding by total number of birds counted in the population.

We attached colored leg bands to all translocated individuals. We also banded as many nestlings and juveniles as possible in each year of the study. However, in most years we did not capture all of the birds that fledged, because not all nests were accessible and not all juveniles were relocated after they left the nest. Thus, we also captured and banded unmarked adults that we discovered while monitoring nests or during annual population counts. We captured unbanded adults in mist nets, either by luring them to the net with recorded vocalizations or, if they were attending a nest, by setting the net outside of the nest cavity. For our analyses, we considered that any individual captured or resighted during the breeding season was alive in that year.

We examined posttranslocation demography using the reverse-time, capture–recapture models of Pradel (1996), as implemented by the Pradel survival and seniority model in Program MARK (White and Burnham 1999). In particular, we used this approach to estimate seniority probability (γ_{t+1}), which is the probability that an individual captured at time $t+1$ was a survivor from time t . This parameter also yields the probability that an individual was recruited to the population during the interval, or $1-\gamma_{t+1}$. Using the maximum-likelihood estimate of γ_{t+1} , based on the best-supported model from among a set of candidate models, we

derived the realized growth rate of the population (λ_t) and the per-capita rate of recruitment (f_t), or average number of new individuals added to the population between time t and time $t+1$ per individual already present in the population. We estimated derived parameters as follows:

$$\lambda_t = \frac{\phi_t}{\gamma_{t+1}} \quad \text{and}$$

$$f_t = \frac{\phi_t(1-\gamma_{t+1})}{\gamma_{t+1}}$$

(Franklin 2001). In addition, following Nichols et al. (2000), we viewed γ as an analog of elasticity of realized λ_t to ϕ and f . For example, $\gamma_{t+1} = 0.5$ would indicate that survivors from N_t and new recruits between time t and time $t+1$ made equal contributions to population growth over the interval; in contrast, $\gamma_{t+1} = 0.75$ would indicate that adult survival between time t and time $t+1$ was 3 times more important to population growth than recruitment over the interval (Nichols et al. 2000). We also examined elasticity of ϕ and f by calculating expected changes in λ_t as a function of proportional changes in ϕ and f (Nichols et al. 2000). We estimated standard error of derived parameters using the Delta method, and we calculated approximate 95% confidence intervals ($1.96 \times \text{SE}$) around the estimate of each derived parameter.

Our primary interest was postreintroduction demography, and we estimated γ_{t+1} and all derived parameters for the period 2001–2007, excluding 2004 when we collected no data. We adjusted parameter estimates to account for the unequal interval lengths by setting the length of the third interval in Program MARK to 2. We did not include in any analysis individuals that we never saw after translocation or that we saw once but that never established a breeding territory. We censored these individuals because of the bias associated with including transients in Cormack–Jolly–Seber (CJS) models (e.g., Johnston et al. 1997, Pradel et al. 1997) and because we were uncertain to what extent transience was induced by conditions at the reintroduction site versus the process of translocation itself, which was not of interest to us. Excluding transients should yield unbiased estimates of survival and capture rates for populations under study (Pradel et al. 1997). Initial attempts to estimate stage-specific (hatch-yr and ad) rates produced unreliable parameter estimates for hatch-year birds (e.g., survival and recapture rates of 0 or 1, with inestimable SEs), apparently because we had few individuals that we marked as juveniles (37 Eastern bluebirds and 19 brown-headed nuthatches). Thus, in subsequent analyses we estimated rates for the adult stage only. To do so, we used encounter histories from all individuals but censored the first survival interval for individuals banded in their hatch year. For example, we treated an individual born and marked in 2000 and resighted in 2001–2003 as if it had been marked as an adult for the first time in 2001.

We evaluated a candidate set that included 8 models: constant survival, recapture, and seniority probabilities ($\phi\rho\gamma$); time-specific variation in one rate ($\phi_{time}\rho\gamma$, $\phi\rho_{time}\gamma$,

and $\phi p \gamma_{time}$); time-specific variation in 2 rates ($\phi_{time} p_{time} \gamma$, $\phi p_{time} \gamma_{time}$, and $\phi_{time} p \gamma_{time}$); and time-specific variation in all rates ($\phi_{time} p_{time} \gamma_{time}$). We evaluated support for each model in the reverse-time, capture-recapture analysis using Akaike's Information Criterion, as adjusted for small sample size and extrabinomial variation (QAIC_c), and the quasi-likelihood-adjusted Akaike weights (w_i) for each model in the candidate set.

General goodness-of-fit tests for the Pradel survival and seniority models cannot be implemented in Program MARK, so we estimated the extent of extrabinomial variation (\hat{c}) from the CJS model $\phi_{time} p_{time}$ implemented in the live-recaptures module of Program MARK (Alisauskas et al. 2004). We used data from all years of the study, with transient individuals and hatch-year encounters censored as described for the Pradel survival and seniority analysis. We estimated \hat{c} by dividing observed \hat{c} from model $\phi_{time} p_{time}$ by the mean of 1,000 simulated values of \hat{c} generated using the parametric bootstrap routine in Program MARK (White and Burnham 1999). We then used this estimate of \hat{c} to adjust model likelihoods in the Pradel survival and seniority analysis. Likelihoods differ for the CJS model and Pradel survival and seniority model because they condition on different parts of the capture history; thus, the use of a goodness-of-fit metric derived from the CJS model is not strictly appropriate as a means to account for overdispersion in the Pradel survival and seniority model. However, we believed that this was the best available approach.

We also used live-recaptures modeling of data from the entire study period to address an important assumption of the reverse-time, capture-recapture analysis, namely, that probability of recapture did not vary as a function of capture history (Franklin 2001). Permanent trap responses in capture probability can bias estimates of λ_x , with trap-happy responses producing small (<0.01) to moderate (0.10) levels of positive bias in λ_x and trap-shy responses yielding a small, negative bias (Nichols and Hines 1999). To test for trap dependence in recapture probability, we first evaluated a set of models that included time-specific variation in adult survival and recapture probability ($\phi_{time} p_{time}$), time-specific variation in one rate ($\phi_{time} p$ or ϕ_{time}), and constant adult survival and recapture probability (ϕp). We evaluated support for each model using QAIC_c and w_i . We then added a parameter to the best-supported model (or best-supported models when the top models were within 2 QAIC_c values of one another) so that initial capture probability was modeled separately from subsequent recapture probabilities. We used the relative support for this trap-dependence model to test the assumption that birds did not exhibit permanent responses to trapping. If individuals showed a positive or negative response to having been trapped, then we expected more support for the model that estimated initial capture probability separately from subsequent recapture probabilities.

RESULTS

We released 47 adult brown-headed nuthatches into Long Pine Key between 1997 and 2001, 21 of which we never saw

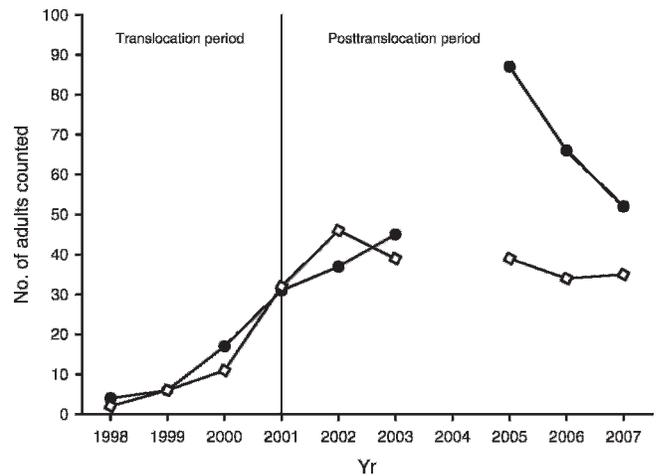


Figure 1. Number of documented breeding territories of brown-headed nuthatches (dark circles) and Eastern bluebirds (clear squares) reintroduced to Long Pine Key, Everglades National Park, Florida, USA, 1998–2007. We collected no data in 2004. The solid vertical line separates translocation and posttranslocation periods.

after release or we saw once but that never established a territory. Remaining birds each established a territory and were present for ≥ 1 year, and 16 of the 26 individuals that established a territory after translocation were present for >2 years. Annual counts of territorial adults suggested that the population grew rapidly after cessation of translocations but declined sharply from 2005 to 2007 (Fig. 1). Over the course of the study, we captured and banded 145 brown-headed nuthatches, including translocated birds. The proportion of marked individuals detected in our annual count was 100% from 1998 to 2002, but it declined to 87% in 2003 and then remained fairly constant from 2005 to 2007 (2005: 54%; 2006: 58%; and 2007: 54%). We translocated 62 Eastern bluebirds (47 ad and 15 nestlings) between 1997 and 2001. Of the 47 adults moved, we never saw 16 after release. Each of the remaining 31 established a territory. Only one of the individuals translocated as a nestling returned after fledging to establish a territory. Annual counts of territorial adults increased during the translocation period but declined gradually throughout the posttranslocation period (Fig. 1). We captured and banded 167 Eastern bluebirds during the study, including translocated individuals. The proportion of marked individuals in our annual count was 100% from 1998 to 2001, but it declined and then remained constant thereafter (2002: 85%; 2003: 85%; 2005: 62%; 2006: 88%; and 2007: 77%).

Approximately 75% of adult brown-headed nuthatches that we observed bred in a given year (95% CI = 61–90), and on average a breeding pair or breeding group fledged 1.9 young per year (95% CI = 1.6–2.3). Mean group size on each territory, averaged across years and territories, was 2.1 (range of annual means = 2.0–2.3), and average group size in a year was positively related to our index of population size in that year ($r = 0.90$, 95% CI = 0.55–0.98; Fig. 2). Renesting was rare among brown-headed nuthatches; only 9.1% ($n = 13$) of breeding pairs made 2 attempts in a year, and we had no evidence that brown-headed nuthatches ever

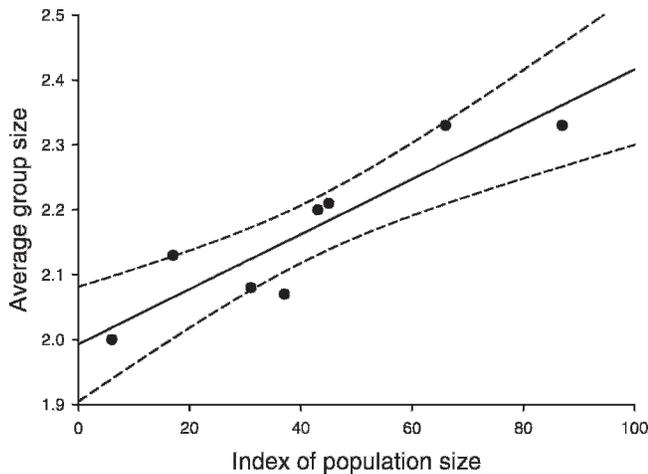


Figure 2. Average size of breeding groups of brown-headed nuthatches in Long Pine Key, Everglades National Park, Florida, USA, as a function of the number of adults counted in the population in each year from 1999 to 2007. Dotted lines are 95% confidence limits.

made >2 nesting attempts in a season. Most renesting attempts (77%, $n = 10$) occurred after nest failure.

Estimated \hat{c} from the most general model of survival ($\phi_{time}p_{time}$) was 1.52. Using data from all years, we found no evidence of a permanent trap response among brown-headed nuthatches (Table 1). Support was split between the time-specific survival and constant recapture model ($\phi_{time}p$) and the constant survival and recapture model (ϕp), and the other models received no support (Table 1). Based on the model $\phi_{time}p$, apparent annual adult survival was high throughout much of the study but dropped during the interval from 2005 to 2007 (Fig. 3). Recapture probability in this model was 0.96 (95% CI = 0.83–0.99). Apparent annual survival of adults under the other well-supported model, ϕp , was 0.63 (95% CI = 0.55–0.71) and recapture probability was 0.94 (95% CI = 0.79–0.98).

In considering reverse-time models, we found strong support for constant seniority and recapture probabilities and time-dependent survival probabilities from 2001 to 2007

Table 1. Candidate models explaining variation in apparent adult survival (ϕ) and capture probability (p) for brown-headed nuthatches in Everglades National Park, Florida, USA, from 1998 to 2007 (excluding 2004).

Model ^a	Model likelihood ^a	$\Delta QAIC_c$ ^{b,c}	w_i ^d	No. of parameters ^e
$\phi_{time}p$	191.0	0	0.44	10
ϕp	206.8	0.8	0.30	3
$\phi_{time}p_{trap\ dependence}$	191.0	2.2	0.14	11
$p_{trap\ dependence}$	206.8	2.9	0.11	4
ϕ_{time}	201.0	7.8	0.01	9
$\phi_{time}p_{time}$	189.0	11.8	0.00	16

^a $-(2 \ln(L)/\hat{c})$, where \hat{c} is a variance inflation factor calculated from the global model $\phi_{time}p_{time}$. For this model set, $\hat{c} = 1.52$.

^b $\Delta QAIC_c$ is the difference between the value of the quasi-likelihood Akaike's Information Criterion, with a small sample correction ($QAIC_c$), for the given model and the model with the lowest $QAIC_c$ score.

^c The lowest $QAIC_c$ score was 212.1.

^d $QAIC_c$ wt (w_i) reflects relative likelihood that the model is the best in the candidate set.

^e Includes an extra parameter for \hat{c} .

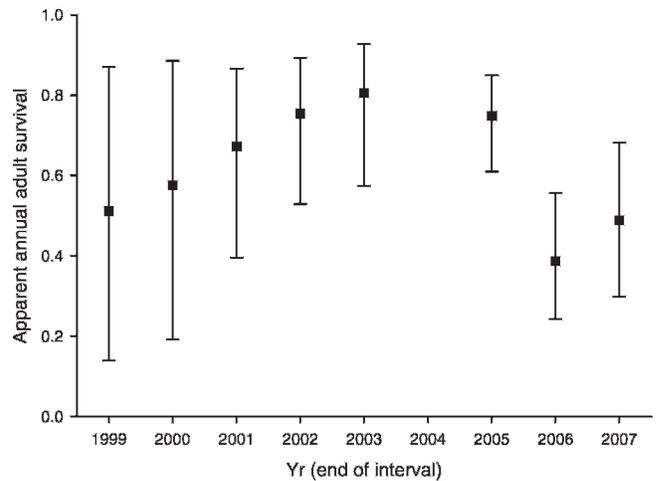


Figure 3. Apparent annual adult survival ($\pm 95\%$ CI) of brown-headed nuthatches reintroduced to Long Pine Key, Everglades National Park, Florida, USA. Translocations ended in 2001, and we collected no data in 2004. We estimated apparent survival from the best-fitting model in a candidate set, ϕp .

2007 (Table 2). Accordingly, we used model $\phi_{time}p\gamma$ to derive λ and f (Table 3). Estimated seniority probability (0.64; 95% CI = 0.57–0.72) was >0.5, which indicated that adult survival accounted for the most change in population size between years. Apparent adult survival was stable until the interval 2005–2007, when it dropped sharply. Estimates of λ_i (Table 3) indicated that the population grew from 2001 to 2005, with an especially large increase from 2002 to 2003, and then declined rapidly during both 2005–2006 and 2006–2007 as adult survival declined.

We used estimated seniority probability to examine the relative effect of hypothetical changes in adult survival on population growth during the 2 years in which estimated population growth was negative. For 2005–2006, we calculated that an increase in adult survival of 77% (from

Table 2. Candidate models explaining variation in apparent adult survival (ϕ), capture probability (p), and seniority probability (γ) for brown-headed nuthatches in Everglades National Park, Florida, during 2001–2007 (excluding 2004).

Model ^a	Model likelihood ^a	$\Delta QAIC_c$ ^{b,c}	w_i ^d	No. of parameters ^e
$\phi_{time}p\gamma$	398.8	0	0.92	8
$\phi_{time}p\gamma_{time}$	396.1	6.2	0.04	12
$\phi p\gamma$	415.3	8.0	0.02	4
$\phi_{time}p_{time}\gamma$	396.6	9.0	0.01	13
$\phi p_{time}\gamma$	407.8	11.2	0.01	9
$\phi_{time}p_{time}\gamma_{time}$	395.1	12.1	0.00	15
$\phi p\gamma_{time}$	411.4	12.6	0.00	8
$\phi p_{time}\gamma_{time}$	405.8	18.2	0.00	13

^a $-(2 \ln(L)/\hat{c})$, where \hat{c} is a variance inflation factor calculated from the global model in a Cormack-Jolly-Seber analysis, $\phi_{time}p_{time}$, for data collected from 1998 to 2007. For this model set, $\hat{c} = 1.52$.

^b $\Delta QAIC_c$ is the difference between the value of the quasi-likelihood Akaike's Information Criterion, with a small sample correction ($QAIC_c$), for the given model and the model with the lowest $QAIC_c$ score.

^c The lowest $QAIC_c$ score was 415.5.

^d $QAIC_c$ wt (w_i) reflects relative likelihood that the model is the best in the candidate set.

^e Includes an extra parameter for \hat{c} .

Table 3. Annual estimates of apparent adult survival (ϕ) and seniority probability (γ) for brown-headed nuthatches in Everglades National Park, Florida, during 2001–2007 (excluding 2004) estimated from the reverse-time, capture–recapture model $\phi\hat{p}\gamma$, and annual estimates of realized population growth (λ) and recruitment (f) derived from estimates of ϕ and γ .

Parameter	Estimated value	95% CI	
		Lower	Upper
γ	0.64	0.57	0.72
ϕ_{2002}	0.74	0.51	0.88
ϕ_{2003}	0.82	0.60	0.94
ϕ_{2005}	0.74	0.61	0.84
ϕ_{2006}	0.43	0.30	0.58
ϕ_{2007}	0.50	0.32	0.68
λ_{2002}	1.15	0.90	1.39
λ_{2003}	1.28	1.05	1.51
λ_{2005}	1.32	0.92	1.71
λ_{2006}	0.67	0.49	0.86
λ_{2007}	0.77	0.52	1.02
f_{2002}	0.41	0.28	0.54
f_{2003}	0.45	0.33	0.58
f_{2005}	0.41	0.34	0.48
f_{2006}	0.24	0.18	0.50
f_{2007}	0.27	0.19	0.36

the observed 0.43 to 0.76) would be required for $\lambda_{2006} = 1.0$. In 2006–2007, $\lambda_{2007} = 1.0$ would have been achieved by a 47% increase in apparent adult survival (from the observed 0.50 to 0.73). We did not use seniority probabilities to examine the response of population growth to changes in recruitment because of the deterministic relationship between adult survival and recruitment that was inherent in our best-fitting model. That is, with constant seniority probability and time-dependent survival probabilities, derived estimates of per-capita recruitment must necessarily track estimates of apparent survival. In this case, derived estimates of recruitment are a direct, linear function of apparent adult survival and therefore do not provide any additional, independent information about population dynamics.

Approximately 84% of adult Eastern bluebirds bred in a given year (95% CI = 80–88). Renesting was more common among Eastern bluebirds than brown-headed nuthatches, and 21% ($n = 56$) of all breeding pairs made 2 nest attempts in any given year. Five pairs made 3 nesting attempts in one breeding season. Most renesting attempts occurred after nest failure ($n = 43$, or 79% of all renesting attempts). Despite an increased propensity to reneest, Eastern bluebirds did not fledge more young than did brown-headed nuthatches, averaging only 1.6 young per year per pair (95% CI = 1.4–1.9).

We calculated $\hat{\epsilon} = 1.33$ using data from all years of the study. We found evidence of permanent trap responses among Eastern bluebirds, with $\phi\hat{p}_{trap\ dependence}$ receiving approximately twice as much support from the data as the reduced model $\phi\hat{p}$ (Table 4). According to the best model, initial capture probability of Eastern bluebirds (0.95; 95% CI = 0.78–0.99) was greater than subsequent recapture probability (0.71; 95% CI = 0.40–0.90), albeit with overlapping confidence intervals, indicating moderate trap-shyness. Thus, our estimate of λ may be slightly negatively

Table 4. Candidate models explaining variation in apparent adult survival (ϕ) and capture probability (\hat{p}) for Eastern bluebirds in Everglades National Park, Florida, USA, from 1998 to 2007 (excluding 2004).

Model ^a	Model likelihood ^a	ΔQAIC_c ^{b,c}	w_i ^d	No. of parameters ^e
$\phi\hat{p}_{trap\ dependence}$	209.3	0	0.64	4
$\phi\hat{p}$	212.7	1.3	0.33	3
$\phi_{time}\hat{p}$	204.5	8.3	0.01	10
$\phi\hat{p}_{time}$	204.7	8.4	0.01	10
$\phi_{time}\hat{p}_{time}$	194.4	12.3	0.01	16

^a $-(2 \ln(L)/\hat{\epsilon})$, where $\hat{\epsilon}$ is a variance inflation factor calculated from the global model $\phi_{time}\hat{p}_{time}$. For this model set, $\hat{\epsilon} = 1.33$.

^b ΔQAIC_c is the difference between the value of the quasi-likelihood Akaike's Information Criterion, with a small sample correction (QAIC_c), for the given model and the model with the lowest QAIC_c score.

^c The lowest QAIC_c score was 218.1.

^d QAIC_c wt (w_i) reflects relative likelihood that the model is the best in the candidate set.

^e Includes an extra parameter for $\hat{\epsilon}$.

biased. Apparent adult survival from 1998 to 2007, averaged across the top 2 models, was 0.62 (unconditional 95% CI = 0.50–0.74).

We found strong support for constant survival and seniority probabilities from 2001 to 2007, with the top 2 models differing only in whether recapture probability was constant (Table 5). Although the 2 best-supported models, $\phi\hat{p}\gamma$ and $\phi\hat{p}_{time}\gamma$, yielded nearly identical estimates for adult survival and seniority, we averaged parameter estimates across these 2 models and used unconditional estimates of standard error to account for model-selection uncertainty. The model-averaged estimate of seniority probability was 0.60 (95% CI = 0.50–0.69), suggesting that adult survival was slightly more important to population growth than recruitment. The model-averaged estimate of apparent survival was 0.55 (95% CI = 0.46–0.64), which was slightly lower than the estimate generated across all years of the study but within the estimated confidence interval. The

Table 5. Candidate models explaining variation in apparent adult survival (ϕ), capture probability (\hat{p}), and seniority probability (γ) for Eastern bluebirds in Everglades National Park, Florida, during 2001–2007 (excluding 2004).

Model ^a	Model likelihood ^a	ΔQAIC_c ^{b,c}	w_i ^d	No. of parameters ^e
$\phi\hat{p}\gamma$	432.2	0	0.64	4
$\phi\hat{p}_{time}\gamma$	423.5	2.1	0.23	9
$\phi\hat{p}'_{time}$	428.6	4.9	0.05	8
$\phi\hat{p}'_{time}\gamma_{time}$	417.4	5.1	0.05	13
$\phi_{time}\hat{p}'\gamma$	430.8	7.3	0.02	8
$\phi_{time}\hat{p}'_{time}\gamma_{time}$	416.5	9.0	0.01	15
$\phi_{time}\hat{p}'\gamma_{time}$	425.0	10.5	0.00	12
$\phi_{time}\hat{p}'_{time}\gamma$	423.2	10.9	0.00	13

^a $-(2 \ln(L)/\hat{\epsilon})$, where $\hat{\epsilon}$ is a variance inflation factor calculated from the global model in a Cormack–Jolly–Seber analysis, $\phi_{time}\hat{p}'_{time}$, for data collected from 1998 to 2007. For this model set, $\hat{\epsilon} = 1.33$.

^b ΔQAIC_c is the difference between the value of the quasi-likelihood Akaike's Information Criterion, with a small sample correction (QAIC_c), for the given model and the model with the lowest QAIC_c score.

^c The lowest QAIC_c score was 440.4.

^d QAIC_c wt (w_i) reflects relative likelihood that the model is the best in the candidate set.

^e Includes an extra parameter for $\hat{\epsilon}$.

model-averaged estimate for λ (0.92, 95% CI = 0.83–1.00) indicated that the reintroduced population of Eastern bluebirds was either stable or slowly declining from 2001–2007. Average annual per-capita recruitment to the Eastern bluebird population ($f = 0.37$, 95% CI = 0.33–0.41) was intermediate to levels estimated from the brown-headed nuthatch population. The estimated recapture probability from model $\phi p \gamma$ was 0.92 (95% CI = 0.76–0.98).

Using estimated seniority probabilities as an analogue to the elasticity of population growth rate to changes in adult survival and recruitment, we calculated that an increase in adult survival of 15% (from 0.55 to 0.70) would be required for $\lambda = 1.0$. At the same time, a 22% increase in the probability of recruitment ($1 - \gamma$; from 0.40 to 0.49) would be required for $\lambda = 1.0$. Substituting this value back into the formula used to calculate per-capita recruitment, f , we calculated that a 22% increase in the probability of recruitment would yield $f = 0.53$, an increase of approximately 43% over the observed $f = 0.37$. Assuming constant juvenile survival, and using the average number of young fledged per year in the posttranslocation period (1.4) as a measure of baseline productivity, a 43% increase in per-capita recruitment would be achieved by increasing the average number of young fledged per year to 2.0, which is within the range of observed values.

DISCUSSION

Although neither of the reintroduced populations grew to the extent predicted by the prereintroduction assessment, we consider the reintroduction of brown-headed nuthatches and Eastern bluebirds to Long Pine Key to be a qualified success. Armstrong and Seddon (2008) proposed that success of a reintroduction is the product of 2 discrete events: population establishment, in which population size increases from low numbers after reintroduction, and population persistence, or the ability to maintain, on average, a non-negative rate of population growth once carrying capacity has been reached. Brown-headed nuthatches and Eastern bluebirds continued to increase in number after translocations ceased, suggesting success in re-establishing populations at Long Pine Key. Less clear, however, is the ability of these populations to persist at Long Pine Key.

Insight into the likelihood of persistence may be gained by examining the possible causes of variation in postreintroduction population growth rate. In doing so, we made 2 assumptions concerning estimates of population growth rate. First, we assumed that estimates of realized λ applied to the entire population even though we included capture histories for adults only. This assumption is valid only if the age distribution is stationary or if most of the variation in population λ is due to variation in adult population size (Nichols et al. 2000). For the species that we studied, in which individuals enter the population of breeding adults within 6–9 months of birth, we suggest that the latter was the case and therefore that trends in adult λ closely approximated trends in population λ . Second, we assumed that immigration into Long Pine Key from other popula-

tions was absent or negligible and thus that changes in population size reflected processes operating within the reintroduced populations. Although we have no way to test this assumption, we believe that it is reasonable given the absence of documented records of brown-headed nuthatches or Eastern bluebirds in Long Pine Key in the 4 decades after their extirpation and preceding their reintroduction. Despite the persistence of nearby (e.g., the source populations for the reintroduction) populations, neither species was known to occur, even as vagrants, in Everglades National Park after their extirpation (see, e.g., Robertson et al. 1996). At the very least, we believe that this argues strongly that immigration played a negligible role in the dynamics of the reintroduced populations.

Given these assumptions, our interpretation of trends in postreintroduction demography depends in large part on the carrying capacity of Long Pine Key. If, as the prereintroduction assessment estimated, Long Pine Key can support 200 breeding territories of both species, then, 6 years after translocations ended, both species occurred at approximately 10% of their expected density (we counted 17 Eastern bluebird territories in 2007, and 23 brown-headed nuthatch territories). Failure of both species to show continued positive rates of population growth when existing at a small fraction of carrying capacity might indicate some systemic problem, such as inbreeding depression or Allee effects. A 6-year comparison (1998–2003) of vital rates between source populations, which we assumed had long-term rates of growth ≥ 1 , and reintroduced populations found higher rates of survival and reproduction in the reintroduced populations, suggesting their growth was not hindered by genetic influences on vital rates (G. L. Slater, Ecostudies Institute, unpublished data). The high proportion of individuals breeding each year at Long Pine Key suggests that Allee effects were not limiting growth of reintroduced populations.

The observed patterns of population growth might also reflect Long Pine Key's position at the southeastern edge of the geographic range of both species. Peripheral populations are generally small and isolated, as is the case at Long Pine Key, and individuals in peripheral populations often are poorly adapted to the rigors of their environment and thus sensitive to even slight variations in environmental conditions (reviewed in Brown et al. 1996). As a consequence, temporal variation in population size is much greater at the edge of a species' range than in the core, and peripheral populations may be more likely to exhibit boom-and-bust cycles in response to fluctuations in abiotic conditions (Thomas et al. 1994, Curnutt et al. 1996). For example, hurricanes—2 of which, Katrina and Wilma, struck Long Pine Key in 2005—may directly cause mortality or may produce indirect effects on survival and reproduction via changes in food availability (e.g., by stripping pine trees of their cones). Other density-independent factors unique to Long Pine Key may have been important. For example, Long Pine Key is bounded on 3 sides by paved roads, and between 1999 and 2006 ≥ 12 hatch-year Eastern bluebirds, which often forage on the grassy roadside verges, were killed

in collisions with motor vehicles. In sum, we cannot exclude the possibility that the patterns of population growth that we observed reflected the action of density-independent factors, which, in these peripheral populations, may limit the importance of density-dependent factors and reduce correlations between rates of population growth and expected carrying capacity.

Alternatively, the carrying capacity of Long Pine Key may have been overestimated during the preintroduction assessment, in which case patterns that we observed may have reflected populations that had reached carrying capacity, albeit at a level lower than expected or desired. The preintroduction assessment may have overestimated carrying capacity for several reasons. First, the estimate of territory density obtained from the source population was based on a small sample ($n = 25$ and $n = 23$ for brown-headed nuthatches and Eastern bluebirds, respectively) collected at 2 sites in Big Cypress National Preserve during one year (Slater 1997). However, estimates of Slater (1997) were nearly identical to those reported from a more extensive study of bird densities in slash pine forests (Land 1986, Land et al. 1989); thus, it seems unlikely that carrying capacity was overestimated because of bias in the underlying estimates of expected territory density.

Carrying capacity also may have been overestimated if, on average, habitat quality was lower at Long Pine Key, for example due to lower food abundance or increased abundance of predators or competitors. Based on point-transect surveys that we conducted between 2005 and 2008, abundance of important nest predators and competitors such as American crow (*Corvus brachyrhynchos*) and red-bellied woodpecker (*Melanerpes carolinensis*) was similar between Long Pine Key and Big Cypress National Preserve (J. D. Lloyd, Ecostudies Institute, unpublished data). We lack data on abundance of other potential predators of adults and young, such as snakes or raccoons (*Procyon lotor*), or any measures of food abundance, but trends towards higher survival and productivity at Long Pine Key suggest that, in general, conditions at the reintroduction site were roughly equivalent to those at Big Cypress National Preserve. However, variation in habitat quality may have been expressed through differences in unmeasured vital rates, such as juvenile survival.

Failure of either population to approach the expected carrying capacity of Long Pine Key also could reflect an overestimate of the amount, rather than quality, of habitat available. For example, despite efforts to impose a short fire-return interval across Long Pine Key, some areas still have dense hardwood understories, which both brown-headed nuthatches and Eastern bluebirds are known to avoid (Gowaty and Plissner 1998, Withgott and Smith 1998). Buildup of hardwood shrubs was most apparent at the ecotone between glades and pine forest, an environment which historically provided abundant nesting habitat for Eastern bluebirds. Thus, the assumption that all pine forest in Long Pine Key was suitable for both species may have been incorrect, leading to an overestimate of the number of individuals that could be supported. Under this scenario, the

patterns of population growth that we observed may have reflected density-dependent limits on population growth as carrying capacity was approached. Although clearly not definitive, the correlation between the average size of breeding groups of brown-headed nuthatches and population size suggests that breeding habitat may have become saturated during years of high abundance and that second-year birds were deferring breeding in favor of remaining on their natal territories as helpers. At the same time, the notion that large parts of Long Pine Key are unsuitable for either species is somewhat perplexing given that all of the pine forest in Long Pine Key has a nearly identical disturbance history (clearcut before the establishment of Everglades National Park and then subject to the same natural [e.g., hurricanes] and anthropogenic [e.g., prescribed fire] disturbances) and thus presumably affords homogenous environmental conditions.

Our data do not allow for a reliable test of any of the preceding hypotheses. Nonetheless, action to increase the size of both populations, which were small enough to remain at risk of extinction from stochastic factors, is warranted. Given the uncertainty about factors that limit reintroduced populations, we recommend a bet-hedging strategy in which long-term management to increase suitable habitat in Long Pine Key is combined with shorter-term actions to relax potential environmental limits on key vital rates. Over the long-term, increases in the extent of breeding habitat may be achieved by continued application of prescribed fire with short return intervals (i.e., 2–3 years), especially along the ecotone between pine forest and glades and in other areas where hardwood shrubs remain dense. However, short fire-return intervals may increase the rate of snag consumption relative to the rate of snag creation, and thus Eastern bluebirds and brown-headed nuthatches, both of which nest in cavities in snags, may benefit from longer fire-return intervals (i.e., >3 years) in areas that have been burned frequently in the past. Overly frequent fire may ultimately reduce habitat availability by reducing density of large snags. Although we have no strong evidence that populations at Long Pine Key are limited by poor survival or reproduction, it may be prudent to undertake short-term measures to boost vital rates and increase population size. Apparent survival of adults had the proportionally greatest influence on population growth rate, but manipulating adult survival or emigration rates in the short term is not feasible. As such, we recommend efforts to increase breeding productivity by placing aluminum flashing above and below occupied cavities, a technique that has shown promise in excluding potential nest predators (e.g., Loeb 1996). Temporarily erecting nest boxes may also help maintain the reintroduced population of Eastern bluebirds until additional habitat becomes available. Finally, anecdotal evidence exists that juvenile survival of Eastern bluebirds might be increased via temporary reductions in speed limits on roads adjacent to Long Pine Key. During the breeding season of 2008, Everglades National Park erected temporary warning signs on one of the roads adjacent to Long Pine Key advising motorists that the area was a “bluebird

crossing” and that vehicle speed should be reduced. Although we have no data regarding efficacy of the warning signs in reducing average vehicle speed, we did not document any Eastern bluebird mortality on that road in 2008, making it the first year since 1999 without ≥ 1 case of road mortality. As of fall 2008, Everglades National Park also reduced the posted speed limit on this road from 72 km per hour to 56 km per hour. Further study of whether warning signs and lowered speed limits reduce mortality of juvenile Eastern bluebirds is warranted.

The re-establishment of brown-headed nuthatch and Eastern bluebird populations in Long Pine Key revealed progress in restoring the pine rockland ecosystem within Everglades National Park. Occupied areas seemed to support levels of survival and reproduction sufficiently high to maintain stable populations, although we could not rule out declines in the Eastern bluebird population. However, neither population was demonstrably secure at the end of the study and continued monitoring and management of both populations is warranted. Continued restoration efforts are needed to create new breeding habitat in Long Pine Key and allow populations of both species to grow to levels that will increase likelihood of long-term persistence.

Management Implications

Management for both species should focus on using prescribed fire to provide open forest with abundant snags (Lloyd and Slater 2007). Short-interval (1–3 years) fires are suitable in areas with dense hardwood understories, but longer return intervals should be considered in other areas so as to optimize the balance between snag creation and snag consumption. We also recommend short-term efforts to boost vital rates, especially for Eastern bluebirds, as a hedge against extinction risk.

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Environmental Factors Affecting Productivity of Brown-Headed Nuthatches

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ABSTRACT Understanding the link between habitat use and components of fitness can yield useful insight into the environmental conditions necessary for population maintenance and can help promote effective habitat management. This information is especially important for species that are in decline or otherwise of conservation concern. Populations of brown-headed nuthatches (*Sitta pusilla*), an obligate cavity nester, have declined throughout their range, primarily due to extensive habitat loss and degradation. To help guide habitat management for this species, we identified habitat features associated with variation in the number of offspring fledged within 2 populations in southern Florida, USA. The most important predictor of productivity was the date on which a nest attempt began, with earlier nests producing more fledglings. The number of large pine (*Pinus elliottii* var. *densa*) snags and, to a lesser extent, the number of small pine trees surrounding a nest site were positively associated with productivity. We recommend that land managers in southern Florida focus on providing abundant large pine snags because doing so will increase productivity and also may increase nest-site availability and the percentage of individuals that breed each year. Prescribed burning may be an effective way to increase the abundance of large pine snags; however, land managers should exercise caution when doing so because of the trade-off between snag recruitment and snag consumption that accompanies the use of fire. We lack the data required to predict the fire-return interval that optimizes this trade-off, but until these data are available we recommend increasing the spatial heterogeneity in fire-return interval and lengthening the fire-return interval in some areas to 5–6 years. (JOURNAL OF WILDLIFE MANAGEMENT 71(6):1968–1975; 2007)

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Habitat selection results in nonrandom, species-specific patterns of resource use. Within a species, variation among individuals in the use of habitat features often is associated with variation in components of fitness (Fretwell and Lucas 1970, Morris 1991, Badyaev 1995, Martin 1998, Murphy 2001). Consequently, examining the relationship between habitat use and fitness components can yield important insight into the environmental conditions necessary for population maintenance (Martin 1992). This information can be used to develop management strategies for plant or animal populations.

Identifying appropriate features for habitat management is especially important for species that are in decline or otherwise of conservation concern. The brown-headed nuthatch (*Sitta pusilla*), a cavity-nesting species of the open pine (*Pinus* spp.) forests of the southeastern United States, has experienced significant long-term population declines (Sauer et al. 2005) that have led to local extinctions (Withgott and Smith 1998) and is a species of conservation concern for the United States Fish and Wildlife Service (2002). Habitat degradation is thought to be the primary cause of population declines (Withgott and Smith 1998). Patterns of habitat use by brown-headed nuthatches are relatively well described; abundance is higher in older pine forests with open understories, relatively large trees, and numerous snags (Conner et al. 1983, O'Halloran and Conner 1987, Wilson et al. 1995, Slater 1997, Wilson and Watts 1999), but little is known of how variation in habitat affects survival or reproduction. However, the success of efforts to restore and maintain high-quality habitat for

brown-headed nuthatches requires an understanding of how vital rates are likely to respond to habitat management, knowledge that can only be gained through direct examinations of reproduction or survival.

We examined how variation in vegetation structure at the nest site (density of small and large pine trees, density of pine snags, and % cover of hardwood shrubs), fire history, and hydrological conditions affected productivity of brown-headed nuthatches in the pine rocklands of southern Florida, USA. The objectives of this study were 2-fold: to identify habitat features that were associated with variation in productivity and to predict how variation in these habitat features would affect productivity. In addressing these objectives, we sought to provide land managers with information on how habitat can be manipulated to improve productivity and, in turn, stem the widespread population declines of this species.

STUDY AREA

We collected data on 2 brown-headed nuthatch populations: a population reintroduced in 1997 to Long Pine Key, Everglades National Park (25.3°N, 80.7°W), and the source population for the reintroduction, 40 km away in Raccoon Point, Big Cypress National Preserve (25.9°N, 80.9°W). Both sites were located in the pine rocklands, a fire-dependent, savannah-like ecosystem restricted to southern Florida and portions of Cuba and the Bahamas (Snyder et al. 1990). Pine rocklands were dominated by a single canopy species, South Florida slash pine (*Pinus elliottii* var. *densa*), and typically had an open understory supporting a diverse mix of tropical hardwoods, palms, and herbaceous plants. Average annual precipitation in this region was 130–150 cm,

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with approximately 80% of rainfall occurring during the wet season, typically May–October (Snyder et al. 1990).

Long Pine Key was an 8,100-ha upland area within Everglades National Park that contained approximately 4,600 ha of pine forest (Snyder 1986). Long Pine Key was the only remaining area of pine rockland in Everglades National Park. Other plant communities embedded within the relatively continuous pine forest included *Muhlenbergia* prairie, hardwood hammock, and cypress (*Taxodium* spp.) forest. The pine forest was even-aged, a result of extensive logging in the 1930s and 1940s, and snags were abundant due to the widespread tree mortality associated with Hurricane Andrew in 1992. Since the mid-1990s, Everglades National Park has instituted an aggressive fire management program to reduce an overdeveloped shrub and palm (*Sabal palmetto* and *Serenoa repens*) understory and the resultant high fuel loads that have arisen after years of fire suppression and Hurricane Andrew. The overall target for fire-return intervals during this study was 3 years, although most areas were burned more frequently during the early years of the study in an effort to reduce high fuel loads that accumulated after Hurricane Andrew.

Raccoon Point contained approximately 9,000 ha of unlogged, old-growth pine forest within a cypress mosaic. Since 1992, a fire program that mimics the natural timing and frequency of fire has been maintained, with a fire-return interval of 3–6 years. This fire regime, coupled with the site's hydric condition, has resulted in a well-developed herbaceous understory with a moderate amount of saw palmetto (*Serenoa repens*) and hardwoods. Brown-headed nuthatches were moderately abundant at this site (1.1 individuals/10 ha [Slater 1997]; median abundance from 12 studies summarized in Withgott and Smith 1998 = 1.2 individuals/10 ha).

METHODS

We collected data in each of the breeding seasons from 1998 to 2003: 4 years during which individuals were translocated from Raccoon Point to Long Pine Key and 2 years posttranslocation. In Long Pine Key, we located brown-headed nuthatch breeding territories by walking systematic transects and using playback vocalizations in areas where individuals were released and, in subsequent years, in territories that had been established postrelease. As the population grew, we also conducted surveys in unoccupied areas where habitat appeared suitable. We used similar methods to locate nests within suitable habitat (i.e., patches of pine forests) at Raccoon Point. We initiated nest searches in both areas in mid-February, when individuals typically begin excavating nest cavities. However, at both sites, we also took advantage of the year-round territoriality of this species by conducting observations throughout the winter to ensure that we did not miss any breeding attempts.

Once we noted excavation and nest-building behaviors, an observer checked nest sites regularly until egg laying began. We defined a nesting attempt as beginning upon completion of the clutch, as indicated by incubation behavior. We

observed nests, usually for a 30-minute observation period, to determine status every 3–5 days until nestlings fledged or the nest failed. To avoid disturbing the birds, we observed activity at nests through binoculars from a distance of approximately 40 m. At this distance, we found that the presence of an observer had no obvious effect on adult breeding behavior (e.g., adults did not abort incubation feeding or nestling feeding attempts). Most adults in both populations were color banded, and thus we were able to follow individuals throughout the breeding season to determine if renesting occurred. Renesting was relatively rare, accounting for only 9.1% ($n = 13$) of all observed nesting attempts, and no pairs were known to make more than 2 nesting attempts in a season. Most renesting attempts (77%; $n = 10$) occurred following nest failure. We excluded renesting attempts from this analysis because they generally took place close to the original nest cavity, or very rarely in the same nest cavity, and thus may not reflect an independent trial of the effect of habitat on breeding productivity. We considered nesting attempts in the same territory in different years to be independent because brown-headed nuthatches almost always excavate new cavities in different trees (Withgott and Smith 1998).

We calculated productivity as the number of young fledged per nesting attempt. We focused on productivity because mark–recapture estimates of apparent adult survival did not vary among years, whereas productivity was highly variable among years and was a strong predictor of population size in the following year (G. L. Slater, Ecostudies Institute, unpublished data). Thus, it is a suitable metric for identifying habitat features that may affect population growth rate. We are confident that we found all nests within the study areas and that our estimates of productivity are unbiased: population sizes were small, most adults were color banded (77% [$n = 103$] of nest attempts had ≥ 1 banded bird), individuals are sedentary and occupy year-round territories, and we monitored territories intensively throughout much of the year. Because we found nests early in the nesting cycle—we located nearly all nests during cavity excavation or nest construction—our sample is not biased towards successful nests.

We examined variation in productivity as a function of vegetation structure around the nest site, fire-return interval, and hydrology. We chose to examine the effect of these variables on productivity because previous studies had shown them to be important in nest-site selection, or because our experience in this system suggested they might affect productivity. We did not consider characteristics of the nest tree or nest cavity (e.g., orientation or ht) because previous work in this system indicated that these variables were not useful in describing patterns of habitat selection or in distinguishing successful and unsuccessful nests (Slater 1997). We also chose to exclude landscape-level factors (e.g., patch size or distance to habitat edge) that may have influenced productivity because we were primarily interested in identifying habitat features that are amenable to management. Most of the variation in landscape-level features in

our study area arises from the naturally patchy distribution of plant communities in this ecosystem and is therefore not a suitable target for management.

We quantified vegetation structure around the nest site by measuring the following variables: the number of large pines (≥ 15 cm dbh), small pines (< 15 cm dbh), and large pine snags (≥ 15 cm dbh) within an 11.3-m radius circle around the nest tree and the percent cover of hardwood shrubs (defined as all hardwoods 1.5–5 m above the ground), estimated visually, within a 1.8-m radius circle around the nest tree. We determined the number of fires within the past 5 years at each nest site using fire-history data obtained from Big Cypress National Preserve and Everglades National Park (National Park Service 2005a, b). We included 2 hydrology variables: mean water depth (i.e., stage ht minus elevation at the nest tree) on each territory during the winter prior to the breeding season (Dec and Jan) and during the breeding season (Mar–May). We estimated mean water depth for each period and each territory by determining mean daily stage height from 4 gauging stations surrounding Long Pine Key and one gauging station in Raccoon Point, and subtracting that value from the elevation at the nest tree. We determined the average elevation of each territory from elevation grids created from the United States Geological Survey (USGS) high-accuracy elevation data program (USGS 2006). In addition to the habitat variables, we also examined whether productivity varied as a function of the date on which incubation began, because seasonal declines in reproductive success are commonly reported for birds (Nilsson 1989, Hochachka 1990). We attempted to capture additional unexplained variation by including a dummy variable for year in our models.

Assuming that previously documented habitat preferences were adaptive, we expected a positive relationship between productivity and the number of large pines and large pine snags around the nest site. We also predicted a negative relationship between productivity and the number of small pines and the percent cover of hardwood shrubs. Furthermore, because the abundance of large snags and the density of mid- and understory vegetation are affected by the frequency of fire (Conner 1981, Wilson and Watts 1999), we expected a positive relationship between fire-return interval and productivity. Finally, although no previous study has examined links between hydrology and habitat selection in brown-headed nuthatches, we predicted that hydrology might have direct and indirect effects on food availability, which might in turn influence breeding productivity. For example, South Florida slash pine utilize relatively deep groundwater sources (Ish-Shalom et al. 1992) and thus seed production might be reduced when water levels are low, in turn reducing food availability and productivity of brown-headed nuthatches. Exceptionally low water levels might also reduce the abundance of insects (Wolda 1978), another important food source for brown-headed nuthatches.

We analyzed the relationship between productivity and habitat features using an information-theoretic, model-

Table 1. Candidate models explaining variation in productivity of brown-headed nuthatches in southern Florida, USA, from 1998 to 2003.

Model ^{a,b}	Model no.
Yr + date	1
Yr + date + fire frequency	2
Yr + date + no. of large pine snags	3
Yr + date + overstory nest-site vegetation	4
Yr + date + understory nest-site vegetation	5
Yr + date + hydrology + fire frequency	6
Yr + date + all nest-site vegetation	7
Yr + date + all nest-site vegetation + fire frequency	8
Yr + date + all nest-site vegetation + hydrology	9
Global	10

^a The following variables were considered: yr, date that incubation began (date), no. of small pines (small pines), no. of large pines (large pines), % cover of hardwood shrubs (shrub cover), no. of large pine snags (large pine snags), 5-yr fire frequency, and water levels during the breeding (Mar–May) season and winter (Dec and Jan).

^b All nest-site vegetation is small pines, large pines, shrub cover, and large pine snags; hydrology is breeding season and winter water levels; understory nest-site vegetation is small pines and shrub cover; and overstory nest-site vegetation is large pines and large pine snags.

selection approach (Burnham and Anderson 1998). We created a candidate set of 10 models, based on linear combinations of variables, that we believed could reasonably explain variation in productivity based on our knowledge of the ecology of brown-headed nuthatches (Table 1). Every model contained a term for year and a term for date of incubation, and the first model contained only these terms. We included this most simple model to evaluate the degree to which adding information about habitat conditions improved the predictive power of our models. The next 2 models that we evaluated were also simple: one included a term for 5-year fire frequency (model 2), which we believed might influence productivity of brown-headed nuthatches through its effect on the plant community, and the other included a term for the number of large pine snags (model 3), which, as the factor limiting the availability of nest sites, might be sufficient by itself to explain variation in productivity. From these simplified models, we next evaluated models that considered productivity as a function of the 2 main elements through which we quantified nest-site vegetation: overstory vegetation (no. of large pines and large pine snags; model 4) and understory vegetation (hardwood shrub cover and no. of small pines; model 5). We evaluated these models separately to determine whether a more simplified description of nest-site vegetation might adequately explain variation in productivity. The next model (model 6) explained variation in productivity as a function of fire frequency, which can influence all of the vegetation variables that we considered important, and hydrology, which might modulate the effect that fire has on plant communities (e.g., Lockwood et al. 2003). In essence, this model suggested that broad categorical descriptions of habitat conditions (i.e., fire-return interval and \bar{x} water level) could adequately capture the same variation described by our more detailed vegetation measurements. The next 3 models (models 7, 8, and 9) started with all of our nest-site vegetation measurements and added, respectively, effects of

Table 2. Observed values for habitat variables included in models of productivity for brown-headed nuthatches in southern Florida, USA, from 1998 to 2003.

Variable	\bar{x}	Range
No. of large pines ^a	5.7	0, 19
No. of large pine snags ^a	2.3	0, 11
No. of small pines ^a	1.4	0, 17
% cover by hardwood shrubs ^b	3.3	0, 70
No. of fires in previous 5 yr	1.4	0, 4
Water stage ht (m)		
Breeding season (Mar–May)	−0.7	−1.5, −0.2
Winter (Dec–Jan)	−0.3	−1.1, 0.6

^a Measured in a 11.3-m-radius circle around the nest.

^b Measured in a 1.8-m-radius circle around the nest.

fire and hydrology. We added the effects of fire and hydrology to models 8 and 9 because we were uncertain whether these terms would provide any additional information not already provided by the nest-site vegetation measurements. Finally, we tested a global model that included all of our variables.

We applied the following transformations to meet the assumptions of multiple linear regression: log (small pines, large pine snags, and date of incubation), square root (large pines), and arcsine (% cover of hardwood shrubs). We assessed the linearity of the relationship between dependent and transformed independent variables by examining plots of observed versus predicted values, we assessed normality via normal probability plots, and we verified homoscedasticity with plots of residuals versus predicted values. We also assessed the extent of multicollinearity among our independent variables by first calculating Pearson coefficients for all pair-wise comparisons, using a sequential Bonferroni correction to maintain the table-wide $\alpha = 0.05$ (Rice 1989). Next, we calculated a variance inflation factor (VIF) for each variable (Neter et al. 1996). Neter et al. (1996) suggested that a VIF ≥ 10 indicates the potential for problems with multicollinearity, but we adopted the more conservative rule of Graham (2003) that multicollinearity poses a problem for any VIF ≥ 2 .

We evaluated the degree of support for each model using Akaike's Information Criterion (AIC; Akaike 1973), as corrected for small sample size (AIC_c; Burnham and Anderson 1998), and normalized Akaike weights (w_i). We based our inferences about sources of variation in productivity by considering models included in a 95% confidence set that we defined by summing w_i from largest to smallest until we reached 0.95 (Burnham and Anderson 1998). We only conducted further analyses with the suite of models in the 95% confidence set.

We interpreted the strength of each predictor variable by estimating model-averaged regression coefficients and 95% confidence intervals based on unconditional standard errors (Burnham and Anderson 1998). We calculated model-averaged regression coefficients as the average value of the regression coefficient across all models, weighted by w_i for each model. We considered variables for which the 95% confidence interval around the model-averaged regression

coefficient included zero unimportant in explaining variation in productivity. Using model-averaged estimates of regression coefficients allowed us to incorporate model-selection uncertainty and provided a more robust indication of the effect of each variable on productivity (Anderson et al. 2000). Unless otherwise noted, values presented are means with upper and lower 95% confidence interval limits.

We examined how changes in important predictor variables (i.e., those for which the 95% CI around the model-averaged regression coeff. did not overlap zero) affected productivity by substituting observed values for one variable while holding all other independent variables in the model constant at their observed mean (e.g., Stephens et al. 2005). We repeated this process for each model, and then calculated an overall estimate of productivity using the estimates generated from each model in the 95% confidence set, as weighted by w_i for each model (Burnham and Anderson 2002).

RESULTS

We located and monitored 141 nests during the course of this study, 50 in Long Pine Key and 91 in Raccoon Point. Relatively few nests failed (35%); nests generally produced 4 (22%), 3 (15%), or 2 (20%) fledglings. Nests with 1 (5%) or 5 (3%) fledglings were much less common. We could not determine clutch size for all nests, and thus it is unclear how much of the variation in productivity among successful nests is due to variation in clutch size versus attrition during the nesting cycle. With the exception of a few cases in which we observed the depredation of nests, we did not know the causes of nest failure. On average, successful nests produced 3.0 fledglings (95% CI = 2.7–3.2), and observed productivity from successful nests was similar in the reintroduced Everglades population at Long Pine Key (3.1 fledglings; 95% CI = 2.7–3.4) and the Big Cypress source population at Raccoon Point (2.9 fledglings; 95% CI = 2.6–3.2). Across all nests located during the course of the study, including those that failed, average productivity was 1.9 fledglings (95% CI = 1.7–2.2); total average productivity was similar at both sites (Long Pine Key: 2.0, 95% CI = 1.6–2.5; Raccoon Point: 1.9, 95% CI = 1.5–2.2). The observed estimate of total average productivity closely matched that predicted by the model-averaged estimate from the 95% confidence set of models (see below; model-averaged prediction of productivity = 1.8, 95% CI = 1.4–2.3).

Observed values for habitat features surrounding nest sites varied widely among nests (Table 2). As is the case with many ecological data sets, several of our independent variables were correlated, albeit weakly. The number of small pines was negatively correlated with the number of large pine snags ($r = -0.327$, $P \leq 0.001$), the number of fires within the previous 5 years was negatively correlated with the percent cover by hardwood shrubs ($r = -0.328$, $P = 0.002$) and positively correlated with the number of large pine snags ($r = 0.398$, $P \leq 0.001$), and mean water depth during the breeding and nonbreeding seasons were positively correlated ($r = 0.635$, $P \leq 0.001$). However, VIFs

Table 3. Akaike's Information Criterion values adjusted for small sample size (AIC_c) for candidate models explaining variation in productivity of brown-headed nuthatches in southern Florida, USA, from 1998 to 2003.

Model	K^a	$\Delta AIC_c^{b,c}$	w_i^d
Yr + date + all nest-site vegetation + ← hydrology	10	0	0.36
Global	11	0.7	0.26
Yr + date + hydrology + fire frequency	7	1.1	0.21
Yr + date + all nest-site vegetation	8	3.6	0.06
Yr + date + all nest-site vegetation + fire frequency	9	5.0	0.03
Yr + date + fire frequency	5	5.4	0.03
Yr + date + understory nest-site vegetation	6	5.9	0.02
Yr + date + no. of large pine snags	5	6.1	0.02
Yr + date	4	7.2	0.01
Yr + date + overstory nest-site vegetation	6	8.3	0.01

^a K is the no. of parameters estimated by the model.

^b ΔAIC_c is the difference between a given model and the model with the lowest AIC_c score.

^c The lowest AIC_c score was 127.5.

^d AIC_c wt (w_i) reflects the relative support for each model.

calculated for each independent variable in our data set ranged from 1.5 to 1.7, which indicates a negligible effect of multicollinearity on our estimates of β (Neter et al. 1996, Graham 2003).

There was substantial model-selection uncertainty in our analysis of variation in productivity, and the 95% confidence set of models included 6 of the 10 models in the candidate set (Table 3). The global model adequately fit the data ($F_{10, 131} = 3.16$, $P = 0.002$, adjusted $R^2 = 0.132$), and therefore our reduced models should fit the data as well (Anderson and Burnham 2002). Lacking a single best approximating model, we used model-averaged regression coefficients to examine the relationship between habitat features and productivity. Model-averaged regression coefficients indicated which habitat features had the strongest and most consistent effect on productivity.

The best predictor of fledgling production was the date on which incubation began (model-averaged $\beta = -4.6$, 95% CI = -1.7 – -7.4). Model-averaged estimates of productivity predicted a steep decline as the breeding season progressed (Fig. 1); for example, averaged across all models and holding all other variables at their mean, we predicted a pair that began incubating at the beginning of the nesting season (Julian date 47) would produce, on average, 2.7 fledglings (95% CI = 2.5–2.8), whereas we predicted a pair at the end of the nesting season (Julian date 134) would produce 0.5 fledglings (95% CI = 0.4–0.6). The number of large pine snags surrounding the nest site had a strong positive effect on productivity (model-averaged $\beta = 1.6$, 95% CI = 0.1–3.0), although the effect was much weaker than that estimated for the date of nesting (Fig. 2). Finally, the number of small pines surrounding the nest site also had a weak positive effect on productivity (model-averaged $\beta = 1.3$, 95% CI = 0.4–2.2; Fig. 3). The 95% confidence intervals surrounding model-averaged regression coefficients for all of the other independent variables overlapped zero.

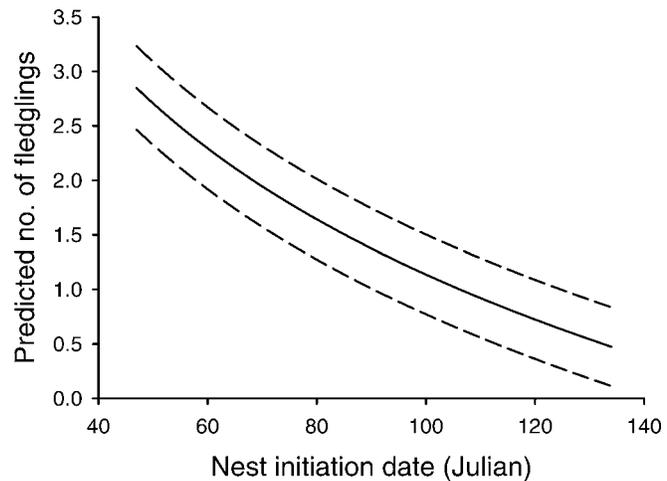


Figure 1. Predicted mean number (solid line; dashed lines are 95% CL) of fledglings produced from brown-headed nuthatch nests in southern Florida, USA (1998–2003), declines as a function of the date on which a nesting attempt began. We calculated the function by substituting different values for nest initiation date into 9 predictive models and generating an Akaike-weighted average across all models; all other independent variables were held at their mean observed value.

DISCUSSION

We found that one feature commonly identified as an important component of habitat for brown-headed nuthatches, the abundance of large pine snags, was positively associated with the number of offspring produced per nesting attempt. We do not know the mechanism underlying the positive association between productivity and the number of large pine snags, but choosing a nest site in an area with a greater number of potential nest sites may diminish the risk of predation (Martin 1988, Martin and

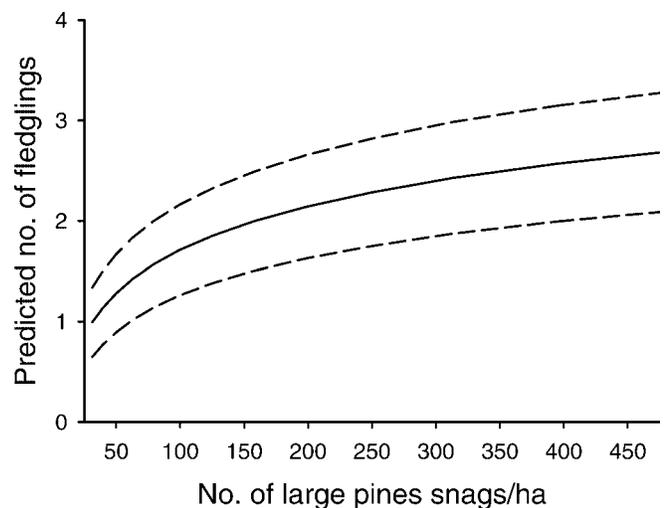


Figure 2. Predicted mean number (solid line; dashed lines are 95% CL) of fledglings produced from brown-headed nuthatch nests in southern Florida, USA (1998–2003), increases as a function of the density of large pine snags. We calculated the function by substituting different values for the number of large pine snags (limited to the range of obs values) into each predictive model containing a term for large pine snags, and then generating an Akaike-weighted average across all models. All other independent variables were held at their mean observed value.

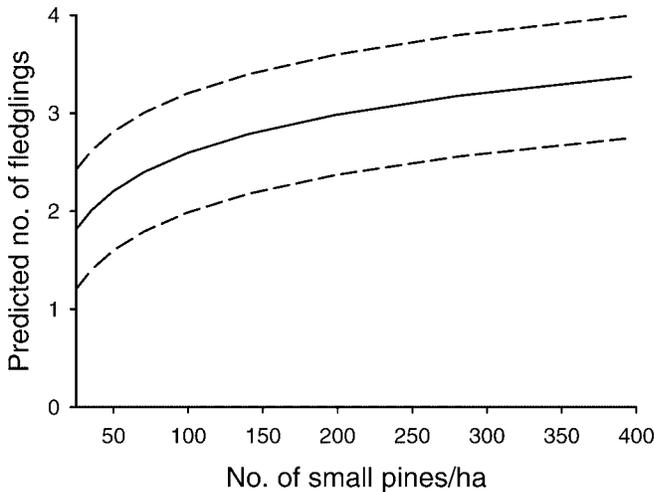


Figure 3. Predicted mean number (solid line; dashed lines are 95% CL) of fledglings produced from brown-headed nuthatch nests in southern Florida, USA (1998–2003), increases as a function of the density of small pine trees. We calculated the function by substituting different values for the number of small pines (limited to the range of obs values) into each predictive model containing a term for small pines, and then generating an Akaike-weighted average across all models. All other independent variables were held at their mean observed value.

Roper 1988, Li and Martin 1991). An increased number of snags also may reduce competition for nest sites with other cavity nesters, reducing the amount of energy that brown-headed nuthatches must expend defending their nest site (brown-headed nuthatches defend nest sites against conspecifics and other cavity-nesting species such as eastern bluebirds [*Sialia sialis*] and red-bellied woodpeckers [*Melanerpes carolinus*]; Slater 1997) and allowing breeding adults to invest more energy in caring for offspring, which may increase productivity.

We also found a positive relationship between the number of young fledged and the number of small live pine trees around the nest site. We lack a good biological explanation for this result because previous studies have found that brown-headed nuthatches prefer mature pine forests (Withgott and Smith 1998) with an open midstory (e.g., Hirth et al. 1991) and that they avoid nesting in areas with numerous small pine trees (Slater 1997). One possible explanation is that high densities of small pines around the nest site increased productivity by reducing the risk of nest predation; for example, dense patches of pines may hinder movement of potential nest predators or may increase concealment of the nest site (Martin 1992). However, we cannot rule out the possibility that the observed correlation was spurious.

The date on which incubation began was by far the most important predictor of productivity. This may be a result of variation in factors beyond the control of land managers, for example, seasonal changes in weather, clutch size, or the abundance of nest predators or competitors for nest sites. However, the seasonal decline in productivity also may have stemmed from seasonal changes in unmeasured components of habitat that influence productivity. Distinguishing among these alternatives is important, given the magnitude of the decline because components of habitat associated with this

decline might be especially useful targets for management, perhaps more so than habitat features traditionally considered important (e.g., abundance of snags or openness of the understory).

None of the variables that we used to quantify fire history and hydrology had a measurable effect on productivity, although models that included an effect of hydrology tended to be well supported relative to models that ignored differences in hydrology. In addition, fire history contributed to variation in the density of large pine snags, which in turn was related to variation in productivity. Snag density was lowest in areas that had not burned in the previous 5 years, and thus the absence of fire was predicted to result in a gradual decline in productivity. We may have also underestimated the importance of hydrology by attempting to document a direct effect on productivity, as other research suggests that the most significant effect of hydrology may be its role as a modulator of fire effects on vegetation structure (Lockwood et al. 2003). For example, fire intensity may be greater in years when water levels are low, which may in turn lead to more pronounced changes in snag abundance (through effects on snag recruitment and mortality) or the condition of understory vegetation.

Fire, which is an important source of mortality in slash pine populations (Menges and Deyrup 2001), may be the most effective tool for increasing snag recruitment while at the same time promoting the understory conditions favored by brown-headed nuthatches (Conner et al. 1983). Typical recommendations for managing brown-headed nuthatch habitat focus on restoring frequent, low-intensity fires, generally at the beginning of the growing season (Dornak et al. 2004), but snag recruitment is lower in frequently burned stands (Menges and Deyrup 2001) and frequent fires may also consume existing snags and shorten snag retention time (e.g., Holden et al. 2006). Fire-return intervals of 1–2 years, as were implemented in the early years of this study, may be useful in restoring long-unburned pine rockland and reducing concomitant increases in fuel loads but, as a long-term management strategy, overly short fire-return intervals may reduce the abundance of snags. Fires in stands that have remained unburned for longer periods (e.g., 6–8 yr) will be significantly hotter and thus produce the greatest number of new snags (Menges and Deyrup 2001, Platt et al. 2002). Longer fire-return intervals likely will also increase snag retention time. At the same time, the density of midstory and understory vegetation increases as fire-return interval increases, and thus a compromise that allows for spatial heterogeneity in fire-return interval may be useful in creating and maintaining high-quality habitat for brown-headed nuthatches. Determining the range of fire-return intervals that will best resolve the conflict between optimal midstory and understory conditions and the number of large snags will require additional information about the role of fire in the population dynamics of snags, including a better understanding of how hydrological conditions and other disturbances (e.g., hurricanes or insect outbreaks) mediate

the effects of fire in the pine rocklands (e.g., Lockwood et al. 2003).

Our conclusions about the relationship between habitat features and productivity of brown-headed nuthatches are applicable throughout most of the remaining pine rockland in southern Florida because of the similar fashion in which these areas are managed. Fire-management targets for the pine rocklands in Everglades National Park and Big Cypress National Preserve generally call for fire-return intervals of 3–6 years, as was applied to our sites at Long Pine Key and Raccoon Point. The other significant tract of pine rockland that remains in southern Florida, in Florida Panther National Wildlife Refuge, had a fire-return interval of approximately 4 years, well within the range of return times observed on our study sites. The only pine rocklands to which our results may not apply are the small and highly fragmented patches that remain within the Miami-Dade County network of parks; however, our own ongoing surveys indicate that brown-headed nuthatches do not occupy these fragments.

MANAGEMENT IMPLICATIONS

Increasing snag density was associated with increased breeding productivity, suggesting that snag creation should be a target of future habitat management for brown-headed nuthatches. We recommend the continued use of fire as a management tool, although our results suggest that more attention should be paid to the trade-off between the length of the fire-return interval and the recruitment of new snags and retention of existing snags. In particular, we recommend increasing the variance in fire-return intervals so that some stands remain unburned for longer than the 3-year to 4-year rotation on which most of the pine rocklands are currently burned. Longer fire-return intervals in some stands will allow for longer snag retention times while increasing the number of snags created following fire. At present, little information exists on which to base precise management recommendations but, as a starting point and until further information is available on the relationship between snag population dynamics and fire-return interval, fire-return interval in some stands should be extended to 5–6 years.

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APPENDIX 3.

PROGRESS REPORT

**AVIAN RESTORATION IN EVERGLADES:
Environmental effects on productivity of Eastern Bluebirds**



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For:
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25 February 2009

INTRODUCTION

The pine rocklands are a globally unique subtropical forest ecosystem considered "critically imperiled" by the U.S. Fish and Wildlife Service (1999). Found only in southern Florida, the Bahamas, and Cuba, pine rocklands have been severely altered by human development, logging, and altered fire and hydrological regimes. In southern Florida, pine rocklands occur in the Florida Keys, Big Cypress Swamp, mostly in Big Cypress National Preserve (BCNP), and along the Atlantic coastal ridge, which terminates in Everglades National Park (ENP). It is along the Atlantic coastal ridge, where the most glaring impacts to pine rocklands have occurred. Once covering over 70,000 ha, the pine rocklands of the Atlantic coastal ridge have been reduced in size by more than 90% (Fig. 1; Doren et al. 1993). The largest remaining tract is the 8,100 ha upland area in Long Pine Key, ENP.

The most visible consequence of habitat loss and degradation in the pine rocklands can be observed in ENP where seven species of breeding bird have been extirpated: Wild Turkey (*Meleagris gallopavo*), Summer Tanager (*Piranga rubra*), Brown-headed Nuthatch (*Sitta pusilla*), Eastern Bluebird (*Sialia sialis*), Red-cockaded Woodpecker (*Picoides borealis*), Southeastern American Kestrel (*Falco sparverius paulus*), and Hairy Woodpecker (*Picoides villosus*). That five of the seven extirpated species are cavity nesters (Nuthatch, Bluebird, Woodpeckers, and Kestrel) suggests that this guild may be especially vulnerable to habitat changes.

Even though the pine rocklands in ENP are no longer threatened by development, the restoration and long-term management of its plant and animal communities present significant challenges. The pine rocklands are fire maintained, requiring a relatively frequent fire interval to deter succession to hardwood hammock, and prescribed fire is an important management tool. South Florida is also the target of a large-scale hydrological restoration program, the Comprehensive Everglades Restoration Plan (CERP), which aims to restore a more natural pattern of hydrology, with respect to quantity and timing, through the Everglades. Yet, little is known about the effects of manipulating fire or hydrological regimes on the terrestrial fauna, particularly birds, and their habitats in pine rocklands. As such, managers lack guidelines for integrating the needs of the pine rockland avifauna into management and restoration plans. Addressing this information gap is critical if restoration and long-term management is to proceed effectively and efficiently in pine rocklands, and studies on the effects of fire and hydrology on

avian communities has been identified as a significant information need in the Department of Interior's Science Plan in support of Everglades Restoration (DOI 2004), the USFWS Multi-species Recovery Plan (USFWS 1999), the Avian Conservation Implementation Plans for ENP and BCNP (Watson 2003), and the Partners in Flight Research and Monitoring Needs Database for Subtropical Florida.

During the period from 1997 to 2001, a reintroduction program was conducted to restore viable populations of Brown-headed Nuthatches and Eastern Bluebirds to ENP (Slater 2001). Both studies were initiated as tests of the progress made in restoring the rare pineland ecosystem (e.g., restoration of natural fire regimes, protection and recovery of the area from logging) represented by Long Pine Key, ENP. Post-release monitoring of nuthatch and bluebird populations indicated the reintroductions were a success, but additional monitoring was recommended to fully evaluate the success of the program (Slater 2005). As part of this effort, we undertook a study to examine how variation in vegetation structure at the nest site, fire history, and hydrological conditions affected productivity of Brown-headed Nuthatches and Eastern Bluebirds in the pine rocklands of southern Florida. The objectives of this study were two-fold: to identify habitat features (vegetation, fire history, hydrology) that were associated with variation in productivity and to provide management recommendation aimed at maintaining the pine rockland avifauna. The results from the Brown-headed Nuthatch analysis have been published (Lloyd and Slater 2007). Below, we include the results from the Eastern Bluebird component, and anticipate that these data will result in a manuscript in the next year.

METHODS

Study area

During the period from 1997 – 2003, we collected data on Eastern Bluebirds from two populations in south Florida. The first was the reintroduced population in Long Pine Key, ENP and the second was the source population for the reintroduction in Raccoon Point, BCNP. Long Pine Key is an 8,100 ha upland area that contains approximately 4,600 ha of pine forest (Snyder et al. 1990, Doren et al. 1993). Within the relatively continuous pine forest, embedded habitats include *Muhlenbergia* prairie, hardwood hammocks, and cypress forest (Olmstead et al. 1983). The pine forest is mostly even-aged, a result of extensive logging in the 1940's, and snags are abundant due to Hurricane Andrew in 1992 (Doren et al. 1993). Raccoon Point is an area of

unlogged, old-growth pine forest within a cypress (*Taxodium* sp.) mosaic. In general, the herbaceous and grass components are well developed, while the shrub layer contains a sparse to moderate amount of saw palmetto and hardwoods.

Demography

We searched for Eastern Bluebird nests beginning in mid-March when they typically begin nest-building. To locate breeding territories, we first searched those areas where territories were detected in the previous year using systematic transects and playback vocalizations. We followed this with transects in areas of formerly unoccupied areas that appeared suitable.

Individuals were followed until their breeding status was determined or a nest attempt was found. Nests were considered active at the onset of incubation (complete clutch) and were monitored every 3-5 days to determine whether the nest fledged or failed. Clutch size was determined using a Tree Top Peeper™ (Sandpiper Technologies, Inc., Manteca, CA). Nests were considered successful if they fledged at least one nestling. The number of juveniles was usually assessed from two counts conducted within three weeks from the date when young fledged. Typically, the first count was performed during the first week after fledging, and the second was conducted within two weeks to determine if juveniles were missed during the initial observation. Regardless of whether a nest was successful or failed, we re-surveyed territories later in the breeding season to account for any re-nesting attempts.

We examined variation in productivity as a function of vegetation structure around the nest site, fire-return interval, and hydrology. We chose to examine the effect of these variables on productivity because previous studies had shown them to be important in nest-site selection, or because our experience in this system suggested they might affect productivity. We did not consider landscape-level factors (e.g., patch size or distance to habitat edge) that may have influenced productivity because we were primarily interested in identifying habitat features that are amenable to management. We analyzed the relationship between productivity and habitat features using an information-theoretic, model selection approach (Burnham and Anderson 1998). We created a candidate set of 13 models, based on linear combinations of variables, that we believed could reasonably explain variation in productivity based on our knowledge of the ecology of Eastern Bluebirds (Table 1).

We evaluated the degree of support for each model using Akaike's Information Criterion (AIC; Akaike 1973), as corrected for small sample size (AICc; Burnham and Anderson 1998), and normalized Akaike weights (w_i). We based our inferences about sources of variation in productivity by considering models included in a 95% confidence set that we defined by summing w_i from largest to smallest until we reached 0.95 (Burnham and Anderson 1998). We only conducted further analyses with the suite of models in the 95% confidence set.

We interpreted the strength of each predictor variable by estimating model-averaged regression coefficients and 95% confidence intervals based on unconditional standard errors (Burnham and Anderson 1998). We calculated model-averaged regression coefficients as the average value of the regression coefficient across all models, weighted by w_i for each model. We considered variables for which the 95% confidence interval around the model-averaged regression coefficient included zero unimportant in explaining variation in productivity. Using model-averaged estimates of regression coefficients allowed us to incorporate model selection uncertainty and provided a more robust indication of the effect of each variable on productivity (Anderson et al. 2000).

We examined how changes in important predictor variables (i.e., those for which the 95% CI around the model-averaged regression coefficient did not overlap zero) affected productivity by substituting observed values for one variable while holding all other independent variables in the model constant at their observed mean (e.g., Stephens et al. 2005). We repeated this process for each model, and then calculated an overall estimate of productivity using the estimates generated from each model in the 95% confidence set, as weighted by w_i for each model (Burnham and Anderson 2002).

RESULTS AND DISCUSSION

We located 82 nests in Long Pine Key and 166 nests in Raccoon Point (Table 2, 3). Productivity ranged from 1.20 (± 0.39) to 4.0 (± 1.87) in Long Pine Key and 0.83 (± 0.28) to 1.52 (± 0.35) in Raccoon Point (Table 2, 3).

Observed habitat (vegetation, fire, hydrology) variables ranged widely at nest sites (Table 4). There was substantial model-selection uncertainty in our analysis, and the 95% confidence set of models included 8 of the 13 models in the candidate set (Table 5). Thus, we used model-

averaged regression coefficients to examine the relationship between habitat features and productivity. The best predictors of bluebird production were snag diameter at breast height, snag height, and distance to a branch (Table 6). These variables likely reflect vulnerability to predation. Productivity was positively associated with snag height and distance to a branch, both of which may deter the ability of a predator to reach a cavity. Curiously, production was negatively associated with snag dbh. We believe this may reflect those bluebird nests in heartwood snags, which are smaller in dbh but have little or no decay on the exterior of the snag, potentially inhibiting predators from accessing the cavity. Figures 1, 2, 3 show predictive values of fledgling production with varying levels of snag height, snag dbh, and distance to branch. In general, these variables alone have a relatively weak effect on productivity.

None of the fire or hydrological variables were directly associated with productivity. However, we know that fire is an important factor in creating and consuming snags and that hydrology is often a modulator of fire intensity (Lockwood et al. 2003). Current fire management primarily focuses on applying frequent, low-intensity fires, generally at the beginning of the growing season. However, snag recruitment is lower in frequently burned stands (Menges and Deyrup 2001) and frequent fires may also consume existing snags and shorten snag retention time (Ecostudies, unpublished data; Holden et al. 2006). Fire-return intervals of 1–2 years, as were implemented in the early years of this study, may be useful in restoring long-unburned pine rockland and reducing concomitant increases in fuel loads but, as a long-term management strategy, overly short fire-return intervals may reduce the abundance of snags. Fires in stands that have remained unburned for longer periods (e.g., 6–8 yr) will be significantly hotter and thus produce the greatest number of new snags (Menges and Deyrup 2001, Platt et al. 2002). Longer fire-return intervals likely will also increase snag retention time. At the same time, as fire-return interval increases the density of midstory and understory vegetation increases, both factors thought to be negatively associated with bluebird nesting (Gowaty and Plissner 1998), even though they were not identified in this study. Thus a compromise that allows for spatial heterogeneity in fire-return interval may be useful in creating and maintaining high-quality habitat for Eastern Bluebirds. Determining the range of fire-return intervals that will best resolve the conflict between optimal midstory and understory conditions and the number of large snags will require additional information about the role of fire in the population dynamics of snags, including a better understanding of how hydrological conditions

and other disturbances (e.g., hurricanes or insect outbreaks) mediate the effects of fire in the pine rocklands (e.g., Lockwood et al. 2003).

Table 1. Summary of candidate models explaining variation in productivity of Eastern bluebirds in southern Florida, U.S.

Model ^a	Model Number
Date	1
DBH + nest height + distance to branch	2
Shrubs (nest site) ^b + bare ground (nest site) ^b	3
Shrubs (nest patch) ^c + bare ground (nest patch) ^c	4
Shrubs (nest site) ^b + bare ground (nest site) ^b + large pines ^d + large pine snags ^d + total trees ^d + total snags ^d	5
DBH + nest height + distance to branch + shrubs (nest site) ^b + bare ground (nest site) ^b	6
Days since fire + water depth	7
DBH + nest height + distance to branch + shrubs (nest site) ^b + bare ground (nest site) ^b + large pines ^d + large pine snags ^d + total trees ^d + total snags ^d	8
Nest height + bare ground (nest site) ^b + large pines ^d + large pine snags ^d	9
Nest height + large pine snags ^d + days since fire + water depth	10
Date + large pine snags ^d	11
Global	12
DBH + nest height + distance to branch + year + site	13

^a The following variables were considered: Date that incubation began (date), diameter at breast height of the nest tree (DBH), nest height, distance from the nest opening to the nearest branch (distance to branch), percent cover of hardwood shrubs at the nest site (shrubs (nest site)), percent bare ground at the nest site (bare ground (nest site)), percent cover of hardwood shrubs within the nest patch (shrubs (nest patch)), percent bare ground within the nest patch (bare ground (nest patch)), number of large pines at the nest site (large pines), number of large pine snags at the nest site (large pine snags), number of trees at the nest site (total trees), number of snags at the nest site (total snags), number of days since the last fire (days since fire), year, and site.

^b Measured in a 5-m radius circle around the nest tree.

^c Mean percent cover measured in 3 5-m radius plots located 40 m north, southeast, and southwest from the nest tree.

^d Measured in a 11.3-m radius circle around the nest tree.

Table 2. Summary of Eastern Bluebird reproductive measures (\pm S.E.) in Long Pine Key, ENP during the period 1998-2003.

	1998	1999	2000	2001	2002	2003
Breeding Territories	1	2	4	16	22	16
No. Nests	1	2	7	27	27	18
Mean incubation date (First attempts)		20 May (± 9)	16 April (± 6)	25 April (± 4)	22 April (± 4)	1 May (± 3)
Clutch size (First attempts)			4.00 (± 0.58)	4.00 (± 0.23)	3.93 (± 0.18)	3.33 (± 0.19)
Productivity (\pm SE; no. young/terr)	2.00	3.00 (± 1.00)	4.00 (± 1.87)	2.38 (± 0.57)	1.73 (± 0.33)	1.20 (± 0.39)

Table 3. Summary of Eastern Bluebird reproductive measures (\pm S.E.) in Raccoon Point, BCNP during the period 1998-2003.

	1998	1999	2000	2001	2002	2003
Breeding Territories	26	25	26	24	18	20
No. Nests	26	28	36	32	18	26
Mean incubation date (First attempts)	1 May (± 3)	1 April (± 2)	3 April (± 6)	26 April (± 4)	21 April (± 4)	16 April (± 4)
Clutch size (First attempts)	3.83 (± 0.17)	4.00	3.87 (± 0.30)	4.29 (± 0.30)	3.80 (± 0.37)	3.80 (± 0.20)
Productivity (\pm SE; no. young/terr)	1.04 (± 0.24)	1.52 (± 0.35)	1.11 (± 0.32)	1.83 (± 0.33)	0.83 (± 0.28)	1.45 (± 0.35)

Table 4. Observed values for habitat variables included in models of productivity for Eastern bluebirds in southern Florida, U.S.

Variable	Mean	Range
Nest tree DBH (cm)	23.4	10.8, 46.0
Nest height (m)	8.7	1.4, 20.2
Distance from nest opening to the nearest branch (m)	2.5	0, 20
Percent cover by hardwood shrubs (nest site) ^a	7.2	0, 40
Percent cover by hardwood shrubs (nest patch) ^b	5.3	0, 31
Percent bare ground (nest site) ^a	44.7	5, 90
Percent bare ground (nest patch) ^b	45.3	7.8, 87.6
Number of large pines ^d	9.5	0, 52
Number of large pine snags ^d	1.5	0, 15
Total number of trees ^d	31.8	0, 215
Total number of snags ^d	6.3	1, 61
Days since last fire	1041.4	6, 2991
Breeding-season water depth (m)	-4.4	-7.0, -0.3

^a Measured in a 5-m radius circle around the nest tree.

^b Mean percent cover measured in 3 5-m radius plots located 40 m north, southeast, and southwest from the nest tree.

^d Measured in a 11.3-m radius circle around the nest tree.

Table 5. Summary of Akaike’s Information Criterion (AIC_c) values for candidate models explaining variation in productivity of Eastern bluebirds in southern Florida, U.S from 1998-2003. K is the number of parameters estimated by the model, ΔAIC_c is the difference between a given model and the model with the lowest AIC_c score^a, and AIC_c weight (w) reflects the relative support for each model.

Model	K	ΔAIC_c	w
DBH + nest height + distance to branch	5	0	0.48
DBH + nest height + distance to branch + shrubs (nest site) ^b + bare ground (nest site) ^b	7	1.2	0.27
DBH + nest height + distance to branch + shrubs (nest site) ^b + bare ground (nest site) ^b + large pines ^d + large pine snags ^d + total trees ^d + total snags ^d	11	3.9	0.07
DBH + nest height + distance to branch + year + site	7	4.1	0.06
Shrubs (nest site) ^b + bare ground (nest site) ^b	4	5.5	0.03
Shrubs (nest patch) ^c + bare ground (nest patch) ^c	4	6.2	0.02
Days since fire + water depth	4	6.3	0.02
Date	3	6.6	0.02
Nest height + large pine snags ^d + days since fire + water depth	6	6.9	0.02
Date + large pine snags ^d	4	7.4	0.01
Nest height + bare ground (nest site) ^b + large pines ^d + large pine snags ^d	6	9.1	0.01
Shrubs (nest site) ^b + bare ground (nest site) ^b + large pines ^d + large pine snags ^d + total trees ^d + total snags ^d	8	11.0	0
Global	18	13.4	0

^a The lowest AIC_c score was 463.3

Table 6. Model-averaged regression coefficients and 95% confidence intervals for habitat variables in models explaining variation in productivity of Eastern bluebirds in southern Florida, U.S from 1998-2003. Variables in bold face have 95% confidence intervals (based on unconditional estimates of variance) around model-averaged estimates of β that do not overlap 1, and therefore were important predictors of fledgling production in this analysis.

Variable ^a	Model-averaged β	Upper 95% Confidence Limit	Lower 95% Confidence Limit
Incubation date	1.00	1.01	0.99
DBH	0.97	0.99	0.93
Nest height	1.07	1.14	1.01
Distance to branch	1.06	1.12	1.00
Hardwood cover, nest site	0.98	1.01	0.95
Hardwood cover, nest patch	0.97	1.01	0.93
Bare ground, nest site	1.00	1.01	0.98
Bare ground, nest patch	0.97	1.04	0.90
Large pines	1.00	1.03	0.97
Total trees	1.00	1.01	0.99
Large pine snags	0.87	1.02	0.74
Total snags	1.01	1.05	0.98
Days since fire	1.00	1.00	1.00
Water depth	1.07	1.20	0.95
Site	0.91	1.62	0.51
Year	0.96	1.13	0.81

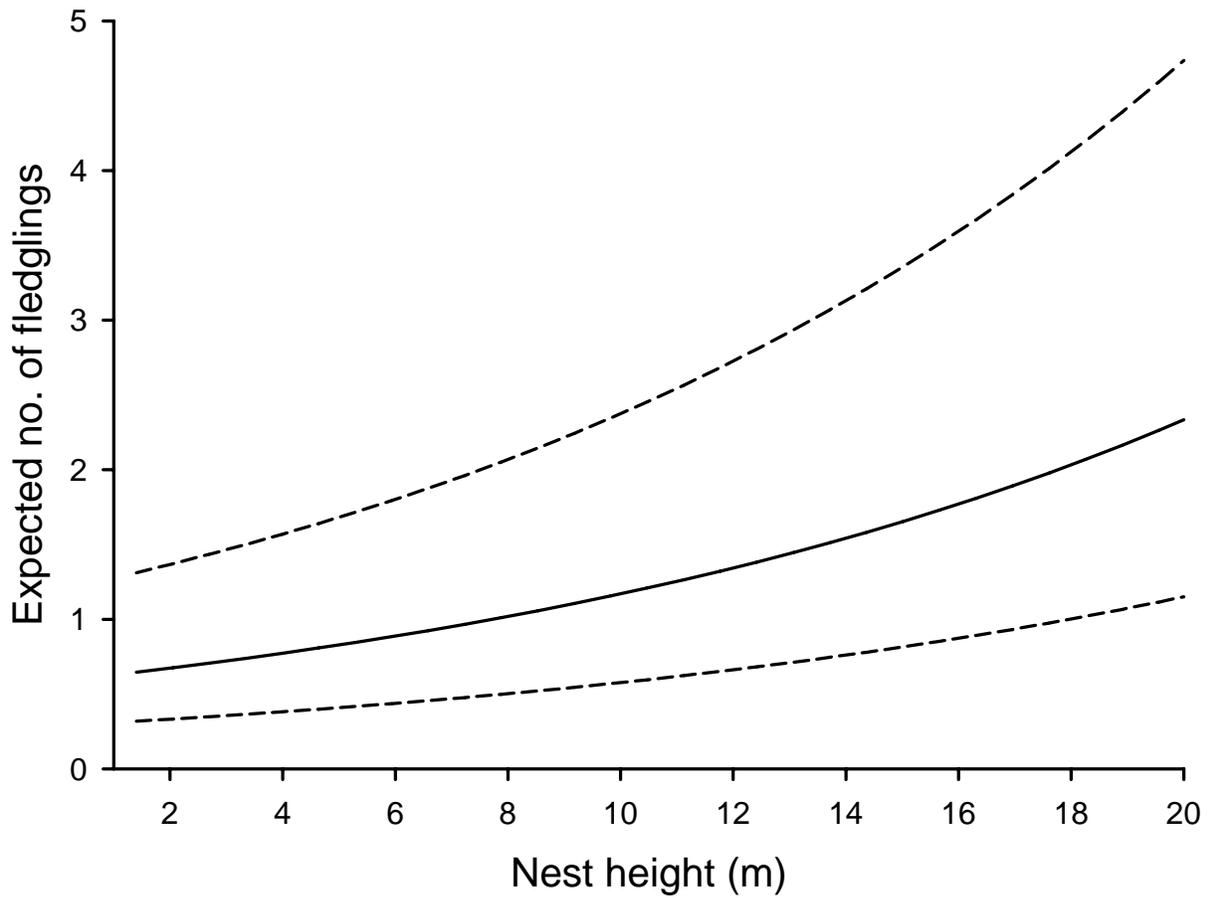


Figure 1. Expected number of fledglings produced by Eastern Bluebirds in south Florida as a function of nest height. Solid line is the weighted mean number of fledglings, averaged across all models that included an effect of nest height. Dashed lines are confidence limits derived from unconditional estimates of variance.

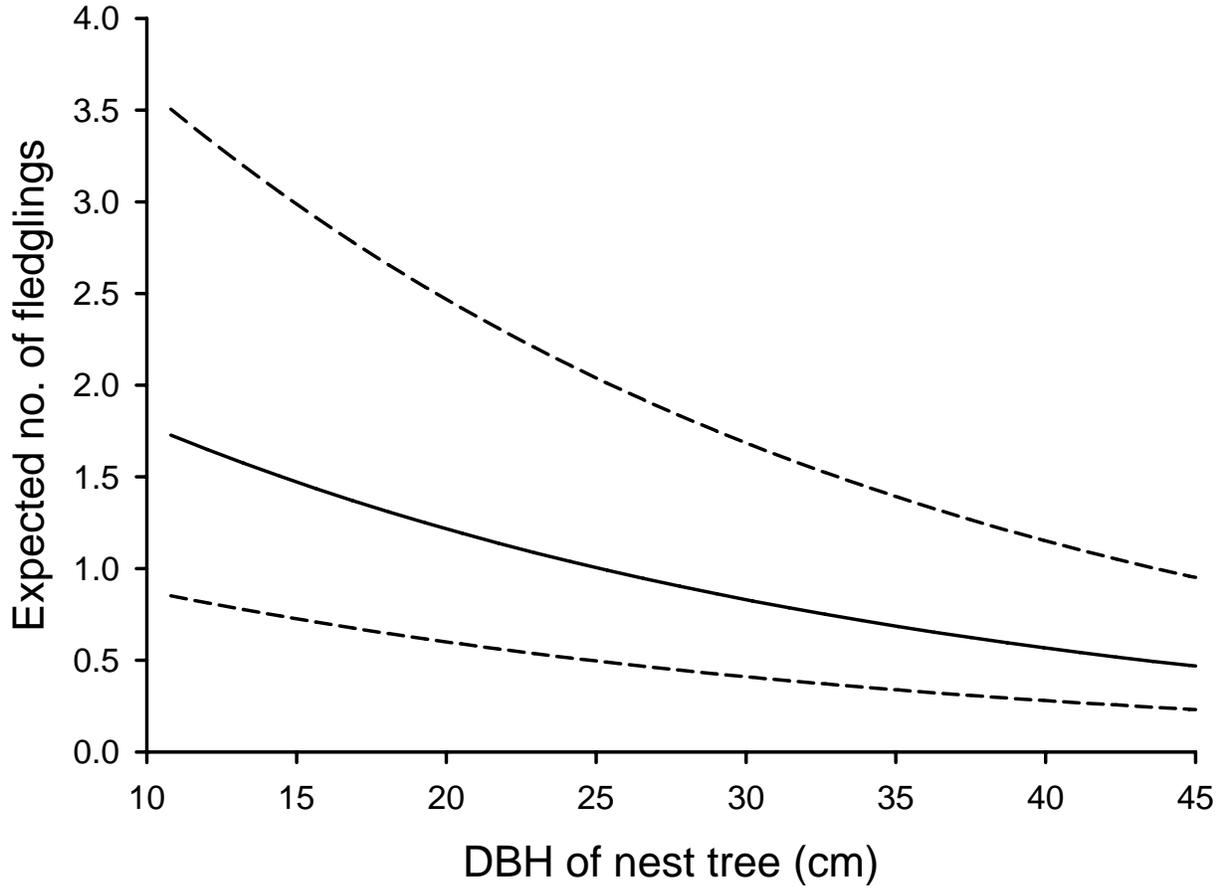


Figure 2. Expected number of fledglings produced by Eastern Bluebirds in south Florida as a function of the diameter at breast height (DBH) of the tree containing the nest. Solid line is the weighted mean number of fledglings, averaged across all models that included an effect of nest height. Dashed lines are confidence limits derived from unconditional estimates of variance.

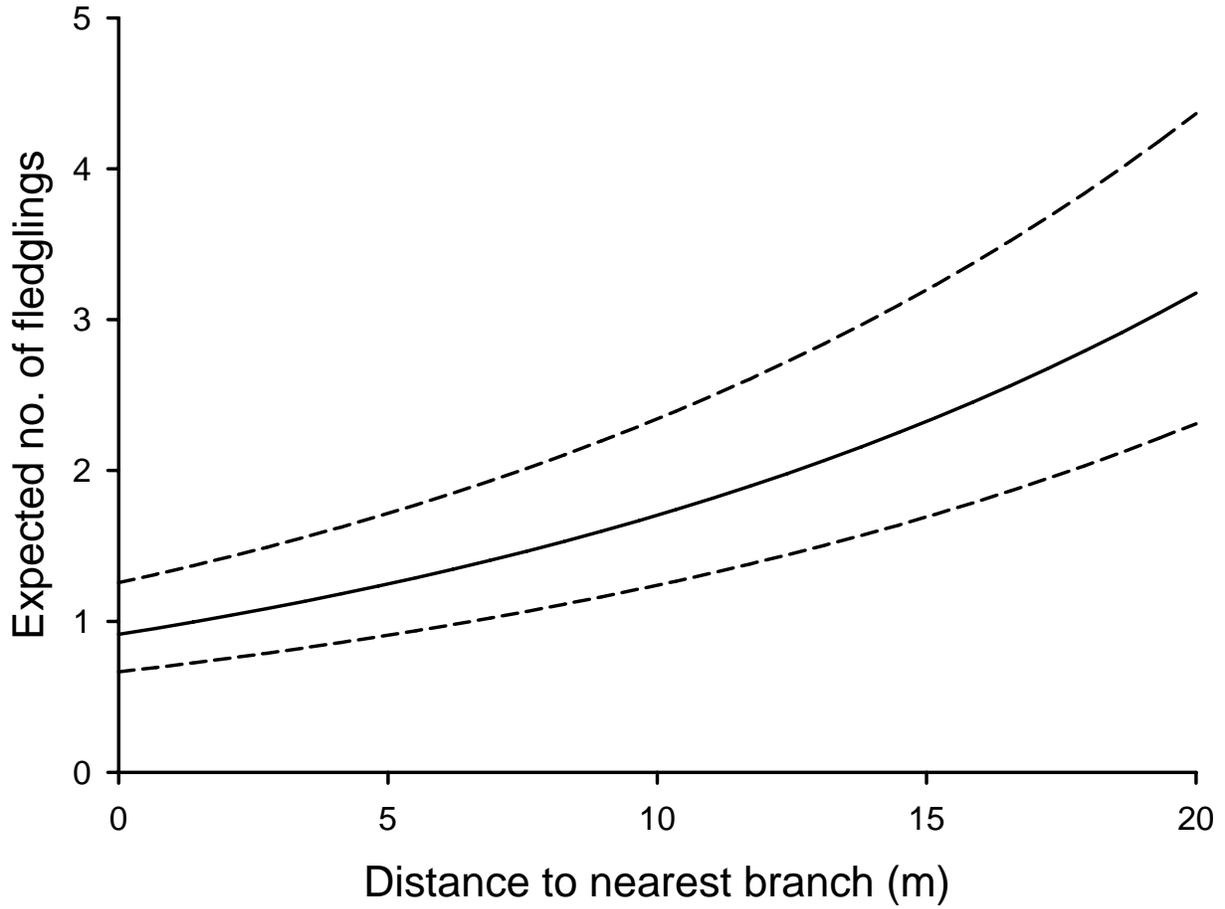


Figure 3. Expected number of fledglings produced by Eastern Bluebirds in south Florida as a function of the distance between the nest opening and the nearest tree branch. Solid line is the weighted mean number of fledglings, averaged across all models that included an effect of nest height. Dashed lines are confidence limits derived from unconditional estimates of variance.

APPENDIX 4.

FINAL REPORT

Effects of fuel treatments and wildfire on the avifauna of the pine rockland ecosystem in southern Florida.

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committed to ecological research and conservation

Project title: Effects of fuel treatments and wildfire on the avifauna of the pine rockland ecosystem in southern Florida.

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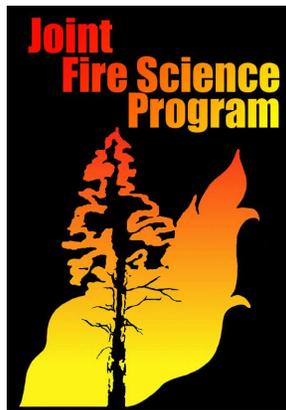
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I. Abstract

The response of plants and animals in the pine forests of southern Florida to variation in fire and hydrological regimes remains inadequately described, hindering the ability of resource managers to manipulate fire and water to achieve desired ecological outcomes. In this study, we took advantage of natural variation in two measures of fire history (the number of days since last fire and the number of times an area had been burned during the previous ten years) and one measure of hydrology (water table elevation) to explore how plants, breeding birds, and wintering birds in slash-pine (*Pinus elliottii* var. *densa*) forests responded to variation in fire and water levels.

At the largest spatial scale, considering samples taken from 441 points located across the range of slash pine in south Florida, variation in the structure and composition of the plant (72% of the explained variation) and both bird assemblages (73% and 80% of the explained variation in the breeding- and wintering-bird assemblages, respectively) is driven primarily by variation in water-table elevation, with the remainder explained by fire history. The relative importance of hydrology was also apparent when examining variation in plant and bird assemblages at a smaller spatial scale. Within study sites, local variation in water-table elevation drove variation in structure and composition of vegetation at 5 of 7 sites, of breeding-bird assemblages at 4 of 7 sites, and of wintering birds at 7 of 7 sites. However, the responses of individual components of vegetation and individual bird species at this smaller scale were not always concordant with patterns observed at the larger spatial scale. Indeed, the effects of variation in water-table elevation that emerged at the smaller scale often ran counter to the effects described at the larger scale. The within-site effects of fire, although generally less important than those associated with variation in water level, were more consistent with patterns described at the larger scale. At both scales of observation, areas burned more recently and frequently tended to contain short, sparse understories and had more standing dead trees, an important component of habitat for several bird species. As expected within a fire-dependent ecosystem, no bird species were associated with fire-suppressed conditions, although most species were able to tolerate fire-return intervals as long as 5 years without any significant effect on abundance.

II. Background and purpose

The south Florida slash pine (*Pinus elliottii* var. *densa*) ecosystem – herein defined to include both the pine rocklands of the Miami Rock Ridge and Tamiami limestone and the more widespread transitional pine flatwoods further to the west – once covered most of the upland area of south Florida. Beginning in the early 20th century, however, large areas of slash pine were cleared for residential, commercial, and agricultural development, and by some estimates the areal extent of this ecosystem is now < 10% of what it once was (Noss and Scott 1997). This ecosystem also supports a large number of endemic and rare plants and animals, and thus management of remaining slash pine is of critical importance to the conservation of biological diversity in southern Florida (USFWS 1999).

Variation in the structure and composition of the slash-pine ecosystem of south Florida is thought to be controlled primarily by fire and, to a lesser extent, by hydrology (Robertson 1953, Alexander 1967, Wade et al. 1980, Gunderson 1994, Platt 1999, Duever 2005). Historic fire-return intervals remain the subject of debate, although consensus estimates suggest that slash-pine forests probably burned, on average, at least twice a decade (Wade et al. 1980, Snyder et al. 1990). As fire-return intervals increase, the density and size of understory shrubs increases, eliminating the open conditions that characterize the ecosystem and reducing the diversity of herbaceous plants, many of which are intolerant of shade (Snyder et al. 1990, Carlson et al. 1993, Platt 1999, Liu et al. 2005). In extreme cases of fire suppression (fire-return intervals >20 years), slash pine is successional to hardwoods (Robertson 1953, Alexander 1967). Because of the difficulty in managing natural fires in this highly fragmented ecosystem, which is embedded within a largely urban matrix, prescribed fire is the primary tool used to restore and maintain the south Florida slash pine ecosystem and its component plants and animals.

Despite its importance as a management tool, questions remain surrounding the ecological effects of prescribed fire in this ecosystem. Most notably, there is relatively little quantitative information describing the response of plants and animals to variation in fire-return interval. This hinders the ability of resource managers to use fire to achieve desired ecological outcomes. In this study, we addressed this information gap by examining the influence of fire history (measured as the time since last fire and the number of times an area had burned during the past 10 years) on present-day variation in vegetation structure and bird abundance at sample points located across the range of south Florida slash pine. In addition, because variation in hydrological conditions can influence the behavior and ecological effects of fire (Lockwood et al. 2003, Slocum et al. 2003), we also examined the relative importance of variation in water-table elevation. Understanding the influence of hydrology on plants and animals in this ecosystem is important as large-scale efforts to restore normative hydrological processes (i.e., the Comprehensive Everglades Restoration Program) begin to impact this system. Finally, we took advantage of experimental fires conducted at a long-term research site on one portion of the study area to examine in detail how variation in the season and frequency of fire affected the demography of standing dead pine trees or snags, which are an important component of habitat for many bird species. The results of this study will help refine fire-management plans for south Florida slash pine forests, identify ecological targets for monitoring and management, and improve our ability

to predict how this ecosystem may change as a result of efforts to restore normative hydrological processes in the greater Everglades.

III. Study description and location

Objective 1) Determine the large-scale patterns of relationship between fire management, hydrology, and abundance and distribution of breeding and wintering landbirds and their habitats.

Bird Survey Stations

During the period from November 2005 – January 2006, we established a network of avian monitoring stations on federal and state lands. Survey stations were established at seven sites dominated by south Florida slash pine. Sites varied in stand age, elevation, fire history, and understory vegetation associations (Fig. 1, Table 1). In Long Pine Key, Everglades National Park (ENP) and Raccoon Point, Big Cypress National Preserve (BCNP) bird survey stations were previously established as part of a long-term monitoring effort of reintroduced cavity-nesting species (G.L. Slater, Ecostudies Institute, unpublished report). One-hundred four and 95 stations were established in Long Pine Key and Raccoon Point, respectively, under the criteria that stations were ≥ 350 m apart and were surrounded by at least 100 m of contiguous pine forest. We used these same criteria to identify survey stations in each of the five remaining sites. To obtain station locations prior to field visits, we first created maps of each site using ArcGIS (ESRI, Redland, CA) and satellite imagery obtained through the South Florida Natural Resource Center's (SFNRC) GIS library and from other agency partners.

Next, we gathered vegetation cover layers for each site to identify all pine-forest patches, and merged annual fire-history layers from each site to determine the time since each patch was burned. Using ArcGIS and our site maps with vegetation and fire history, we overlaid a 50m grid and selected points to serve as survey stations in a stratified random fashion, with strata based on time since an area last burned.

In the field, using the criteria identified above, we determined whether the station could be established at the pre-determined location or whether it needed to be removed from the list. In total, we established 238 points in addition to the 199 established in Long Pine Key and Raccoon Point, for a total of 441 (Table 1).

Bird Surveys

We conducted avian surveys during the non-breeding (15 Dec – 15 Feb) and breeding (15 Apr – 1 Jun) seasons. Each station was visited once per season. Each survey consisted of a seven-minute count, during which observers recorded the radial distance from the sampling station to all birds detected. Surveys were conducted between sunrise and 10:00 as long as weather conditions remained suitable (i.e., light winds and light or no precipitation).

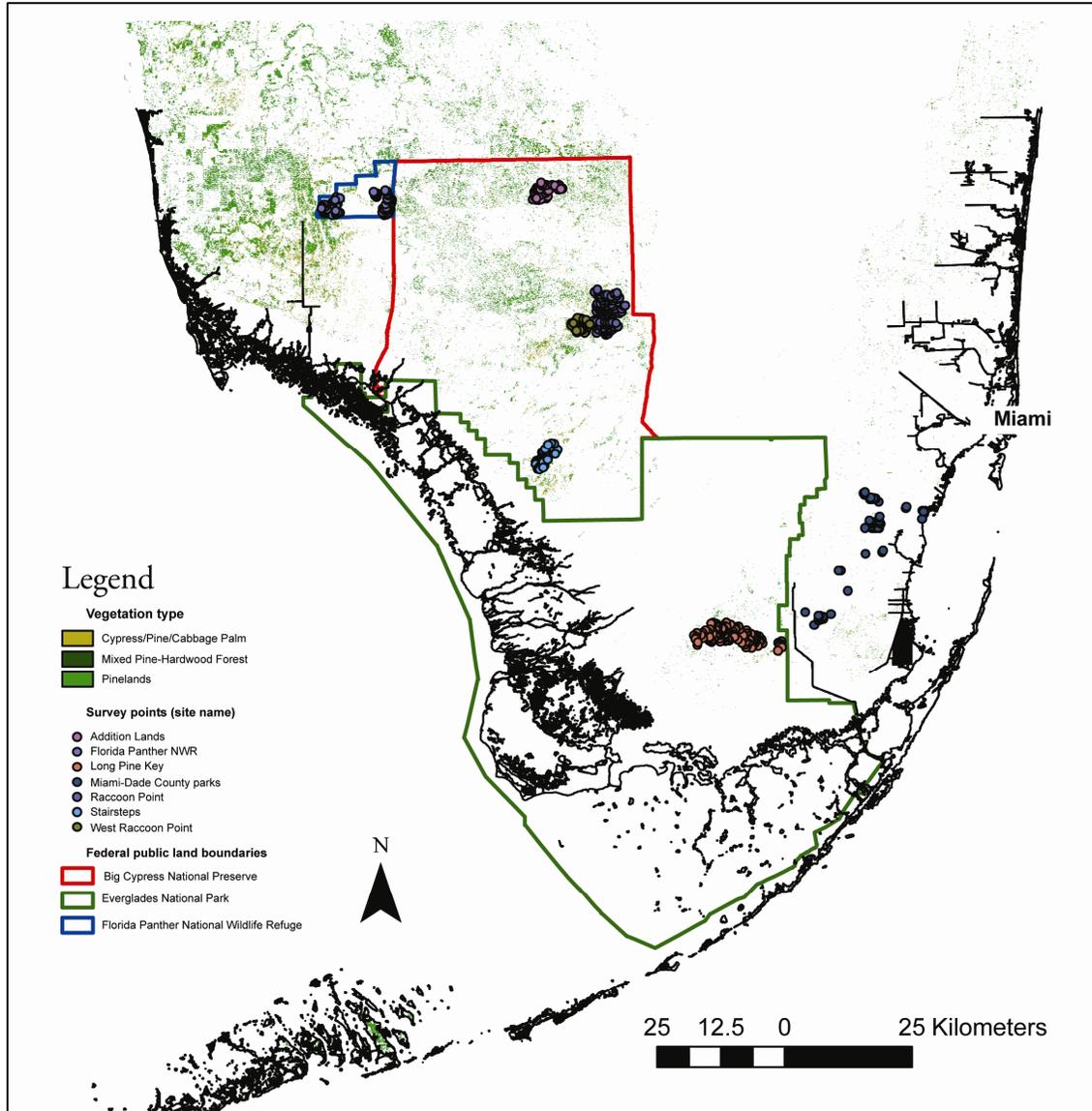


Figure 1. Map of south Florida, USA, showing the 441 points in south Florida slash pine (*Pinus elliottii* var. *densa*) at which bird abundance and vegetation structure were sampled during 2005-2008.

Table 1. Location and description of study areas.**Administrative Unit: Everglades National Park**

Long Pine Key, ENP. 104 survey points. This 8,100 ha upland site lies at the southwest tip of the Miami Rock Ridge and contains ~4,600 ha of pine forest. Within the relatively continuous pine forest, embedded habitats include *Muhlenbergia* prairie, hardwood hammocks, and cypress forest. The pine forest is mostly even-aged, a result of extensive logging in the 1940's. After a decade of fire suppression, the fire management program in the mid-1990s began implementing prescribed burns at the peak of the natural fire ignition season (May- July) to reduce fuel loads and restore natural vegetation communities. Recent fire management applies prescribed fire on a 3-year average fire interval during the natural fire ignition season. Access to this site is through fire roads within the area.

Administrative Unit: Big Cypress National Preserve

Raccoon Point, BCNP. 95 survey points. Raccoon Point contains approximately 9,000 ha of virgin, old-growth pine forest within a cypress mosaic. Raccoon Point has a shallow sandy substrate underlain by limestone bedrock, making it transitional between the pine rocklands further south and the pine flatwoods to the north. For the last decade, this area has been the site of an experimental fire study directed by Dr. Jim Snyder. As is typical for hydric pinelands, the herbaceous and grass components are well developed, while the shrub layer contains a small to moderate amount of hardwoods and saw palmetto. Access to this site is by a gravel road created for oil-extraction activities.

West Raccoon Point, BCNP. 40 survey points. West Raccoon Point lies 5 km west of Raccoon Point separated by large cypress strands. Forest structure and understory composition is similar to Raccoon Point, but the area is slightly lower in elevation than Raccoon Point and has not received prescribed fire as frequently as Raccoon Point. Access to this site is by off-road buggy trails.

Addition Land, BCNP. 42 points. The Addition Land site is approximately 30 km north of Raccoon Point and contains ~30,000 ha of pine forest. This pine forest/dry prairie mosaic floods infrequently because of its higher elevation. The Addition Land was acquired by BCNP in 1996. Its management history included logging, grazing, and frequent burning during the non-growing season, which has resulted in a mature, even-aged forest, a low density of trees and snags, and an overdeveloped shrub layer of palmetto and hardwoods. The area has been subject to several recent wildfires.

Stairsteps, BCNP. 42 points. Stairsteps lies at the northwestern extent of the pine rocklands and contains ~25,000 ha of pine forest. This site was logged in the mid 1900's, resulting in largely second growth stands of pines. Pines are interspersed with wet prairies and small hardwood hammocks. This site has the lowest elevation of all the sites and has a mostly herbaceous understory. Prescribed fire was applied to this area in 2001.

Administrative Unit: Florida Panther National Wildlife Refuge

Florida Panther National Wildlife Refuge. 66 points (only 60 sampled for wintering birds). This site occurs within the Big Cypress Basin, west of Big Cypress National Preserve, and contains ~2,300 ha of pine forest. The refuge is separated into 52 management compartments, originally outlined by old roads and vegetation features. Compartments are burned on a 4-year rotation primarily during the non-growing season (January-February). Cabbage palm dominates the shrub layer, and its abundance may be a consequence of reduced hydrological flow through the refuge. Mechanical thinning was applied to 500 ha in 2005.

Administrative Unit: Miami-Dade County Parks and Recreation Department

Miami-Dade County Parks and Recreation. 52 points. This county agency is responsible for the management of more than 50 environmentally sensitive and natural areas in Miami-Dade County. These sites are relatively small, highly fragmented, and lie in a mostly urban matrix. Less than 2% of Miami-Dade County's original pine rocklands remain. Fire management is difficult due to restrictions related to smoke, citizen acceptance, and availability of personnel to conduct fuel treatments. Wildfire is the primary fuel management technique within these lands, and most sites have high fuel loads as a result of long-term fire suppression.

Vegetation sampling

We sampled vegetation at each survey station annually between 1 December and 1 March, unless vegetation changed due to fire between the non-breeding- and breeding-season bird surveys. In those instances, we resampled vegetation at the survey point.

Vegetation sampling was based on a “spoke and wheel” structure. We determined the number and diameter of living and dead trees (by species) around the survey point in a 11.3-m-radius circular plot and at three additional sampling plots centered 40 m from the survey station at bearings of 0, 120, and 240. At the survey point and at each of the three sampling plots centered 40 m from the survey point we determined percent ground cover (<1.5 m) and shrub cover (between 1.5 and 8 m) for understory habitat components (e.g., herbs and graminoids, hardwood species, and palms) within a 5-m-radius circular plot. Finally, we visually estimated the tallest and average heights of pines, hardwoods, and palms within the shrub layer (<8 meters) along each of the spoke transects.

Estimating fire history

Using annual fire history layers from each organizational unit, we determined the time since each survey point was burned and the total number of times it had been burned in the last 10 years. Time since last fire ranged from 3 days to > 10 years, and the number of burns in the past 10 years ranged from 0 to 9.

Estimating water-table elevation

We downloaded daily stage data for all stations within 25 km of a survey point (data are from the South Florida Water Management District, available online at http://my.sfwmd.gov/dbhydroplsql/show_dbkey_info.main_menu). We then averaged the daily data to estimate the average water stage for each sampling period (defined by the non-breeding [15 Dec – 15 Feb] and breeding [15 Apr – 1 Jun] seasons). Average stage elevations were then kriged using Universal Kriging (spherical model with anisotropy), and the value for each bird-survey station was extracted. We calculated the elevation above sea level (ASL) using the ATLSS elevation model (NGVD M) and the Southwest Florida Feasibility Study elevation model (NAVD FT). If a bird-survey station had data from both models (i.e., models overlapped at that point), then we used the average of the 2 models. We then subtracted the elevation ASL at each survey point from the estimated stage height to calculate water-table elevation.

Based on a random-effects analysis, most (50%) of the variation in our estimates of water-table elevation was due to differences between sampling periods – that is, differences between the wet and dry seasons. Within a season, however, variation was attributable to differences among sites (22%) and among points within sites (27%). Less than 1% of the variation was due to annual variation in water-table elevation. Thus, our measure of water-table elevation served as a measure of average differences among survey points and study sites, essentially allowing us insight into how plants and birds respond along a gradient from relatively dry to relatively wet locations.

Statistical analyses

We began by screening out vegetation variables that we suspected were of little biological significance using two criteria: variables included in the analysis had to have a median value >0 and had to have a non-zero value on >70% of the points. After screening, we retained 22 vegetation variables for inclusion in subsequent analyses (Table 2). We used multivariate multiple regression to examine whether vegetation structure at each survey point exhibited a statistically significant relationship with the independent (i.e.,

Table 2. Twenty-two measures of vegetation structure used in multivariate analyses.

Percent bare ground
Percent ground cover: herbaceous ¹
Percent ground cover: hardwoods
Percent ground cover: palms
Percent ground cover: total
Percent shrub cover: hardwoods ²
Percent shrub cover: palms ³
Percent shrub cover: total
Average height of understory palms
Average height of understory pines
Maximum height of understory palms
Maximum height of understory hardwoods
Maximum total height of the understory
Number of small pine trees (<10.5 cm diameter at breast height [DBH])
Number medium pine trees (10.5 – 18.5 cm DBH)
Number of large pine trees (>18.5 cm DBH)
Total basal area of pine trees
Number of small snags
Number of medium snags
Number of large pine snags
Total number of pine snags
Total basal area of pine snags

¹ Ground cover was defined as the amount of ground surface overlain by vegetation <1.5 m tall

² Shrub cover was defined as the amount of ground surface overlain by vegetation between 1.5 m and 8 m tall

³ Palms lumped to include saw palmetto (*Serona repens*) and cabbage palm (*Sabal palmetto*).

predictor) variables describing fire history and hydrological conditions, while controlling for the random effect of site. However, we expected that these data were unlikely to meet the assumptions of a traditional multiple regression, in particular that the errors were normally distributed (because, e.g., of the abundance of zeros in the data set). Thus we used permutation tests (Anderson 2001, McArdle and Anderson 2001) to calculate the probability of obtaining a squared partial correlation coefficient greater than that observed, given a distribution of possible squared partial correlation coefficients created by permutation. No exact permutation test for a partial regression is possible (Anderson and Robinson 2001), and so we used an approximate test (Freedman and Lane 1983) in which residuals, rather than observations, are exchanged during permutation.

Complicating the interpretation of these tests was the autocorrelation introduced by sampling vegetation at the same survey point in multiple years. Treating measurements from the same point in multiple years would artificially inflate the sample size and potentially increase the risk of committing a Type I error. To address this problem, we restricted permutations such that the residuals associated with a location sampled in multiple years were permuted as a unit, rather than individually (Anderson and ter Braak 2002). As a consequence, the tests were based on 441 permutable units (the number of points in the study), as opposed to the total number of observations taken during the

course of the study (1,516). The multivariate multiple regression was implemented in DISTLM (Anderson 2004). Data were transformed to $x = \ln(x+1)$ prior to analysis to account for the different scales (i.e., integer or percentage) on which dependent (i.e., response) variables were measured. We used Euclidean distances in the analysis, and did not standardize any of the dependent variables prior to analysis (in preliminary analyses we explored standardization and different distance measures, but found no appreciable effect on the outcome of any analysis).

We then used partial canonical analysis of principal coordinates (CAP; Anderson and Willis 2003, Legendre 2008), controlling for the effect of site, to visualize the multivariate patterns suggested by the results of the multivariate multiple regression analysis. We conducted this analysis using the *capscale* function in R (R Development Core Team 2008). As with the multivariate multiple regression, we transformed data to $x = \ln(x+1)$, did not standardize values of dependent variables, and used Euclidean distances. To evaluate the contribution of each measured vegetation variable to the multivariate pattern identified in the CAP, we calculated product-moment correlation coefficients between the CAP score for each point on each axis and the \ln -transformed value of each vegetation variable at that point. We only considered correlation coefficients ≥ 0.20 ; values less than this were assumed to reflect biologically insignificant relationships. We also conducted these ordinations on a site-by-site basis to examine the extent to which patterns observed at the largest spatial scale – the entire study area – were concordant with patterns of variation observed within individual study sites.

We estimated density and abundance of birds using distance sampling as implemented by Program Distance (Thomas *et al.* 2006). Distance sampling is one of several methods that can be used to adjust counts of birds to address imperfect detectability. For these analyses we excluded all individuals detected visually because the detection functions for these encounters were almost certainly different from detection functions generated from encounters with birds that were singing or calling. We could not conduct separate analyses for males and females as most of the detections were not identifiable to sex, either because both sexes share a vocal array (e.g., Northern Cardinal (*Cardinalis cardinalis*) or because the individual was detected via a vocalization used by both sexes.

Following Buckland *et al.* (2001), we modeled detection functions using one of three key functions (uniform, half normal, and hazard rate). In addition, we examined whether changing the shape of each key function via a series expansion term (cosine, simple polynomial, or hermite polynomial) improved the ability of the model to describe the detection function. We evaluated the strength of support for each model using Akaike's Information Criteria, as adjusted for small sample size (AIC_c) and normalized Akaike weights (w_i). We always used the best model for inference, even when AIC_c and w_i indicated model-selection uncertainty, as we found that the estimates of density produced by distance sampling were robust to variation in the specified detection function, and model-averaged values were always identical, or nearly so, to those obtained from the best model. We used the estimated probability of detection from the best model to adjust the counts at each survey point, using only detections from within a 50-m radius of the point. We limited detections to this area so that our estimates of bird density covered approximately the same area as our measures of vegetation structure. We examined relationships between density of birds at each point and fire history and hydrology using

Multivariate multiple regression and CAP, as described for the analysis of variation in vegetation structure. Bird densities were not transformed prior to analysis.

Objective 2) Investigate the role of fire in snag dynamics.

Study area and field methods

We conducted this experiment in Raccoon Point, BCNP, which contains the most extensive unlogged stands of south Florida slash pine in the region (see Table 1). In 1993, we established 12 1-ha plots in the pinelands of Raccoon Point and in each plot we tagged and mapped all trees with diameter at breast height (dbh) > 5.0 cm and assigned each to one of four states: live, low-to-moderately decayed snag (decay state 1; ranged from newly created snags with tight bark and intact branches to moderately decayed snags with some or no bark remaining, branches mostly absent, and sapwood intact to sloughing), heavily decayed snag (decay state 2; only heartwood remaining), and down (any tree or snag completely consumed by fire, completely uprooted or broken at a height of < 1.5 m, or decayed to a height of < 1.5 m or DBH < 5.0 cm). We chose to categorize the continuous process of decay in this fashion because these categories generally reflect the suitability of a snag for Brown-headed Nuthatches and Eastern Bluebirds (G.L. Slater and J. D. Lloyd, unpublished data), both of which are species of management concern. Brown-headed Nuthatches are weak excavators, capable of creating cavities in soft wood, but Eastern Bluebirds are secondary cavity nesters and rely exclusively on cavities excavated by other species. Both species rarely use cavities in live trees or in snags in decay state 2, presumably due to the hardness of the heartwood of south Florida slash pine. Indeed, 91% (n = 409) of nests of both species that we monitored from 1998-2005 were found in snags in decay state 1 (Ecological Institute, unpubl. data). Given that many of the cavities used by these two species were likely excavated by other species (e.g., Red-bellied Woodpeckers (*Melanerpes carolinus*) or Downy Woodpeckers (*Picoides pubescens*), we assume that other cavity-nesting birds show similar preferences.

The initial state of each tree or snag was recorded between July 1993 and October 1994. The final state of each tree was recorded 15 years later in January 2008. All plots were subject to baseline prescribed fires conducted during January-February 1990 and March-April 1994. Thereafter, between 1996 and 2006, each plot was treated with experimental burns on a short interval (target interval was 3 years between successive fires) or a long interval (target interval was 6 years between successive fires), with fires conducted during the dry (November – April) or wet season (May – October). Plots were randomly assigned to one of the four treatment combinations (dry season, long interval; dry season, short interval; wet season, short interval; and wet season, long interval). Each treatment combination was applied to three study plots. We used bark char height (height of the blackened trunk) as an index of fire intensity (e.g., Waldrop and Van Lear 1984). Bark char height was measured one month after the first experimental fire in all plots, and one month after the second experimental fire in four of the plots that received the short-interval treatment.

Statistical analyses

We examined treatment effects on transition probabilities – as measured by the state of each marked tree or snag at the first (1993/1994) and final (2008) sampling periods - by fitting baseline category, multinomial logit models using the *multinom* function in R (Venables and Ripley 2002). We evaluated a set of five candidate models in this analysis (Table 3).

Table 3. Candidate set of multinomial logit models used to examine the effects of season of burn (dry or wet season) and fire-return interval (short or long) on the dynamics of snags in a south Florida slash pine forest in Big Cypress National Preserve, Florida.

Model name	Model structure ^a
Null model	Initial state + observation days + DBH
Season	Initial state + observation days + DBH + season
Interval	Initial state + observation days + DBH + interval
Season + interval	Initial state + observation days + DBH + season + interval
Season*interval	Initial state + observation days + DBH + season + interval + season*interval

The response variable for the analysis was the state of the tree or snag at the final observation. Every model in the candidate set included a term for the initial state of the tree or snag because the initial state determined which states were possible at the final observation (e.g., a snag in decay state 2 at the initial observation could end the experiment in only one of two possible states (decay state 2 or down), whereas a tree that was alive at the initial observation had four possible endpoints: still alive, decay state 1, decay state 2, or down). Every model also included a term for the number of days that a tree was under observation, which accounted for any differences in transition probability arising from variation in the date of the initial observation, and a term for DBH, which can have an important effect on the persistence of snags (Morrison and Raphael 1993). We examined the effects of our treatments by including dummy variables for season of burn and fire-return interval in some of the models. We evaluated the degree of support for each model using Akaike's Information Criteria (AIC) and Akaike weights (w), which reflect the relative likelihood that a given model is the best model in the set of candidate models. Akaike weights were calculated as

$$w_i = \frac{\exp\left(-\frac{1}{2}\Delta_i\right)}{\sum_{r=1}^R \exp\left(-\frac{1}{2}\Delta_r\right)}$$

where Δ_i is the difference in the AIC value between model i and the model with the minimum AIC.

We used the coefficients from the best-fitting model to evaluate the strength and direction of treatment effects on transition probabilities. We also calculated the percentage change in the odds of a transition occurring as a function of a change in the value of our predictor variables by subtracting 1 from the odds ratio (calculated as the exponent of the coefficient of the predictor variable) and multiplying the value by 100. We did not interpret odds ratios for predictor variables where 95% confidence limits overlapped one. However, interpretation of the coefficients in a multinomial logit model is non-intuitive, as they represent the relative log-odds of membership in a particular decay state as opposed to the baseline category, which in this case was a live tree. As such, much of our interpretation is based on the fitted values for transition probabilities as generated from the best-supported model (Fox and Anderson 2006), which we believe provide a more readily interpretable presentation of the results. When comparing transition probabilities among different treatment groups, we considered non-overlapping 95% confidence intervals as evidence of statistically significant differences. We also calculated average lifespan of snags as

$$l = \frac{-1}{\ln(S)},$$

where (S) was the annualized probability that a snag in either decay state remained in that decay state, as calculated from the estimated probability from the best model that a snag that began the study in either decay state remained in that same state until the end of the study.

To further ease in the interpretation of our results, we also used the fitted transition probabilities from the best-supported model to project changes in the population of snags at Raccoon Point under each of the treatment combinations. The purpose of these simulations was not to predict future changes in the population of snags, but rather to translate the matrix of predicted transition probabilities into a more intuitive quantity (density of trees and snags). We used 50 15-year time steps, which appeared sufficient to illustrate the consequences of variation in transition probabilities among treatments (i.e., an approximate asymptote was reached by 50 time steps). Starting vectors were equal to the average density of individuals in each state at the final observation. Changes in the density of snags and trees between time steps were based on transition probabilities randomly drawn from a beta distribution centered on the estimated mean for each transition and with the estimated variance for each transition (means and variances were estimated from the best-supported model in the candidate set). We chose to use the beta distribution as it is considered the most reasonable choice for simulating matrix elements, such as the transition probabilities reported here, that are limited to values between 0 and 1 (Morris and Doak 2002). For example, the number of stage 1 snags at time $t+1$ was projected as

$$N_{1(t+1)} = N_{0(t)} (P_{0,1}) + N_{1(t)} (P_{1,1})$$

where

$N_{0(t)}$ = number of live trees at time t ,

$P_{0,1}$ = probability that a live tree enters decay state 1 between time t and $t+1$,

$N_{1(t)}$ = number of snags in decay state 1 at time t , and

$P_{1,1}$ = probability that a tree in decay state 1 remains in decay state 1 from time t to $t+1$.

The fifty-step projection for each treatment combination was repeated 1,000 times. We estimated the mean projected number of snags in each decay state at each time step from the 1,000 simulated projections, and used the 2.5 and 97.5 percentiles from the simulations to define the 95% confidence limits around each mean.

IV. Key findings

Objective 1) Determine the large-scale patterns of relationship between fire management, hydrology, and abundance and distribution of breeding and wintering landbirds and their habitats.

Key finding 1) Variation in hydrology was the primary driver of large-scale variation in vegetation structure and the distribution and abundance of breeding and wintering birds across the range of south Florida slash pine. Fire history was of secondary importance.

Water-table elevation, the number of days since last fire, and the number of times a sample plot had burned explained a significant ($P < 0.001$) amount of variation in both vegetation structure and abundance of breeding birds. Only water-table elevation explained a significant amount of variation in the density of birds during the winter; neither of the axes related to the two fire variables explained a significant amount of variation ($P > 0.3$). Despite the statistically significant relationship between our predictor and response variables, the vast majority of variation in the structure of vegetation and the distribution and abundance of birds was not accounted for by either hydrology or fire history (for plants, 6.8% of the total variation was explained by the predictor variables; for breeding birds, 4.8%; and for wintering birds, only 0.5%). Of the variation explained by the predictor variables, water-table elevation accounted for the majority: 72% of the explained variation in vegetation structure, 73% of the explained variation in breeding-bird densities, and 77% of the explained variation in densities of wintering birds.

Across all sample points, hardwood ground and shrub cover, the maximum height of understory hardwoods, the number of small and medium pines and pine snags, the total number of snags, and the height of understory pines all declined as water-table elevation increased (Fig. 2). The average height of understory palms, the maximum height of understory palms, palm shrub cover, the number of large pine trees, and total basal area of pine trees all increased as sites grew wetter (Fig. 2). Points burned more recently and frequently were characterized by less grass cover, less ground and shrub cover by hardwoods, shorter understory hardwoods, less total shrub cover, shorter understory pines, fewer small and medium pines, and more pine snags of all size classes (Fig. 2).

Breeding birds associated with wetter sites included Blue-gray Gnatcatcher (*Polioptila caerulea*; BGGN), Brown-headed Nuthatch (*Sitta pusilla*; BHNU), Carolina Wren (*Thryothorus ludovicianus*; CARW), Common Yellowthroat (*Geothlypis trichas*; COYE), Downy Woodpecker (DOWO), Eastern Bluebird (EABL; *Sialia sialis*; EABL), Great-crested Flycatcher (*Myiarchus crinitus*; GCFL), Pine Warbler (*Dendroica pinus*; PIWA), Red-bellied Woodpecker (RBWO); Tufted Titmouse (*Baeolophus bicolor*; TUTI), and White-eyed Vireo (*Vireo griseus*; WEVI) (Fig. 3). Only two species were more common on dry sites: Blue Jay (*Cyanocitta cristata*; BLJA) and Northern Mockingbird (*Mimus polyglottos*; NOMO) (Fig. 3). Downy Woodpecker, Great-crested Flycatcher, Northern Cardinal, Pine Warbler and Red-bellied Woodpecker were also associated with more recently or frequently burned points; no species was associated with fire-suppressed conditions (Fig. 3).

Wintering birds showed weaker associations in general with all of the independent variables. Three species – American Robin (*Turdus migratorius*; AMRO), Pine Warbler, and Yellow-rumped Warbler (*Dendroica coronata*; YRWA) – were associated with wetter sites; none were associated with lower water-table elevations (Fig. 4). Densities of wintering birds were not associated with any particular fire history (Fig. 4).

The relatively poor performance of the models summarized in Figures 2, 3, and 4 suggests the importance of biotic interactions, other past and ongoing sources of disturbance, and biogeographical considerations in understanding the present appearance of the pine ecosystems of southern Florida, at least when viewed at large spatial scales.

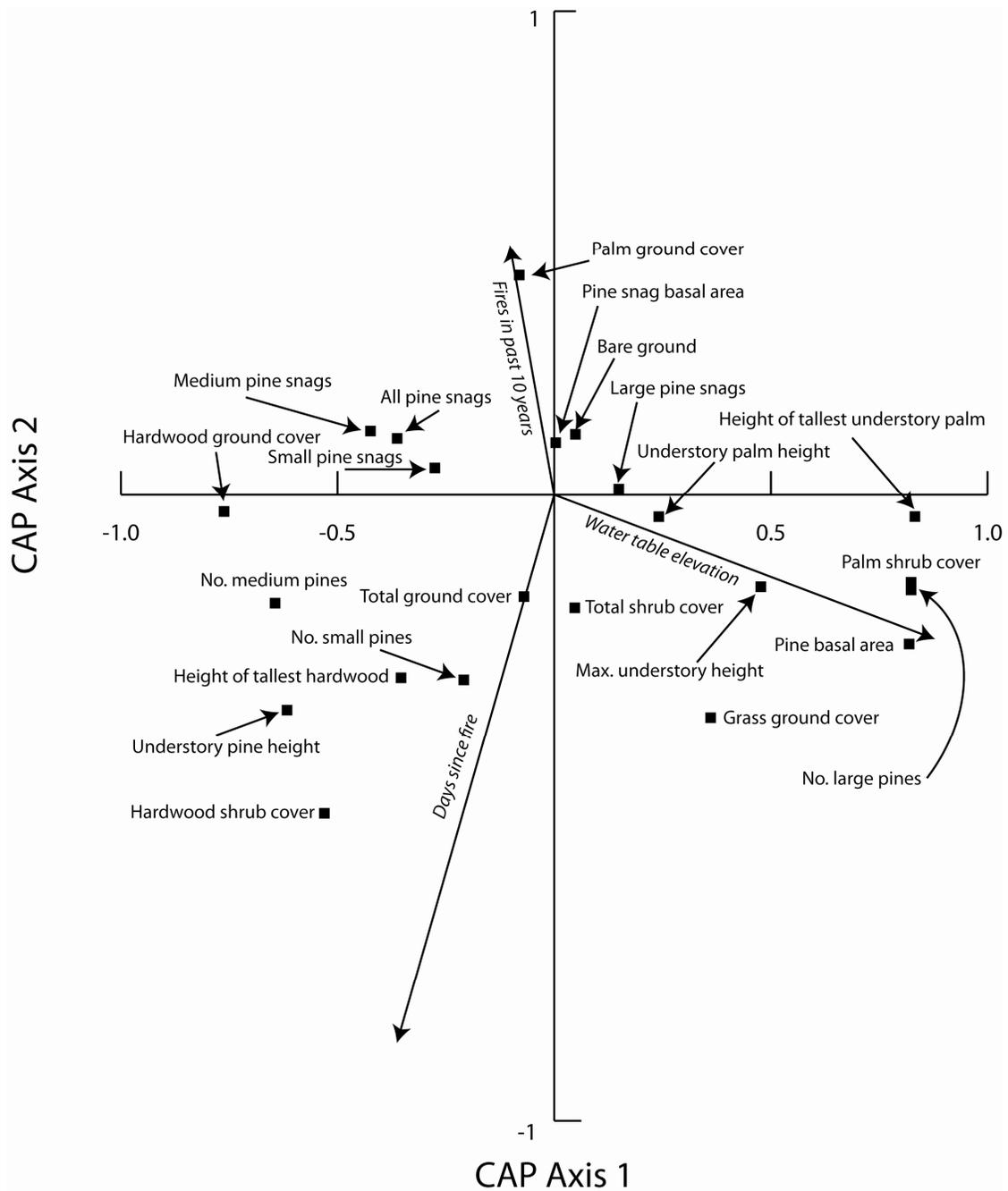


Figure 2. Results of a partial canonical analysis of principal coordinates (CAP) conducted on vegetation variables measured in slash-pine (*Pinus elliottii* var. *densa*) forests across southern Florida, 2005-2008. CAP axis 1 is positively correlated with water-table elevation ($r = 0.50$), and CAP axis 2 is negatively correlated with the time since last fire ($r = -0.34$) and positively correlated with the number of times a point burned ($r = 0.20$). The position of each vegetation variable indicates the strength and sign of its association with the independent variables.

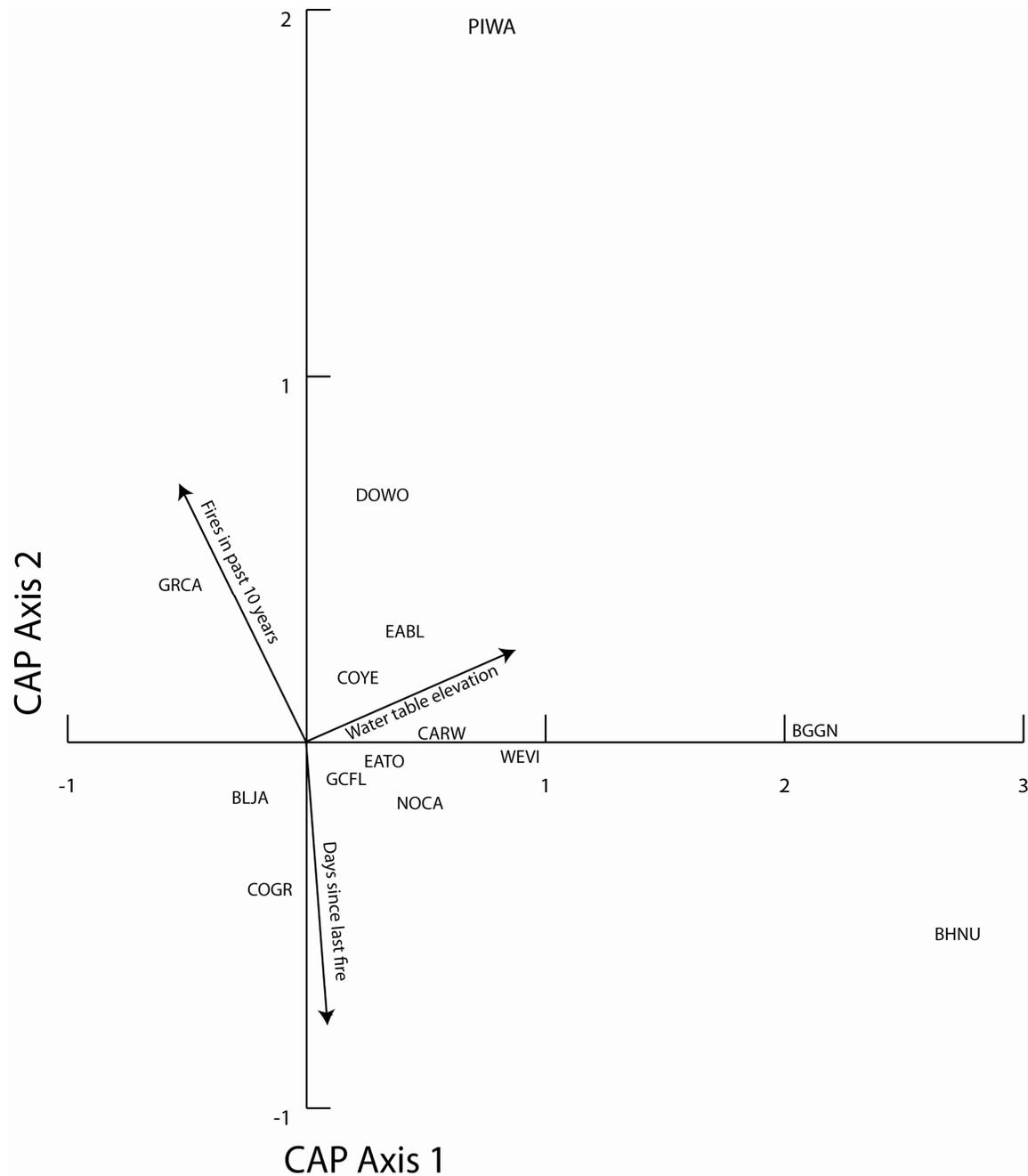


Figure 3. Results of a partial canonical analysis of principal coordinates (CAP) conducted on breeding-bird densities estimated in slash-pine (*Pinus elliottii* var. *densa*) forests across southern Florida, 2005-2008. CAP axis 1 is positively correlated with water-table elevation ($r = 0.40$), and CAP axis 2 is negatively correlated the time since last fire ($r = -0.15$) and positively correlated with the number of times a point burned ($r = 0.22$). The position of each bird species code indicates the strength and sign of its association with the independent variables.

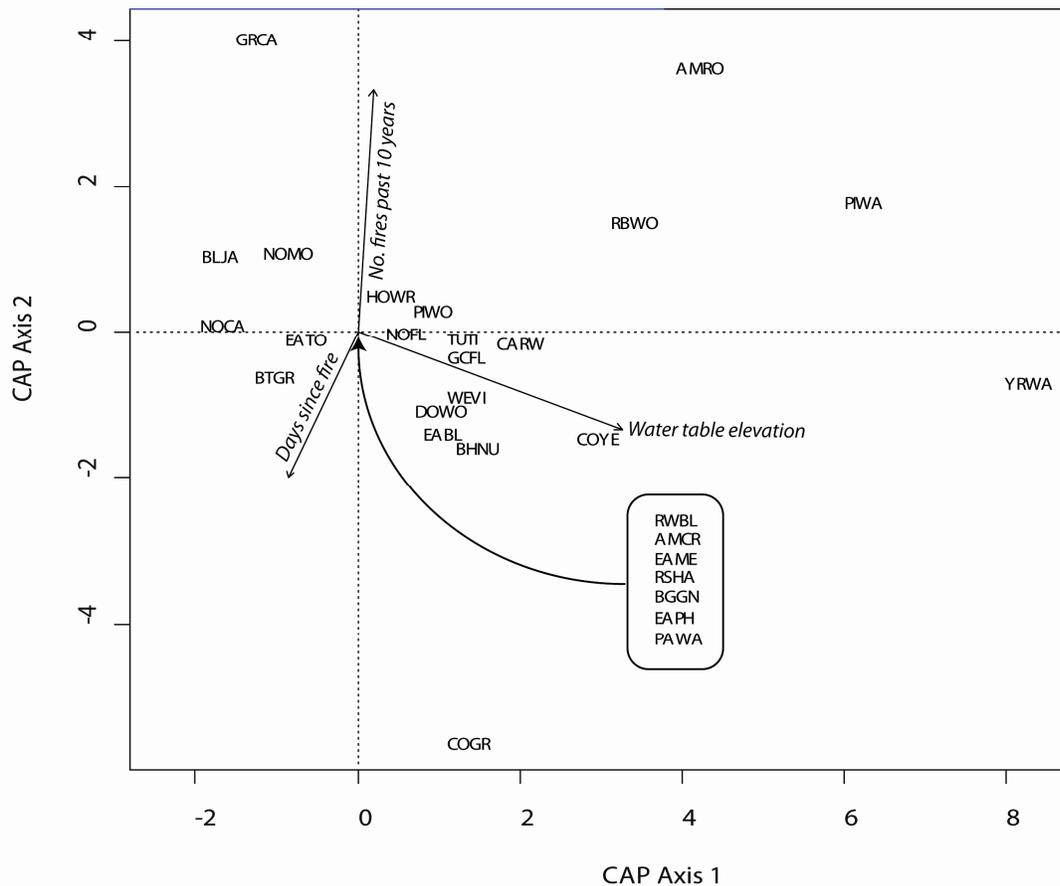


Figure 4. Results of a partial canonical analysis of principal coordinates (CAP) conducted on nonbreeding-bird densities estimated in slash-pine (*Pinus elliottii* var. *densa*) forests across southern Florida, 2005-2008. CAP axis 1 is positively correlated with water-table elevation ($r = 0.11$). CAP axis 2 was non-significantly related to time since last fire ($r = -0.05$) and the number of times a point burned ($r = 0.09$). The position of each bird species code indicates the strength and sign of its association with the independent variables. Species clustered near the origin show no relation to any of the independent variables.

Key finding 2) Variation in hydrology was the primary driver of within-site variation in vegetation structure and the distribution and abundance of breeding and wintering birds. Fire history was of secondary importance.

Variation in water-table elevation was more strongly associated with among-point, within-site variation in vegetation structure at 5 of 7 study sites than either of the fire variables. The number of days since last fire was the strongest predictor of vegetation structure at the other two study sites (Addition Land and Miami-Dade County parks). Variation in breeding-bird density was most strongly related to variation in water-table elevation at 4 of 7 sites, with breeding-bird density responding to the number of days since fire at a single site (Addition Land) and to the number of fires at two sites (Miami-Dade

County parks and Raccoon Point). Water-table elevation was always the strongest predictor of bird densities during the non-breeding season.

Key finding 3) The effects of hydrology on plants were strongly scale dependent, and effects described at large spatial scales were not always concordant with effects described at smaller scales. The effects of hydrology on birds tended to be more consistent among scales.

Some of the large-scale associations between water-table elevation and birds and plants reflect differences among study sites, and these relationships were often not apparent when examined within study sites. For example, at the largest scale, hardwood ground and shrub cover was negatively associated with water-table elevation; that is, hardwood shrubs were denser and more abundant at wetter sites. This is concordant with existing understanding of the role of hydrology in shaping the structure and composition of south Florida slash pine forests: sites with shallower water tables have fewer hardwood shrubs and grassier understories than drier sites, presumably because an elevated water table reduces the ability of hardwoods to establish and grow (e.g., Duever 2005). However, within each of the seven sites we studied, the opposite was true: hardwood ground and shrub cover was greater at points with higher water tables. This may reflect the inability of prescribed fires to carry across wetter portions of a site, resulting in longer fire-return intervals for these points and decreased mortality of hardwoods.

The relationship between water-table elevation and breeding bird densities tended to be more consistent across spatial scales. For example, Pine Warbler, Red-bellied Woodpecker, and Common Yellowthroat were all positively associated with water-table elevation at the largest spatial scale and were also positively associated with water-table elevation within each of the study sites.

Relationships between bird densities during the winter and water-table elevation differed between scales, which is not unexpected given the weak association between non-breeding birds and any of the measured environmental variables. For example, Yellow-rumped Warbler, which was strongly and positively associated with water-table elevation at the largest scale, was more abundant at dry points at 4 out of 6 study sites (and showed no relationship with water-table elevation at the remaining site).

Key finding 4) In contrast to hydrology, the effects of fire history on plants were consistent between scales, but the effects of fire history on bird densities were variable between scales.

Recent and frequent fires produced a similar vegetation structure regardless of the scale of observation: more bare ground, less shrub cover, shorter understories, and fewer overstory pine trees. The effects of fire on breeding birds were less consistent. Pine Warblers were strongly and positively associated with fire across all study sites, but within 4 of the study sites their abundance increased as the number of burns decreased and the time since last fire increased. At two sites they showed the expected positive relationship with fire, and at the remaining site abundance of Pine Warblers was not associated with variation in fire history.

Key finding 5) Breeding birds tolerated fire-return intervals of as long 5 years without significant changes in density. For some species, density declined when fire-return intervals exceeded 5 years.

For the five species (Northern Cardinal, Pine Warbler, Red-bellied Woodpecker, Downy Woodpecker, Great-crested Flycatcher) that showed a positive association with fire across all study sites, all showed a similar pattern of abundance in relation to the time since last fire, wherein abundance remained constant for up to 5 years after fire, then declined sharply from 5-7 years post-fire, and finally leveled out at > 7 years post-fire (e.g., Figs. 5 and 6). Three of these species are cavity nesters – Downy Woodpecker, Red-bellied Woodpecker, and Great-crested Flycatcher – and their increased abundance at shorter fire-return intervals may be due to the increase in number of snags associated with recent fires.

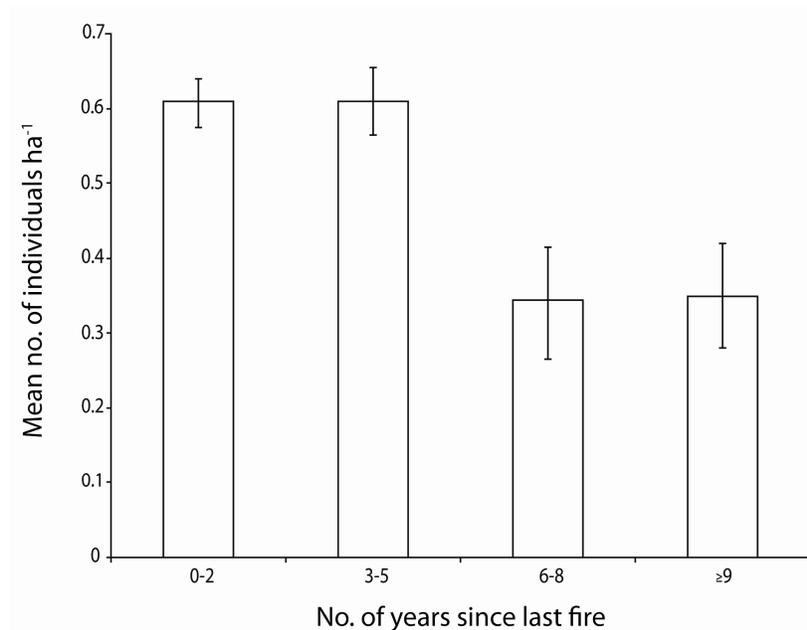


Figure 5. Abundance of Pine Warblers (*Dendroica pinus*) breeding in slash-pine (*Pinus elliotii* var. *densa*) forests of south Florida during 2005-2008 as a function of time since last fire.

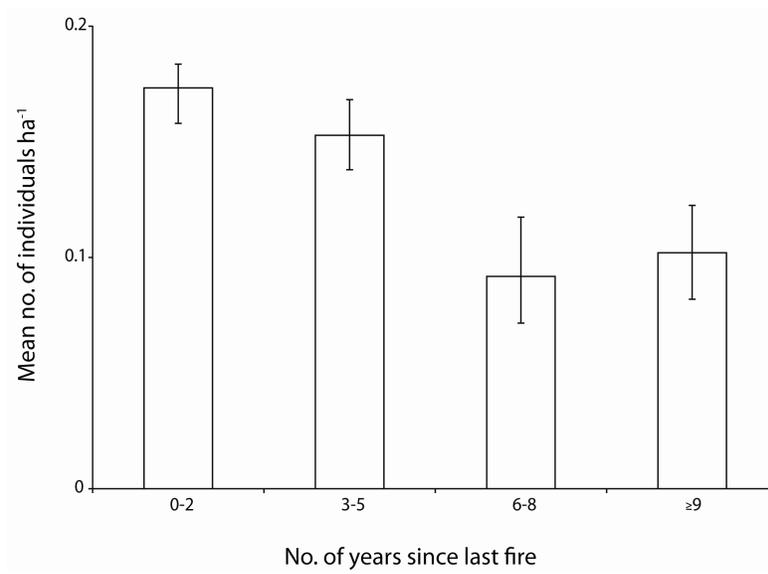


Figure 6. Abundance of Great-crested Flycatchers (*Myiarchus crinitus*) breeding in slash-pine (*Pinus elliottii* var. *densa*) forests of south Florida during 2005-2008 as a function of time since last fire.

Objective 2) Investigate the role of fire in snag dynamics.

Key finding 1) Dry-season burns were more intense than wet-season burns and resulted in a higher probability of snag creation than wet-season burns.

Fire intensity, as measured by the average height of bark char one-month postfire, was much greater for dry-season burns (4.4 m, 95% CI = 2.9 – 6.0) than for wet-season burns (2.7, 95% CI = 1.7 – 3.7). Increased fire intensity during dry-season burns may have led to increased mortality of pine trees, resulting in an increase in the probability of snag creation. The estimated probability that a live tree became a snag over the course of the study was significantly greater for plots burned during the dry season (Fig. 7).

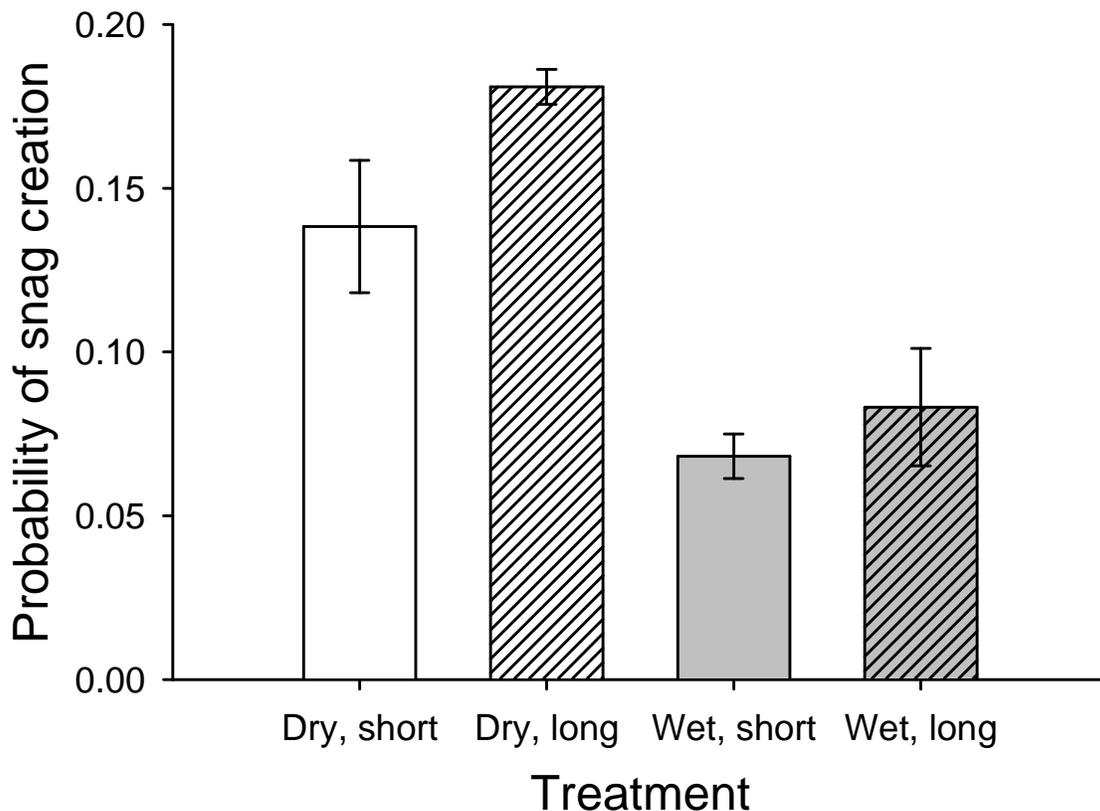


Figure 7. Predicted probability (\pm 95% confidence interval) that a live south Florida slash pine (*Pinus elliottii* var. *densa*) tree in Big Cypress National Preserve, Florida became a snag between 1993 and 2008 as a function of experimental treatment. Treatments included burning during the dry season and with a short (c.a. 3 years) fire-return interval (dry, short), during the dry season and with a long (c.a. 6 years) fire-return interval (dry, long), during the wet season and with a short fire-return interval (wet, short), and during the wet season and with a long fire-return interval (wet, long). Predictions were generated from the best-supported multinomial logit model.

Key finding 2) Burns conducted at longer intervals increased the probability of snag creation.

We found that live trees on plots burned on a longer interval had a significantly greater probability of becoming a snag (Fig. 7). Return interval had a smaller effect on snag creation than did season of burn (Key Finding #1).

Key finding 3) The probability of mortality of existing snags (snag loss) was greater with wet-season burns.

The mortality rate for snags – that is, the probability that a snag was consumed, fell over, or decayed completely – was significantly lower on plots that were burned during the dry season (Fig. 8). Given the general correlation between fire intensity and the rate of spread, wet-season burns likely moved more slowly, with lower flame heights, and thus may have concentrated heat transfer around the base of trees and snags, which may explain why wet-season burns killed fewer live pine trees. At the same time, by focusing heat transfer at the base of existing snags, wet-season burns may have been more likely to consume sapwood at the base of low-to-moderately decayed snags, weakening structural support and increasing the probability that these snags fell during the course of the study.

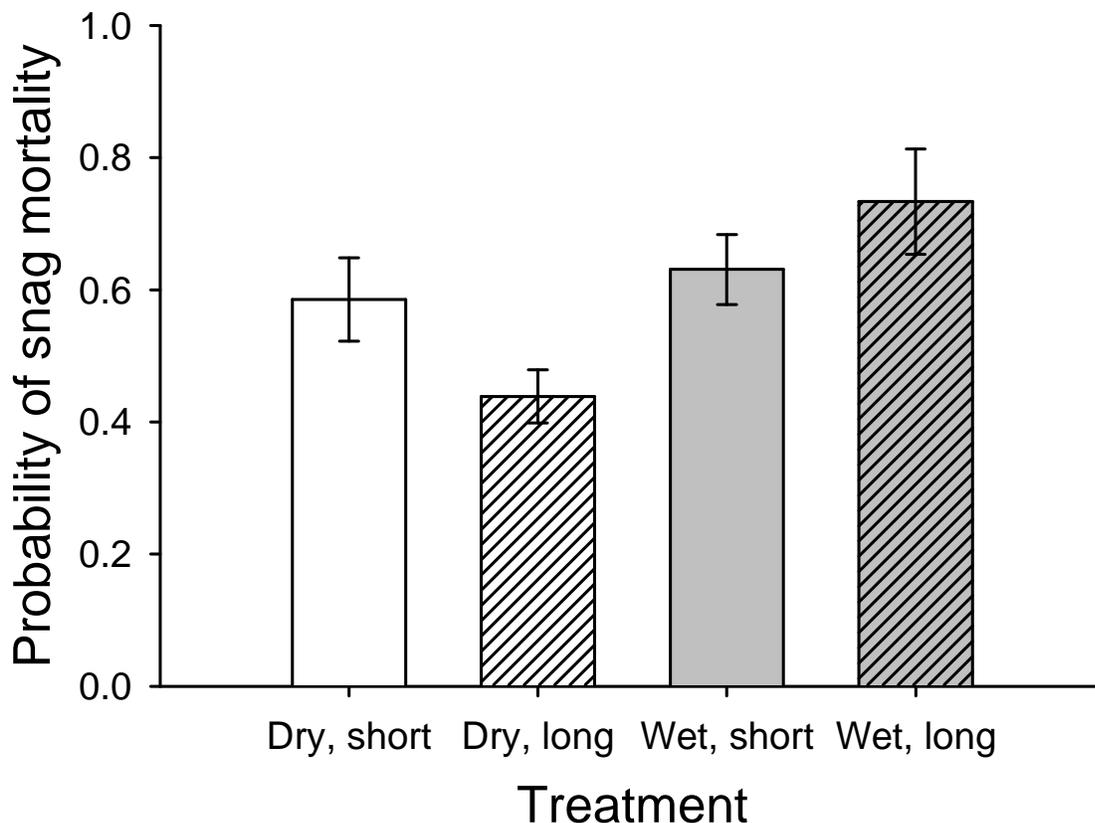


Figure 8. Predicted probability (\pm 95% confidence interval) that a south Florida slash pine (*Pinus elliottii* var. *densa*) snag in Big Cypress National Preserve, Florida fell down or decayed or was consumed by fire to a height of < 1.5 m or a dbh of < 5.0 cm between 1993 and 2008 as a function of experimental treatment. Treatments included burning during the dry season and with a short (c.a. 3 years) fire-return interval (dry, short), during the dry season and with a long (c.a. 6 years) fire-return interval (dry, long), during the wet season and with a short fire-return interval (wet, short), and during the wet season and with a long fire-return interval (wet, long). Predictions were generated from the best-supported multinomial logit model.

Key finding 4) Burning in the dry season and with longer return intervals is predicted to yield greater densities of snags.

Live pines were more likely to become lightly to moderately decayed snags, and snags in this state were more likely to remain in this state, on plots burned during the dry season and on a longer interval, which resulted in a substantially greater projected density of lightly to moderately decayed snags than in other treatments (Fig. 9). In general, these lightly to moderately decayed snags are more valuable to cavity-nesting birds than other decay classes. Dry-season burns were also projected to result in a greater density of heavily decayed snags than wet-season burns, apparently because of an increase in transition probabilities to this state from live trees and low-to-moderately decayed snags (Fig. 10).

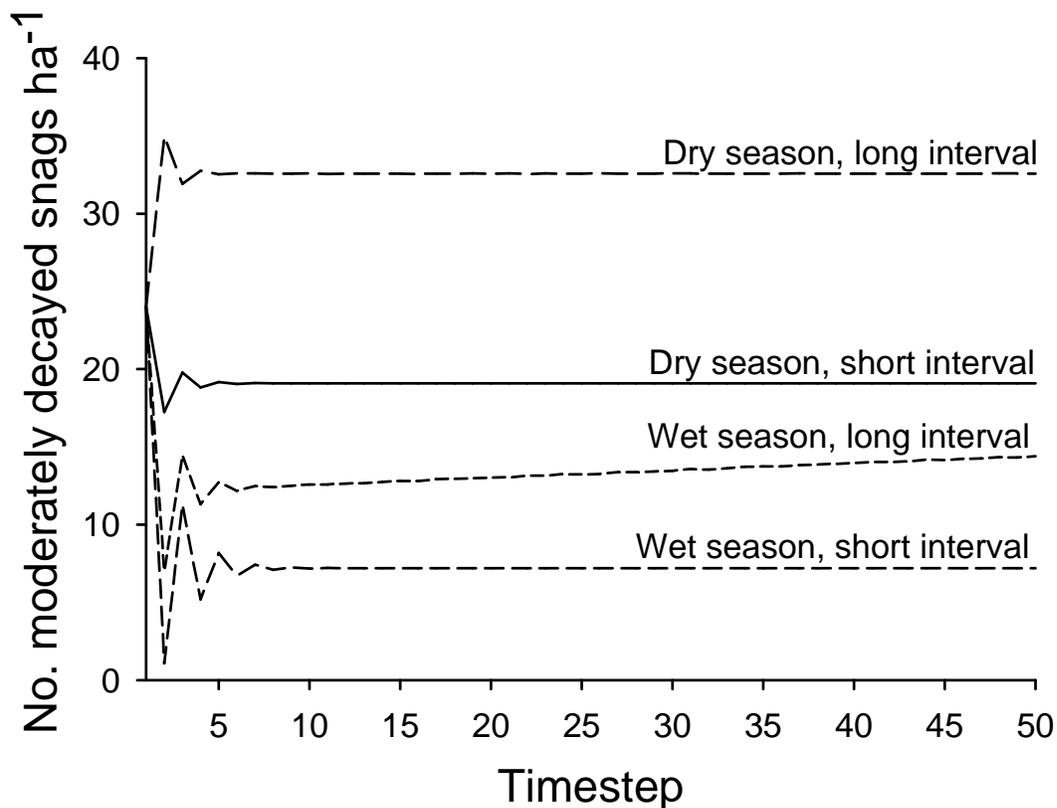


Figure 9. Projected density of low-to-moderately decayed south Florida slash pine (*Pinus elliottii* var. *densa*) snags in Big Cypress National Preserve, Florida as a function of experimental treatment. Treatments included burning during the dry season and with a short (c.a. 3 years) fire-return interval (dry, short), during the dry season and with a long (c.a. 6 years) fire-return interval (dry, long), during the wet season and with a short fire-return interval (wet, short), and during the wet season and with a long fire-return interval (wet, long). Projections were based on beta-distributed random numbers with a mean and variance equal to those predicted from the best-supported multinomial logit model, fit to data on the fate of marked trees and snags between 1993 and 2008. Densities at each 15-year timestep were calculated as the average from 10,000 simulations; 95% confidence intervals were estimated but were too narrow to be represented on the y-axis.

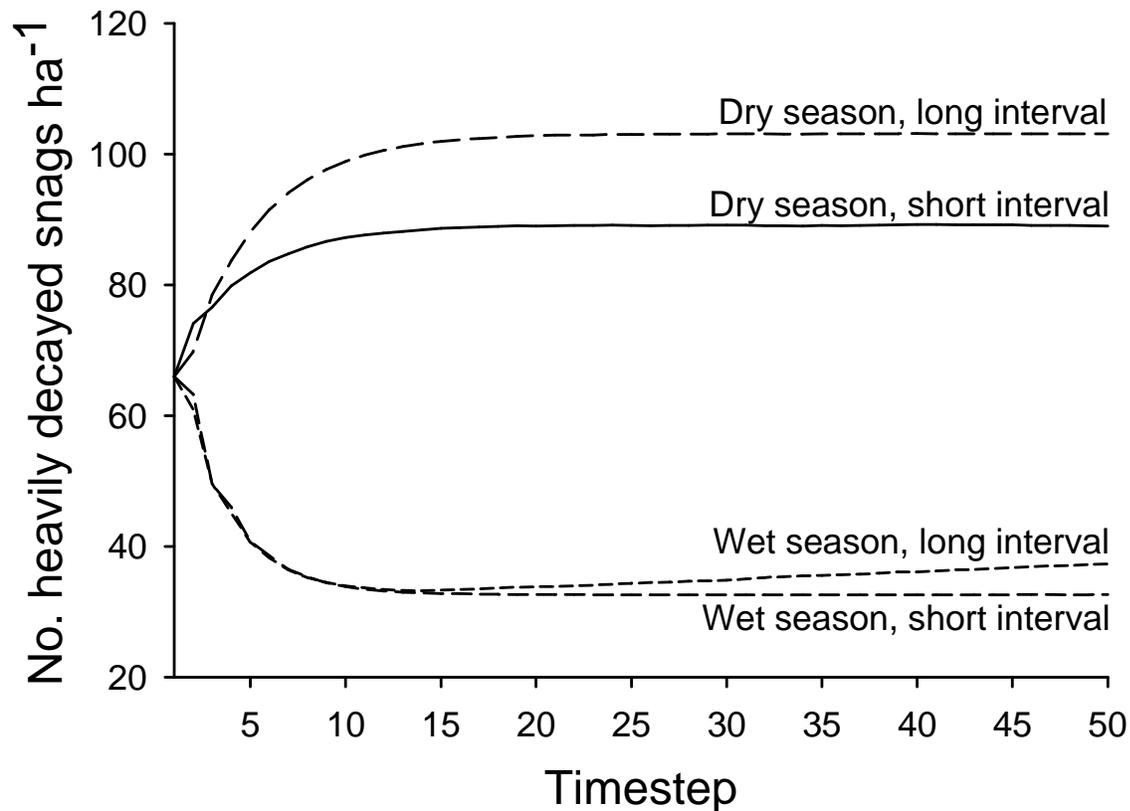


Figure 10. Projected density of heavily decayed south Florida slash pine (*Pinus elliottii densa*) snags in Big Cypress National Preserve, Florida as a function of experimental treatment. Treatments included burning during the dry season and with a short (c.a. 3 years) fire-return interval (dry, short), during the dry season and with a long (c.a. 6 years) fire-return interval (dry, long), during the wet season and with a short fire-return interval (wet, short), and during the wet season and with a long fire-return interval (wet, long). Projections were based on beta-distributed random numbers with a mean and variance equal to those predicted from the best-supported multinomial logit model, fit to data on the fate of marked trees and snags between 1993 and 2008. Densities at each 15-year timestep were calculated as the average from 10,000 simulations; 95% confidence intervals were estimated but were too narrow to be represented on the y-axis.

V. Management implications

Management implication 1) Hydrological restoration in the greater Everglades that results in widespread changes to water-table elevation will effect change in the structure and composition of plant and animal communities even in the pine-dominated uplands.

Annual cycles of flooding and drought play a key role in structuring biological communities in the greater Everglades. As such, “getting the water right” is viewed as the critical element in the restoration of the greater Everglades. Wetland communities are expected to respond strongly to the restoration of a more natural hydropattern, but relatively little is known of how hydrological restoration will affect upland communities. Our results suggest that changes in water-table elevation have the potential to produce large and direct changes in the physiognomy of slash-pine forests and in the structure and composition of the bird assemblages inhabiting these forests. Changes in the structure of the plant community brought about by changes in water-table elevation will likely produce changes in the composition and accumulation rate of fuels, which may produce changes in fire behavior. Changes in fire behavior may in turn produce further changes in the structure of the plant community. Our understanding of how fire shapes the slash-pine forests – including the results presented here - has developed during a period when water-table elevations have been unnaturally low, and as a consequence fire-management plans may require substantial revision as hydrological restoration proceeds.

Management implication 2) Fire-return intervals of up to 5 years are appropriate for maintaining populations of most breeding birds

All of the breeding birds that exhibited a strong relationship with fire history had a similar pattern of abundance, wherein abundance dropped sharply when fire-return intervals extended beyond 5 years. This suggests a general adaptation among breeding birds in this ecosystem to fires that occur, on average, approximately 2 times per decade. No species was positively associated with fire-suppressed conditions; however, data from our experimental study of fire effects on snag populations (Objective 2) suggest that snag densities may decline under short (i.e., 3 years) fire-return intervals and thus species that require snags for nesting may benefit from occasional fire-free periods that last 6-7 years. Incorporating variable fire-return intervals into fire-management planning may be a useful approach for creating conditions suitable to a variety of bird species.

Management implication 3) Density of snags – a key element of habitat for several bird species of concern – can be increased by applying fire at longer intervals and by burning at the transition between dry-season and wet-season.

Prescribed burns could be an effective tool for increasing the density of snags, provided that the fire is sufficiently intense to induce mortality among live trees and moving rapidly enough to limit the consumption of sapwood on existing snags. Prescribed burns conducted early in the wet season might be useful for increasing snag densities as long as fuel and weather conditions are suitable to support a relatively intense fire. As the wet season progresses, and fuel moistures increase, prescribed burns will likely have an

increasingly detrimental effect on snag density because fewer snags will be created and more will be consumed. Longer fire-return intervals might also help optimize the balance between snag creation and snag consumption during a fire, and lengthening this interval on occasion to > 5 years may be beneficial. Doing so would increase the density of low-to-moderately decayed snags, which, in slash-pine forests, afford suitable nest sites for a variety of cavity-nesting birds, including species of management concern such as Eastern Bluebirds, Brown-headed Nuthatches, and Hairy Woodpeckers.

VI. Relationship to other recent findings and ongoing work on this topic

Objective 1) Determine the large-scale patterns of relationship between fire management, hydrology, and abundance and distribution of breeding and wintering landbirds and their habitats.

There has been relatively little study of the role that hydrology plays in shaping the composition of slash-pine forests. However, hydrology is an important determinant of species composition and physiognomy in other southeastern pine ecosystems (Peet and Allard 1993, Allen et al. 2006), and Duever (2005) argued that hydroperiod and water-table elevation played a major role in shaping the composition of slash-pine forests in BCNP. Fire has received more attention as a key ecosystem process, and the seminal work of Robertson (1953) and Alexander (1967) established the controlling role of fire in preventing the succession of slash pine to hardwood hammock. More recently, Slocum et al. (2003) found that higher-elevation locations in the pine rocklands at Long Pine Key tended to experience more intense and uniform fires than did relatively low-elevation locations in the same area. This pattern may have arisen due to an interaction with hydrology, whereby fuel moistures were greater at the wetter, lower-elevation sites. Although he presented no data in support of his contention, Loveless (1959) made a similar argument, suggesting that the effects of fire were strongly influenced by water levels.

Almost nothing is known of the response of birds in slash-pine forests to variation in hydrology. Working in a longleaf-pine forest, Allen et al. (2006) found substantial variation in the structure and composition of a breeding-bird community along moisture gradients, although the gradient in that study encompassed non-pine plant communities (e.g., pocosins). The few studies that have examined the effect of fire on birds in slash-pine forests have not found any strong relationship between bird abundance and fire history (Emlen 1970, Johnson and Landers 1982), although both studies examined a limited range of fire-return intervals (the maximum time since fire was 5 years). Given our results, which indicate that most species tolerate fire-return intervals of up to 5 years with no appreciable effect on abundance, studies examining the importance of fire may need to include a wider range of fire histories.

Objective 2) Investigate the role of fire in snag dynamics.

Elevated rates of snag creation observed in the dry-season treatments were due in large part to increased mortality of live trees. Whether this pattern was actually a result of season of burn is unclear, however. The average height of charred bark, a good indicator of fire intensity (e.g., Wade and Johansen 1986, Williams et al. 1998), tended to be greater during dry-season burns: fire intensities during the dry season were medium (91 – 235

BTU sec⁻¹ ft⁻¹) and were low (1 – 90 BTU sec⁻¹ ft⁻¹) to medium during the wet season (using the scale in Byram 1959). Fire intensity, in turn, is often a good predictor of mortality of pine trees following fire (Glitzenstein et al. 1995, Menges and Deyrup 2001, Snyder et al. 2005). Although several *a priori* reasons exist to expect that season of burn may affect the ecological outcome of fire (see review of hypotheses in Glitzenstein et al. 1995), in this case season is confounded with variation in fire intensity and in general there is scant evidence for consistent seasonal patterns in the vulnerability of south Florida slash pine and related species to fire, in part because fire intensity does not vary consistently with season of burn (see, e.g., Snyder 1986, Glitzenstein et al. 1995, Menges and Deyrup 2001, Snyder et al. 2005). Although higher rates of tree mortality – presumably a consequence of more intense fires – were the primary reason that the dry-season burns were projected to yield higher densities of snags, dry-season burns also had lower rates of snag mortality, particularly for low-to-moderately decayed snags. Again, this was likely a consequence of differences in the behavior of fires initiated in different seasons. Given the general correlation between fire intensity and the rate of spread (Rothermel 1972), wet-season burns likely moved more slowly, with lower flame heights, and thus may have concentrated heat transfer around the base of trees and snags (Wade and Johansen 1986). Snyder (1986) reported a similar phenomenon, in which a wet-season prescribed burn in the pine rocklands of Everglades National Park produced much hotter median ground-level temperatures (316°C v. 232°C) than a faster and more intense dry-season burn conducted on a nearby study plot. Wade and Johansen (1986) argued that most fire-related mortality in southern pines with dbh > 5.0 cm is due to crown damage, rather than cambial damage on the tree bole or root damage, which may explain why wet-season burns killed fewer live pine trees. At the same time, by focusing heat transfer at the base of existing snags, wet-season burns may have been more likely to consume sapwood at the base of low-to-moderately decayed snags, weakening structural support and increasing the probability that these snags fell during the course of the study. Snags of south Florida slash pine that had lost all sapwood from ground level to approximately 1 m above ground level were frequently observed (J. D. Lloyd and G. L. Slater, personal observation), and may reflect the action of slow-moving, low-intensity ground fires that may be typical of wet-season burns.

Longer fire-return intervals generally result in increased fuel loads, more intense fires, and higher mortality of live trees (Sackett 1975, Platt et al. 1991, Outcalt and Wade 2004). To the contrary, we found no evidence of consistent variation in fire intensity among plots burned at different intervals, which may explain why the effect of fire-return interval on rates of snag creation and mortality was somewhat muted relative to the effect of season of burn. In general, shorter return intervals increased the mortality rate of live trees, but more of these trees had fallen down by the end of the study than on plots burned at longer intervals. This pattern could have arisen either because short-return intervals increased the probability of live trees transitioning directly to the down category or because short-return intervals hastened the process of decay (i.e., trees that were alive at the beginning of the study were killed by fire and transitioned through one or both decay states to the down category prior to the final sampling period). Given that fire intensity did not vary with return interval (at least for the first two intervals), we have no reason to believe that live trees were more likely to be consumed or toppled by fire on plots with shorter return intervals. Rather, we believe that the latter scenario is more likely, and that short-return intervals simply hastened the transition from live tree to snag to downed

wood. Indeed, the estimated lifespan of snags in decay state 1 was somewhat lower on plots burned at shorter intervals. Holden et al. (2006) reported similar findings for snags of ponderosa pine (*Pinus ponderosa*), in that snag density in areas burned multiple times was significantly lower than snag density in areas that had only been burned once. They attributed this result to the gradual consumption of wood at the base of snags during successive fires, which reduced structural support for the snag and led to increased fall rates.

Our results suggest that prescribed burns could be an effective tool for increasing the density of snags, provided that the fire is sufficiently intense to induce mortality among live trees and moving rapidly enough to limit the consumption of sapwood on existing snags. In this study, these conditions occurred during burns conducted during the dry season and on plots with longer fire-return intervals. Other studies have found the opposite (e.g., Glitzenstein et al. 1995, Snyder et al. 2005), and thus we believe that our results reflect less on the role of season of burn than they do on the role of fire intensity, which may not vary predictably among seasons. For example, Snyder (1986) found that prescribed burns conducted during the wet and dry seasons had similar characteristics (in terms of fuel consumption, rate of spread, and intensity) when fuel moisture was similar, but that the characteristics of dry- and wet-season burns diverged strongly when conducted with different fuel moisture levels. Indeed, any effort to increase the density of snags should be incorporated into the overall goals of a fire management plan, most of which now seek to replicate natural fire regimes (e.g., Slocum et al. 2003). In southern Florida, lightning-started fires are most frequent in July, although the area burned by lightning-started fires peaks in May during the transition between dry and wet seasons when lightning strikes are frequent and fuels remain dry (Komarek 1964, Doren et al. 1993, Gunderson and Snyder 1994). Thus, whereas prescribed burns used to be most common from October – March (Snyder 1991, Gunderson and Snyder 1994), they tend now to be initiated during the early wet season (Slocum et al. 2003). Prescribed burns conducted early in the wet season might be useful for increasing snag densities as long as fuel and weather conditions are suitable to support a relatively intense fire. As the wet season progresses, and fuel moistures increase, prescribed burns will likely have an increasingly detrimental effect on snag density because fewer snags will be created and more will be consumed.

Longer fire-return intervals might also help optimize the balance between snag creation and snag consumption during a fire. All things being equal, longer fire-return intervals should allow for the build-up of greater fuel loads, which in turn should promote more intense fires (Rothermel 1972, Sackett 1975). The historic fire-return interval in the slash pine forests of southern Florida is thought to have ranged from as short as 2-3 years to as long as 10 or 15 years, although general agreement exists that, on average, most areas would have burned at least every 5 years (Snyder et al. 1990, USFWS 1999, Slocum et al. 2003). After many years of fire suppression, followed by infrequent fires that were often set outside of the historic fire season, prescribed fire regimes now generally attempt to mimic the putative natural pattern of fire and tend to use short fire-return intervals; for example, the pine rocklands of Everglades National Park have been burned on a 2-3 year rotation (Slocum et al. 2003). In situations where snags suitable for use by cavity-nesting birds are limiting, which McComb et al. (1986) argue is generally the case for all of Florida's slash pine forests, lengthening this interval in some areas, perhaps to 4 – 6 years, may be

beneficial. Doing so would increase the density of low-to-moderately decayed snags, which, in slash pine forests, afford suitable nest sites for a variety of cavity-nesting birds, including species of management concern such as Eastern Bluebirds, Brown-headed Nuthatches, and Hairy Woodpeckers. Other considerations may preclude burning large blocks of slash pine forest at longer intervals, but a similar effect might be achieved by relying on ignition methods that promote patchy fires (e.g., single-source ignition methods) and result in a mosaic of return intervals with at least some patches remaining unburned for longer periods.

VII. Future work needed

1) Predictive models of the response of pineland birds to the changes in hydrology (e.g., water-table elevation)

We have described a pattern in which variation in water-table elevation is the primary correlate of large-scale (i.e., region-wide) variation in vegetation structure and in the distribution and abundance of breeding and wintering birds. However, our analysis is purely exploratory, and confirmatory research, utilizing a study design explicitly targeted at revealing the effects of variation in water-table elevation and other hydrological variables (i.e., hydroperiod) is needed. Future work on this question should take advantage of naturally occurring gradients, at a variety of spatial scales, to test hypotheses about the influence of water-table elevation on birds and plants. This work should also recognize that large-scale gradients in water-table elevation are confounded with many other factors (soil type, biogeographical history, climate) and that small-scale gradients may be confounded with variation in fire history.

2) Predictive models of the response of fuels to changes in hydrology (e.g., water-table elevation)

By using ordination analyses that identify orthogonal (i.e., uncorrelated) axes of variation, we have identified how hydrology and fire can shape the structure and composition of the South Florida slash-pine ecosystem independent of one another. In reality, however, hydrology and fire interact with one another to shape biological communities. We have identified spatial variation in vegetation structure that is correlated with spatial variation in water-table elevation, and it is reasonable to assume that temporal variation in water-table elevation and hydroperiod, as will accompany restoration of normative hydrological processes, will produce similar changes within a location. These changes in vegetation structure will likely produce changes in the composition of fuels and in the rate at which they accumulate, both of which may influence fire behavior. These changes could be modulated by accompanying changes in fire-management plans (e.g., increase rate of fuel accumulation might mandate a shortened fire-return interval), but refining these plans will require a better understanding of how changes in water-table elevation will change patterns of fuel accumulation. Again, these future studies might take advantage of existing gradients in water-table elevation as means for predicting how conditions at a particular location may change over time. Future studies addressing these questions should identify relevant response variables with the input of fire managers.

3) Identification of ecological targets for monitoring the response of the south Florida slash pine ecosystem to changes in water-table elevation and other hydrological factors (e.g., hydroperiod)

Predictive models can be a useful tool in evaluating management alternatives, but should always be verified with monitoring data. Understanding the response of the South Florida slash-pine ecosystem to hydrological restoration of the greater Everglades will require long-term monitoring, but the best targets for such monitoring remain unidentified. Based on our results, measures such as the relative understory dominance by hardwoods and palms, the height of the understory, growth rate of slash pine, basal area of slash pine, size-class distribution of slash pine, and density of pine snags might all be useful in assessing the effects of hydrological restoration on this ecosystem. However, confirmatory studies – such as those that might also address needs 1 and 2 – would be useful in identifying the most sensitive and biologically significant targets for long-term monitoring.

4) Identification of ecological targets for monitoring the response of slash-pine ecosystem to variation in fire regime

Although the effects of fire on plants and animals in the South Florida slash-pine ecosystem are better understood than are the effects of hydrology, long-term ecological monitoring is needed for continued refinement of the scientific basis underlying existing fire-management plans. In addition to monitoring fuels, the efficacy of fire-management plans might be enhanced by identifying targets for monitoring that relate directly to ecological goals for slash-pine forests. In some cases, these may overlap with targets for fuels monitoring. Based on our results, many of the same response variables that might be measured for monitoring the effects of changes in water-table elevation (see need 3) could also be used to monitor the long-term effects of variation in fire regime.

5) Research into the factors that drive snag recruitment and mortality.

Snags are both an important fuel source and an important ecological attribute. They are an essential component of habitat for cavity-nesting birds, many of which are considered to be species of high importance by land managers. Our research highlights the relationship between fire and snag dynamics, but more detailed monitoring is needed to improve our understanding of the links between prescribed fire, fire intensity, fuel moisture, and hydrology with snag recruitment and mortality. In addition, this information needs to be assessed with the knowledge that many remaining pine forests in south Florida are small and fragmented, and thus attempts to mimic the full variation of naturally occurring fires through prescribed fire may prove difficult.

6) Research into factors that drive variation in the distribution and abundance of cavity-nesting birds.

The extirpation of as many as five cavity-nesting birds in sites throughout the south Florida slash pine ecosystem indicates the sensitivity of this avian group to management actions, particularly fire. Snag abundance is an important factor in regulating populations of this group and the long-term viability of existing and reintroduced populations will rely

on effective management strategies (i.e., prescribed fire) that maintain snags. Understanding how the distribution and abundance of cavity-nesters varies with respect to snag abundance and other factors will provide critical information to guide fire management, including the development of ecological targets for cavity-nesting birds.

7) Research into factors that drive variation in the distribution and abundance of wintering birds

The slash-pine forests of south Florida are an important wintering ground for a variety of species, including Yellow-rumped Warbler and Palm Warbler, yet little is known of the habitat requirements of species wintering there. Evaluating the effects of management actions (most importantly, efforts at restoring natural hydrological and fire regimes) on wintering birds is difficult absent a better understanding of the factors that shape their distribution and abundance. Given the importance of fruit in the diet of many wintering birds (e.g. Eastern Bluebird, Yellow-rumped Warbler, and American Robin), future studies might profitably examine the relationship between bird abundance and fruiting shrubs, and whether variation in fire regime has any effect on the availability of fruit.

IX. Status of deliverables

Deliverable	Description	Date delivered
1st Annual Progress Report	Summarization of winter bird surveys; compilation of snag data.	Completed
2nd Annual Progress Report	Summarization of first year of data collection; preliminary results of fire, fuel, bird models.	Completed
3rd Annual Progress Report	Summarization of second year of data collection; preliminary results of snag models.	Completed
Final Report	Report will include executive summary, introduction of topic, description of methods, work results (including all models, figures, graphs, and tables), discussions, and conclusions.	Completed
Peer-reviewed article	Snag dynamics in the pine rocklands: the role of fuel treatments.	In revision; Canadian Journal of Forest Research
Peer-reviewed article	The ecological effect of fire management: large-scale patterns between breeding and wintering birds and wildland fuels in south Florida	In preparation
Peer-reviewed article	Effects of fuel treatment on the biological condition of pine rocklands as represented by bird communities	In preparation
State of the Science Symposium	Symposium to disseminate results of this study, present ongoing work by other researchers and managers working in the pine rockland ecosystem, develop a biennial action plan, and encourage inter-agency communication and collaboration	In preparation; Pine Rockland Conference February 2010

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