ABUNDANCE, DISTRIBUTION, AND BREEDING-SEASON HABITAT REQUIREMENTS OF MANGROVE LANDBIRDS IN SOUTHERN FLORIDA

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Cover photos by: John Lloyd (kayak), Steve Mullin (Mangrove Cuckoo), and Marc Ryckaert (mangrove island) and Laura Gooch (Cuban Yellow Warbler) via Wikimedia Commons.
EXECUTIVE SUMMARY

Florida’s mangroves are home to 6 bird taxa that occur nowhere else in North America: White-crowned Pigeon, Gray Kingbird, Mangrove Cuckoo, and Black-whiskered Vireo, and endemic subspecies of Yellow Warbler and Prairie Warbler. We know very little about any of these birds. Basic patterns of distribution and abundance remain unknown, as do habitat requirements. Effective conservation measures cannot be implemented absent this information. To address these gaps in our knowledge, we initiated a study with the goal of developing a better understanding of the assemblage of birds breeding in the mangroves. We focused on three objectives. We generated estimates of abundance and distribution, quantified habitat associations, and tested and developed a series of protocols for conducting long-term population monitoring of these birds to help inform future conservation efforts.

We conducted 1,425 surveys, each 10 minutes in length, at 347 points on public land in south Florida between 2008 and 2011. We found that the assemblage of mangrove landbirds was numerically dominated by 4 widespread and common species. Red-bellied Woodpeckers were the most widespread and most abundant, followed by Northern Cardinal, Prairie Warbler, and White-eyed Vireo. Two other mangrove-dependent birds, Black-whiskered Vireo and White-crowned Pigeon, were also relatively abundant, although White-crowned Pigeons had a narrow distribution and were not found outside of the Keys and Florida Bay. Black-whiskered Vireos were relatively widespread. Gray Kingbirds and Yellow Warblers were not common overall but were abundant in the few places that they occurred, primarily the Keys and islands of Florida Bay.

Of the mangrove specialists, Mangrove Cuckoo was by far the rarest species. It occurred most commonly in the Ten Thousand Islands region, on the northern end of Key Largo, in the fringing forests along Biscayne Bay in Biscayne National Park, and along some of the major rivers of the southwest coast in Everglades National Park. We suspect that this species is more common than our surveys indicate because it vocalizes infrequently and is thus difficult to detect even when it is present. In an experiment conducted in 2010 and 2011, we found that the use of recorded vocalization significantly increased the rate of detection for cuckoos. We suggest that future efforts to survey or monitor this species should employ call-playbacks during surveys.

We found few significant relationships between vegetation type and the abundance of any species and found no examples in which a species was limited to any particular type of mangrove forest or shrubland. However, some of the distributional patterns that we observed were likely related to habitat preferences. For example, Pileated Woodpeckers declined in abundance from north to south,
which may reflect a similar latitudinal trend in the availability of large trees suitable as nest sites. White-crowned Pigeons were limited to the far southern portion of the study area, probably by the joint distribution of poisonwood, an important food species, and predator-free islands for nesting. Many other species showed restricted distributions that were not obviously related to habitat. Carolina Wren, American Crow, and White-eyed Vireo were rarely found south of the mainland, even though apparently suitable habitat exists further south. Conversely, Yellow Warbler and Gray Kingbird were found only in the Keys and Florida Bay and did not occupy similar vegetation types further north. Future studies of habitat use among mangrove landbirds should focus on how individual birds use their environment, for example by quantifying vegetation structure at nest sites or on individual territories or home ranges.

Implementing a long-term population-monitoring program for mangrove landbirds would significantly enhance prospects for the conservation of this assemblage. Our results indicate that such a program is feasible. Adequate power to detect biologically important trends could be achieved by sampling approximately 150 point every 5 years, which requires a relatively small commitment of resources. Methodological challenges that we identified included determining an appropriate sampling frame, identifying robust methods for accounting for imperfect detectability of birds during surveys, and incorporating methods to enhance the detectability of Mangrove Cuckoos, which proved difficult to detect using standard point-count methods. Identifying reliable funding sources is the primary logistical challenge that must be overcome to implement a monitoring program.
INTRODUCTION

Mangrove ecosystems, or mangroves as they are known colloquially, are characteristic of sheltered tropical and subtropical marine shorelines. They are characterized by a group of taxonomically unrelated tree species – also known individually as mangroves – that have in common a suite of adaptations, some quite spectacular, that allow them to thrive in anoxic, saline environments (Lugo and Snedaker 1974). Mangroves provide many economically valuable services and, not coincidentally, most aspects of the structure and function of mangrove ecosystems, especially in south Florida, have been well studied (e.g., Odum et al. 1982). Surprisingly, one of the most significant gaps in our understanding of this system concerns the faunal assemblages that inhabit mangroves (Snedaker 1989). Odum et al. (1982) compiled lists of the fish, herptiles, mammals, and birds believed to inhabit Florida’s mangroves during some portion of their life, but many key pieces of information about these assemblages were unavailable then and remain so today. For example, Odum et al. (1982) found few species for which quantitative estimates of abundance or distribution were available, and equally few for which habitat requirements were well described. Nearly three decades later, relatively little progress has been made in addressing these information gaps.

In this study, we focused on filling some of these gaps for the arboreal landbirds that breed in Florida’s mangroves. Research on mangrove landbirds is needed because of this group’s significance to biodiversity conservation; six of the landbirds that breed in Florida’s mangroves are found nowhere else in North America. Two of these, Yellow Warbler (Dendroica petechia gundlachi) and Prairie Warbler (Dendroica discolor paludicola), are subspecies endemic to mangroves (Howell 1930, Bond 1930, Buerkle 1999, AOU 1983), whereas the remaining four – White-crowned Pigeon (Patagioenas leucocephala), Mangrove Cuckoo (Coccyzus minor), Gray Kingbird (Tyrannus dominicensis), and Black-whiskered Vireo (Vireo altiloquus) – occur in other vegetation types outside of North America, but in Florida rely on mangroves for habitat. With the exception of Gray Kingbird, which appears to be expanding its breeding range beyond the mangroves, all are listed as “species of conservation concern” by the US Fish and Wildlife Service. This designation is reserved for “migratory nongame birds that, without additional conservation actions, are likely to become candidates for listing under the Endangered Species Act (ESA) of 1973” (USFWS 2008).

The primary causes of concern for these species are their limited North American distribution, presumed small population sizes, and uncertain population trends (Rich et al. 2004). Addressing these uncertainties, and implementing effective conservation strategies where needed, will require that we quantify basic ecological patterns of abundance and distribution and identify habitat requirements. It
will also require developing and implementing an effective population-monitoring program, because at present mangrove birds are not monitored by any standardized, regional monitoring scheme. Without population monitoring, natural-resource managers lack any means by which to identify species at risk or to evaluate the effects of management actions on bird populations, effectively precluding the use of adaptive management principles. Each of these issues – inventoring the assemblage of breeding landbirds, identifying habitat relationships, and developing a protocol for long-term monitoring of populations – was identified as a research priority by an informal working group formed in 2003. The group consisted of Gary Slater of Ecostudies Institute, Terry Doyle of Ten Thousand Island National Wildlife Refuge, Karl Miller of Florida Fish and Wildlife Conservation Commission, and Sonny Bass of Everglades National Park.

This report describes progress made in achieving the three primary objectives identified by the working group. In particular, we discuss patterns of distribution and abundance and habitat associations, and provide recommendations for developing a long-term monitoring protocol capable of tracking the health of mangrove landbirds populations in the future. In addressing the last objective, we evaluated four issues considered important by the working group to the development of a monitoring program: sample design, sample size, timing of sampling, and analysis of data. During the course of the study, we added a fifth component to this objective based on our initial experiences conducting surveys. In particular, we became concerned that the point-count method used to conduct surveys of bird populations was inadequate for detecting Mangrove Cuckoos, and so we implemented an experiment to determine whether the use of recorded vocalizations could increase detection rates for this species.

METHODS

Study area

We conducted this study in the mangrove forests of southern Florida (Fig. 1). All survey locations were on public land in Great White Heron National Wildlife Refuge, National Key Deer Refuge, John Pennekamp Coral Reef State Park, Biscayne National Park, Everglades National Park, Collier-Seminole State Park, Fakahatchee Strand Preserve State Park, Ten Thousand Islands National Wildlife Refuge, and Rookery Bay National Estuarine Reserve. In total, we sampled the landbird assemblage at 347 points (Figs. 2-5): 108 were in the Ten Thousand Islands region (mostly in Ten Thousand Islands National Wildlife Refuge and Rookery Bay National Estuarine Research Reserve; 8 points fell just outside the refuge boundaries in Collier-Seminole State Park and Fakahatchee Strand Preserve State Park), 163
in Everglades National Park, 37 in Biscayne National Park, 11 in John Pennekamp Coral Reef State Park, and 38 in Great White Heron National Wildlife Refuge and the National Key Deer Refuge.

Mangroves occupy most of south Florida's coastline, ranging as far north as Cedar Key on the Gulf coast and Ponce de Leon inlet on the Atlantic coast (USFWS 1999), but are most extensive and well developed in coastal areas between Naples and Miami. Florida’s mangroves range in structure from shrublands to open woodlands to tall, closed-canopy forests, and are dominated by four different tree species: red mangrove (*Rhizophora mangle*), black mangrove (*Avicennia germinans*), white mangrove (*Laguncularia racemosa*), and buttonwood (*Conocarpus erectus*). Forests in the Ten Thousand Island Region and along the tidal rivers north of Cape Sable in Everglades National Park consist of mixed-species forests and are especially well-developed, with average canopy height often exceeding 15 m (Simard et al. 2006). Canopy height and structural complexity declined towards the ecotone with inland freshwater marsh, although some of the inland survey locations included large tropical hardwood hammocks.

Survey locations in the Whitewater Bay area of Everglades National Park were dominated by relatively short and dense stands of red mangrove. Further south, along the coast of Florida Bay, open woodlands of buttonwood and black mangrove mixed with coastal prairie to form an almost savannah-like ecosystem. The islands of Florida Bay were generally covered by a narrow fringe of mangroves, usually a mix of red and black mangrove, with an open area of salt pans and coastal prairie in their center. Many of the islands also had small patches of hardwoods and stands of buttonwood. The forests along the inland coast of Biscayne Bay were intermediate in height, generally dominated by a mix of black, red, and white mangrove, and were relatively narrow and constrained on their inland side by canals and levees. In the Keys, mangroves were uniformly short and tended to be dominated by a mix of red and black mangrove, with some white mangrove in interior portions. Some of the mangroves in the Keys were notably dry and had adopted a shrubby physiognomy. Representative photographs of habitat types are provided in Appendix B.

**Establishing survey points**

We began the study in 2008 by establishing 87 points at which to conduct bird surveys. We established these points systematically in mangrove forests in Everglades National Park and Biscayne National Park, placing points in locations accessible by foot, kayak, or power boat. Points were located along tidal creeks, hiking trails, and on islands in Florida Bay. Points were separated from one another by at least 350 m. We also required that the area within a 150-m radius around each point have > 50% cover by mangrove vegetation (red mangrove, black mangrove, white mangrove, or buttonwood, alone or as mixed-species assemblages).
We established 152 new survey points during 2009, adding to the 87 established in 2008, using a general randomized tessellation stratified (GRTS) survey design (Stevens and Olsen 2004). First, we defined the sampling frame using the Florida mangroves shapefile created by the Florida Fish and Wildlife Commission (available for download at the Florida Geographic Data Library; http://www.fgdl.org). This shapefile was based on land-use and land-cover data from Florida’s water management districts. We laid a systematic grid of points throughout the polygons containing mangrove vegetation. Each point in this grid represented a potential survey location. There were 9,654 points in the grid; points were separated from each other by 500 m. We then used S-Draw (T. L. McDonald, unpublished software; available at http://www.west-inc.com) to generate a GRTS sample of 1,000 potential survey points.

We then applied two decision rules to determine which of the 1,000 potential points would be included in the survey. Beginning with the point drawn first, we applied the first decision rule:

1) Is the point accessible?
   a) Yes; the selected point is along an existing trail, on a navigable creek <50 m wide, or on an island that can be surveyed on foot. Consider further.
   b) No; the point is not accessible. Discard and move to the next selected point.

Points for which we answered “Yes” to the first decision rule were used as the midpoint of a route of up to 10 points, each separated by 350 m. A route was defined as a series of points, each directly accessible from at least one other point in the route, that followed a navigable creek or series of navigable creeks or that followed an existing trail. When the selected point was on an island, additional points were added as space allowed, although many islands contain routes consisting of only the single point selected in the GRTS draw. Once all potential routes were identified, we then applied a second decision rule:

2) Does the area within a 150-m radius of the point contain at least 50% mangrove vegetation based on existing land cover data, inspection of high-resolution digital orthophotos, and physical inspection of the site?
   a) Yes; survey points along the route.
   b) No; discard route.

This process defined our sampling frame, and thus the entity about which we can draw inference. The sampling frame for this study can be described as: all areas in south Florida containing mangrove vegetation that are within 150 m (the width of our survey area around each point) of a navigable creek.
<50 m wide, islands containing mangrove vegetation on which observers can move on foot, and areas containing mangrove vegetation that are within 150 m of an existing trail.

In addition, in 2009, we added 108 existing points at Ten Thousand Islands National Wildlife Refuge that had been surveyed during previous years by Terry Doyle. Although these points were not randomly selected during the GRTS draw, we included them in our sample because they provide continuity with existing, baseline data.

**Bird surveys**

We surveyed bird populations at each of the 347 points in the study area, although not every point was surveyed in each year. We surveyed the original 87 points in 2008, 347 points in 2009, 325 points in 2010, and 331 points in 2011. Reasons for omitting points in 2010 and 2011 included the time required to access them (Johnston Key and Horseshoe Key), the determination that the points did not meet our criteria for coverage by mangrove vegetation (Oil Pad Road in Ten Thousand Islands National Wildlife Refuge), and various logistical issues that prevented us from reaching the points (e.g., points on Big Torch Key were flooded in 2010). Some points were surveyed more than once in each year; when that occurred, we based analyses on average counts. In 2008, we conducted 122 surveys between 1 May and June 4. In 2009, we conducted 445 surveys from 25 April until 21 June. In 2010, we conducted 423 surveys from 26 April to 28 June. In 2011, we conducted 433 surveys from 1 May to 22 June. The range of dates during which we conducted surveys was based on the experience of other biologists who had conducted bird surveys in mangrove forests in Florida (T. Doyle, USFWS, personal communication) and on our own observations of when singing activity was at its peak (see Results, *Optimal seasonal timing of surveys*).

Surveys began at sunrise and continued for up to 3 hours. Observers were given the discretion to end surveys earlier if they observed that singing rates were declining (e.g., due to elevated air temperatures) or if their perceived ability to detect birds was declining, for example due to increasing wind speeds. Surveys were not conducted in inclement weather (e.g., moderate or heavy rain) or when average wind speed was in excess of 12 km per hour. Surveys consisted of a ten-minute count period, during which observers recorded the first detection of each individual and assigned that detection to one of ten 1-minute intervals. Once an individual was detected, it was not recorded in subsequent intervals. Observers also assigned each detection to one of five distance bands: 0-25 m from the survey point, 25.1-50 m, 50.1-100m, 100.1 m-150m, and > 150 m.
Statistical analyses

Abundance and distribution of mangrove landbirds.

We generated estimates of abundance in 4 ways. First, we estimated the mean number of detections per point per species. We used a non-parametric bootstrap routine to estimate the variance and 95% confidence intervals around estimated means. Second, we estimated the mean number of detections at points where a species was detected at least once, and estimated the variance and 95% confidence intervals using a non-parametric bootstrap routine. Summarizing the average number of detections at points where a species was detected at least once eliminated the distorting effect of averaging across points that either did not include suitable habitat or that were outside of the geographic distribution of a species. Neither of these methods accounts for the imperfect detectability of individual birds, so we also estimated average abundance for each species after correcting for detectability using removal models and distance sampling.

We used Program SURVIV to implement the removal models of Farnsworth et al. (2002). As applied to point counts, removal models estimate abundance based on the decline in the number of new individuals detected through time. In the case of our study, the decline was measured as the number of new individuals recorded in each of the one-minute intervals that composed a point count. In theory, the probability of detection generated from removal models reflects both the probability that an individual is available for detection (e.g., by singing or making itself visible to an observer), given that it is present, and the probability that an individual available for detection is detected by an observer (e.g., the observer hears and correctly identifies a singing bird) (Nichols et al. 2008). In estimating detectability, we used all detections, both visual and aural. Visual detections accounted for only 8.3% of all detections. We did not estimate detectability for species with <99 detections because the results were unreliable (e.g., confidence intervals for detectability that included large, negative values).

We used Akaike’s Information Criteria (AIC) to select between models $M_c$, which assumes that individuals of a species differ in their likelihood of detection (i.e., it assumes individual heterogeneity in detectability) and $M$, which assumes all individuals of a species have the same likelihood of detection (Farnsworth et al. 2002). We also used AIC to compare the same models when detections were grouped into 5, 2-minute intervals. The model with the lowest AIC was used to estimate the probability of detection, $p$. We then used $p$ to calculate the total number of individuals estimated to have occurred within the area surveyed (i.e., the total number counted as adjusted for $p$). We divided the estimated total number of individuals in the survey area by the total area surveyed to calculate density. In estimating density, we excluded uninhabitable space such as open water, buildings, or roads from our
estimate of the total area surveyed, and so our estimates of density accounted for differences among points in the amount of potentially habitable areas included within the count circle (see *Habitat associations of mangrove landbirds* for details on methods). We also estimated average abundance per point, calculated as the total number of individuals per species divided by the number of surveys conducted. Standard errors for density estimates were calculated using the formula in Farnsworth et al. (2002). Standard errors for the estimates of the total number of individuals and per-point abundance were calculated using the formula in Nichols et al. (2000).

We also used distance sampling to generate estimates of abundance corrected for detectability. We used multiple-covariate distance sampling (Marques et al. 2007) as implemented in Program Distance (Thomas et al. 2010) to estimate the probability of detection, density, and to determine the effect of covariates on detectability. We considered two covariates: date of the survey and time of the survey. We also attempted to fit models with habitat and observer as covariates, but found that the likelihoods for these more complex models could not be estimated. We began with a simple model without covariates, one in which detectability declines with distance as a half-normal function, generally a useful model for the analysis of point-transect data (Buckland et al. 2001). We also evaluated models with a hazard-rate key function, but found that they rarely converged, and so analyses were conducted with the half-normal function. Building on the initial model, we added covariates one at a time, stopping when the more complex model failed to converge. For example, we began with a model describing detectability as a half-normal function. To this model, we added a covariate for the effect of date and then, in a separate model, an effect of time of day. If these models converged, we created a fourth model with both date and time of day as covariates. We checked for convergence by comparing the log-likelihood of the model with covariates to the model without covariates; the simple model, nested within the more complex model, should always have a log-likelihood greater than or equal to the more complex model.

We assessed the degree of support for each converged model using AIC, as adjusted for small sample size (*AICc*) (Burnham and Anderson 2002). We used only the best-supported model for inference. In general, multi-model inference (i.e., using parameter estimates averaged among competing models) is thought to better address uncertainty in model selection, but in practice estimates produced by distance sampling are robust to model definition and little is gained by using model-averaged estimates (J. Lloyd, personal observation).

All of the estimates of abundance and density, whether corrected for detectability or not, are based primarily on detections of vocalizing individuals that we assume were defending territories or
advertising their presence to conspecifics of the opposite sex. For some species (e.g., Black-whiskered Vireos, White-eyed Vireos \([\text{Vireo griseus}]\), Yellow Warblers, and Prairie Warblers) most of our detections were of singing males. For these species, the estimates of abundance and density presented in the report probably pertain only to territorial males, and do not include the females paired with these males, nor do they include silent, non-territorial individuals of either sex (i.e., floaters). For other species (e.g., Gray Kingbird, Red-bellied Woodpecker \([\text{Melanerpes carolinus}]\), Great Crested Flycatcher \([\text{Myiarchus crinitus}]\), and perhaps Mangrove Cuckoo), both sexes share a vocal repertoire and so our detections may reflect the abundance of both males and females. If both sexes vocalize at equal rates (unknown for any of the species in the study), then the estimates of abundance and density probably provide an estimate of the size of the breeding population at any given survey point. In sum, the parameter estimated by the measures of abundance and density presented in this report may vary among species (e.g., number of local territorial males vs. number of local breeding individuals), and so caution is needed in comparing results across species.

Habitat associations of mangrove landbirds.

We examined habitat associations in two ways. First, we used a purely subjective approach, in which we examined the distribution of each species relative to gross differences in the environment at each point (e.g., inland points near the marsh ecotone vs. points along major tidal rivers). Using this approach, we were able to search for broad patterns in all species, not just those with sample sizes sufficient to permit a more rigorous quantification of habitat relationships. Second, for a subset of species, we attempted to quantify relationships between abundance and vegetation cover using generalized additive models. To describe vegetation cover, we began by using high-resolution aerial photos to identify homogenous patches of vegetation within the 150-m radius survey area around each point. We then visited each point and then circumnavigated each patch – either on foot or by boat – and assigned each patch to one of 8 vegetation types. We also visually estimated maximum canopy height to the nearest meter in each patch. Vegetation types included 7 forest types and one shrub type. Forest types included: red-mangrove forest (closed-canopy patches with >75% cover by red mangrove), black-mangrove forest (closed-canopy patches with >75% cover by black mangrove), white-mangrove forest (closed-canopy patches with >75% cover by white mangrove), mixed-mangrove forest (closed-canopy patches with > 2 species, neither of which covered >75% of the stand area), buttonwood woodland (open canopy woodland with >75% cover by buttonwood), black-mangrove woodland (open canopy woodland with >75% cover by black mangrove), and tropical-hardwood forest (closed-canopy forest dominated by hardwoods). We also quantified the percentage of area covered by scrub
mangrove (any pure or mixed-association of mangroves with a canopy height < 2 m). Finally, we quantified the area covered by open water and by non-vegetated surfaces, including roads, buildings, and trails.

We used generalized additive models to explore the relationship between abundance of each species within the 150-m patch, as corrected for detectability (estimated using removal models), and the percentage of the survey area covered by each of the 8 different vegetation types and open water. We also included as predictors the maximum canopy height in the survey area and the average canopy height of all vegetation patches in the survey area.

In fitting the generalized additive models, we assumed a Poisson distribution of errors, which is generally appropriate for count data. We examined plots of residuals to address the assumptions of the generalized additive models; in no case did we find significant departures from assumptions and so we made no effort to transform data or use alternative error distributions. As a smoothing term in the models, we used thin-plate regression splines with a ridge penalty added that allowed terms with little relationship to bird abundance to be smoothed to zero and thus effectively eliminated from the model. This allowed us to simplify our models and eliminate non-significant predictor variables. We used estimated P-values of terms that remained in the model to interpret the significance of any relationship between abundance and our predictor variables. All analyses were conducted in R using the package mgcv (Wood 2003).

A long-term monitoring protocol for mangrove landbirds.

Under this objective, we considered four issues of general importance in developing a population-monitoring scheme: sampling design, sample size, timing of sampling, and data analysis. We also considered a fifth issue of importance to mangrove landbirds in particular, namely how to survey for Mangrove Cuckoos efficiently. To address the issue of sampling design, we evaluated GRTS, an approach used often in the design of monitoring programs (see Methods - Establishing survey points). In the Discussion, we present our opinions on the challenges and potential benefits of using this approach. As reported in Methods - Estimating bird abundance we used two different methods, distance sampling and removal models, to analyze count data collected during surveys. Results from both methods are presented under Results - Estimating bird abundance, and in the Discussion, we provide a qualitative review of the merits of both approaches. Detailed methods for the remaining issues addressed under this objective are presented below.
**Optimal number of points to survey**

An important consideration in developing a scheme to monitor trends in abundance is sample size: how many points must be sampled, and how often must they be sampled, in order to estimate trends at the desired level of precision? To address this issue, we examined the statistical power – in this case, the probability of detecting a change in abundance over time, given that a change has occurred – of three different monitoring scenarios. In each scenario, the trend of interest was a 50% decline over a 20-year period, an arbitrary but widely accepted goal for landbird monitoring programs (Butcher et al. 1993, Bart et al. 2004). In one scenario, we examined the probability of detecting this trend, at a significance level of \( \alpha = 0.05 \), as a function of the number of points surveyed, with the assumption that each point was surveyed annually. In the second scenario, we assumed that each point was surveyed every other year, and in the final scenario we assumed that points were surveyed only every 5 years. We chose these scenarios to reflect what we felt was a reasonable range of resource investment, from a relatively labor-intensive annual survey to a survey conducted only twice a decade. For all scenarios, the starting value was set to the average abundance of Black-whiskered Vireo (0.36 birds per point), a moderately abundant mangrove specialist.

Although statistical power can often be estimated analytically, for more complicated analyses, such as those typically used to estimate population trends, there is no analytical solution for calculating power. In these cases, power can only be estimated via simulation. To do so, we began by formulating a linear model \( y_i = a + b \cdot t \), 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) that described the trend of interest; in this case, a 50% decline over 20 years. We then 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, where time was considered as a fixed effect and sampling point was analyzed as a random effect. This mirrors the approach used to estimate real trends in the 2000-2008 count data from Ten Thousand Islands National Wildlife Refuge (Lloyd and Doyle 2011). 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 $P \leq 0.05$.

**Optimal seasonal timing of surveys**

The composition of a local bird assemblage will change markedly over the course of a year. Large changes are possible even within a breeding season due to births, deaths, and movement in and out of the area. Apparent changes in the composition of local assemblages can also occur due to behavioral changes, for example the decline in singing rate that often occurs in later stages of nesting. The goal of most point-count surveys is to provide a snapshot of the number of individuals present at a location at a given time, and to do so the timing of each survey should correspond to the period when the most individuals are the most active. Surveys that are conducted too early may miss some late arrivals or inadvertently sample migrating individuals, whereas surveys that are conducted too late may count post-breeding wanderers or may fail to detect individuals busy with raising nestlings or caring for fledglings. At the same time, an overly narrow sampling window reduces the number of points that can be sampled per observer and may therefore limit overall sample size. We examined the nature of this tradeoff by plotting the total number of detections by species for each day that surveys were conducted. To aid in the visualization of general patterns in these data, we calculated and plotted one-week moving averages of the number of individuals counted.

**Use of playbacks to increase detectability of Mangrove Cuckoo.**

After analysis of data collected in 2008 and 2009 field seasons, we realized that our surveys yielded far fewer detections of Mangrove Cuckoos than we had anticipated. We suspected that this species’ cryptic behavior resulted in low detection rates during our surveys, thus leading to an exaggerated appearance of rarity. Previous unpublished research conducted at JN “Ding” Darling National Wildlife Refuge suggested that detections of Mangrove Cuckoos could be increased by broadcasting playbacks of recorded Mangrove Cuckoo vocalizations (J. Palmer, USFWS, personal communication), and so we implemented a playback experiment in 2010 and 2011 to test this hypothesis.

In 2010, we randomly selected 67 points to include in this experiment. Points were randomly assigned to either the treatment group (n=42) or the control group (n=25). For points assigned to the treatment group, we conducted a standard, 10-minute survey and then broadcast a recording of the typical vocalization (the guttural series of repeated “cah” notes) of this species that is heard during the
breeding season. The function of this vocalization is unknown, but that it is limited to the breeding season argues for a role in territorial defense and advertisement. The recording lasted for 8 seconds, at which point we then listened quietly for 52 seconds. This process was repeated 4 more times, for a total survey period of 5 minutes. During the survey period, we sat quietly in one place and noted the presence of any Mangrove Cuckoo heard or seen. At control points, the initial 10-minute point count was followed by 5 minutes of quietly listening and looking for Mangrove Cuckoos. No playbacks were used at control points.

In 2011, we used the same procedures, but selected points in a somewhat different fashion. As in 2010, we drew from existing survey points, but we narrowed the set of possible points by drawing from only those points at which ≥1 Mangrove Cuckoo had been detected during the 2008-2010 survey of mangrove landbirds. The change in methodology from 2010 to 2011 was done because a preliminary power analysis indicated that a large sample size (n>500) would be required to achieve desired statistical power given the number of points at which no cuckoos were detected. We suspected that a low response rate was partly due to the fact that many of the points that we included in 2010 were not inhabited by Mangrove Cuckoos (e.g., due to a lack of suitable habitat). Increasing the power of the experiment required eliminating, to the extent possible, points at which cuckoos were absent. We do not understand cuckoo habitat requirements sufficiently to predict where they occur based on habitat, so as an alternative we used the presence of cuckoos in previous years as a surrogate that allowed us to predict where they might occur in 2011. Of the points at which ≥1 Mangrove Cuckoo had been detected, we randomly selected 43 points to include in this study. As in 2010, we randomly allocated points to the treatment group (n=21) or the control group (n=22). Points surveyed in 2010 were not surveyed again in 2011. Thus, our total sample size was N=110. In both years, we conducted all surveys between 0600 and 1000 during May and June, putatively the peak breeding season for Mangrove Cuckoos in Florida (Hughes 1997). All survey locations were accessed by boat, usually a kayak but sometimes a motor boat.

We compared the frequency of responses between treatment and control points using a generalized linear model. We assumed a binomial distribution of errors and used the logit link function. In addition to the main predictor of interest - a dummy variable indicating whether a playback was used at the point – we also included several covariates that we thought might independently influence the probability of detecting a Mangrove Cuckoo. First, we included the date on which each point was visited, because we suspected that the propensity of Mangrove Cuckoos to vocalize spontaneously or to respond to broadcasts of recorded vocalizations might vary seasonally as a function
of what, if any, breeding activities were underway. Second, because the sample of points included in this experiment was drawn without knowledge of whether Mangrove Cuckoos were present, we also included a dummy variable that indicated whether any Mangrove Cuckoos had been detected that year at the point during surveys conducted as part of the separate study of mangrove landbirds. We assumed that a positive response to either the control or treatment was more likely at points at which individuals had been detected during the current year. Similarly, we also assumed that the presence of Mangrove Cuckoos in previous years was also likely a positive indicator of garnering a response during the experiment, and so we also included a dummy variable indicating whether any Mangrove Cuckoos had been detected at the point in surveys conducted during previous years (because of how we chose points in 2011, all points surveyed in that year had the same value for the dummy variable). All analyses were conducted using R, version 2.12.2.

RESULTS

Abundance and distribution of mangrove landbirds.

We detected a total of 80 species during our surveys (Appendix A), including 31 landbird species that breed in south Florida and 8 landbirds that winter in south Florida but breed further north or that pass through Florida during spring migration. All of the 31 landbird species that we detected that breed in south Florida probably nest at least occasionally in the mangroves, with the possible exception of Boat-tailed Grackle, which we detected in mangroves only once. We were able to generate estimates of detectability for species with \( \geq 90 \) detections. The numbers of detections for all species appear in Appendix A.

The landbird assemblage was dominated numerically by species that are abundant in a variety of habitats across south Florida (Table 1). Of the 4 most abundant and widespread species, only the Prairie Warbler is restricted to mangrove forests. A second group – Red-winged Blackbird (\textit{Agelaius phoeniceus}), Great Crested Flycatcher, Pileated Woodpecker (\textit{Dryocopus pileatus}), American Crow (\textit{Corvus brachyrhynchos}), Black-whiskered Vireo, and White-crowned Pigeon – consisted of species that were either widespread but somewhat less abundant, such as Great Crested Flycatcher, or locally common but narrowly distributed, such as White-crowned Pigeon (Table 1). The remaining species in Table 1 were detected in much lower numbers, although, as with the preceding group, in some cases the apparently low number of detections reflected a narrow distribution within the study area rather than overall scarcity (e.g., Yellow Warbler).
Uncorrected counts were strongly correlated with abundance estimates adjusted for detectability using removal models ($r = 0.99$, 95% CI = 0.96 -0.99, $P < 0.001$; Table 1). We had few problems fitting removal models to our data. Most of the detections were recorded during the first 3 minutes of each survey, and 75% were recorded within the first 5 minutes; the only notable outlier from this relationship was Mangrove Cuckoo (Fig. 6). The accumulation rate for Mangrove Cuckoo detections fell outside of the 95% confidence interval for all other species, indicating a relatively low rate of detectability. Distance sampling generated very different results than those produced by the removal models (Table 2). Model fit was adequate in 10 of the 16 species considered (Table 2). We found no commonalities among the species in which model fit to the data was poor; some showed evidence of detections clustered close to the observer, whereas some showed far more detections at greater distances than expected under the model. Overall, detectability estimates appeared unreasonably low. For example, estimated detectability for White-eyed Vireo and Prairie Warbler seemed low given that we observed both species vocalizing loudly and frequently. As a result, we suspect that abundance estimates generated by distance sampling were biased high, and so we based our interpretations on the estimates produced by removal models. We note, however, that we lack any independent measure of abundance against which to compare the different approaches for adjusting raw counts for detectability, and so we cannot rule out the possibility that estimates produced by distance sampling are in fact more accurate than those based on removal models.

Overall patterns of abundance are, in some cases, confounded with patterns of distribution. For species with a limited distribution in the study area, average abundance when present (Table 3) is much greater than abundance averaged across all points. We identified three main patterns of distribution: species that were widespread and found at points throughout the study area, species with a southerly distribution that were restricted mostly to the Florida Keys and islands of Florida Bay, and species with a northerly distribution that were restricted mostly to the mainland mangrove forests and 10,000 Islands region. Widespread species included Red-bellied Woodpeckers, Northern Cardinal (*Cardinalis cardinalis*), Prairie Warbler, Black-whiskered Vireo, Great Crested Flycatcher, and Mangrove Cuckoo (Figs. 7-12). With the exception of Mangrove Cuckoo, which was found at very low abundance, all of these species were abundant in addition to being widespread. In general, each of these species was found at roughly similar levels of abundance in all of the areas sampled (Table 4). Except for Mangrove Cuckoo, all of these species were abundant when present (Table 3).

Species with a more southerly distribution within the study area included Red-winged Blackbird, White-crowned Pigeon, Gray Kingbird, Yellow Warbler, Common Yellowthroat (*Geothlypis trichas*), and
Common Grackle (*Quiscalus quiscula*) ([Figs. 13-18](#)). Some of these species – notably, White-crowned Pigeon, Gray Kingbird, and Yellow Warbler – have global distributions centered on the West Indies, so we assume that their prevalence in the southern portion of the study area reflects a combination of proximity to recent colonization sources (e.g., Yellow Warblers) and the distribution of suitable habitat conditions within the study area. Other members of this group, including Red-winged Blackbird, Common Yellowthroat, and Common Grackle, are more temperate in their geographic affiliation, and their greater abundance at southerly points likely reflects habitat preferences. In particular, open, marshy vegetation, which all of these species prefer in other parts of their range, is widespread in the Keys and on many of the islands in Florida Bay. Support for this hypothesis was also evident in the occurrence of these species at inland points near the transition from mangrove to freshwater marsh (e.g., Common Yellowthroat, [Fig. 17](#)). Three species in this group – Red-winged Blackbird, Yellow Warbler, and White-crowned Pigeon – were quite abundant when present ([Table 3](#)).

Four species had distributions centered on the northern portion of the study area ([Figs. 19-22](#)): White-eyed Vireo, American Crow, Carolina Wren (*Thryothorus ludovicianus*), and Pileated Woodpecker. Each of these species is distributed widely in other habitats across south Florida, and probably only used mangroves under certain conditions (e.g., Pileated Woodpeckers only occur in those forests where large-diameter trees are widespread). Both Pileated Woodpecker and American Crow were fairly common when present; Carolina Wren tended to occur at relatively low abundance wherever it occurred ([Table 3](#)).

Mourning Doves exhibited a unique distributional pattern, reaching their greatest abundance in the far south of the study area and the far north of their study area. Abundant in the Keys, they were mostly absent from Biscayne National Park and Everglades National Park, reappearing again at relatively high abundance in the Ten Thousand Islands region ([Table 4; Fig. 23](#)).

Four species were detected that did not fit into any of these broad distributional groupings, primarily because they were detected very infrequently. This group included Eastern Towhee (*Pipilo erythrophthalmus*), Blue Jay (*Cyanocitta cristata*), Northern Flicker (*Colaptes auratus*), and Downy Woodpecker (*Picoides pubescens*) ([Appendix A](#)). Blue Jay, Northern Flicker, and Downy Woodpecker are probably only occasional residents or casual visitors to mangrove forests. Eastern Towhees present a more interesting case, as they were consistently found at points in Hell’s Bay, suggesting the existence of a small, perhaps remnant, population in the mangroves. Robertson and Kushlan (1974), citing a communication from J. B Edscorn, mention that Eastern Towhees may have “sparse breeding populations” in the mangroves, perhaps alluding to the population of birds at Hell’s Bay.
Habitat associations of mangrove landbirds.

Red-mangrove forest was the dominant vegetation type at 50.3% of points, and averaged 50.3% cover (95% CI = 45.2% – 55.4%) (see Appendix B for photos). Mixed-mangrove forest was the dominant cover type at 28.4% of points, with an average cover of 30.2% (95% CI = 25.7% – 34.7%). None of the other cover types was dominant at >5% of points, and none had an average cover of >5%. Open water covered 0 – 70% of the survey area at points. The average percent cover by water was 17.8% (95% CI = 16.2% – 19.5%). The distribution of cover types was not even across the study area, although red-mangrove forest was dominant at some points in every region. Mixed-mangrove forest dominated at least some points in all regions except in the Keys. Black-mangrove forest was dominant only at points on islands in Florida Bay. Tropical hardwoods were a significant component of vegetation cover only at sites in the interior of Everglades National Park, including the Canepatch area, Lane’s Bay, and Hell’s Bay. Extensive buttonwood and black-mangrove woodlands were found only near the Snake Bight and Rowdy Bend trails. Scrub mangrove was limited primarily to the Keys. Canopy height was greatest in the forest along the major rivers of the southwest coast, averaging 11.3 m (95% CI = 10.8 m – 11.8 m). Intermediate heights were observed on the islands of Florida Bay (9.4 m, 95% CI = 9.0 m – 9.8 m) and at Ten Thousand Islands (9.0 m, 95% CI = 8.7 m – 9.4 m). Lower values were observed at interior sites in Everglades National Park (8.1 m, 95% CI = 7.5 m – 8.6 m), at Biscayne National Park (7.7 m, 95% CI = 6.9 m – 8.5 m), and finally in the Keys, where the lowest average canopy heights were observed (6.6 m, 95% CI = 5.8 m – 7.0 m).

We focused our quantitative modeling efforts on 5 mangrove specialists: White-crowned Pigeon, Gray Kingbird, Black-whiskered Vireo, Prairie Warbler, and Yellow Warbler. We did not attempt to model habitat relationships for Mangrove Cuckoo because we were concerned that the low detectability of this species may have resulted in a large number of false absences. False absences can produce biased estimates of the association between abundance and vegetation cover.

Abundance of White-crowned Pigeons was significantly associated with the extent of red-mangrove forest (P<0.001), black-mangrove forest (P=0.01), buttonwood (P=0.004), maximum canopy height (P<0.001), and scrub mangrove (P<0.001) (Fig. 24). The model that included only these terms explained 27% of the variation in counts of White-crowned Pigeons. Variation in Gray Kingbird abundance was significantly associated with the extent of black-mangrove forest (P<0.001), maximum canopy height (P<0.001), scrub mangrove (P<0.001) (Fig. 25). The model that included only these terms explained 19% of the variation in the number of Gray Kingbirds counted at a point. Black-whiskered Vireo abundance was related to the extent of black-mangrove forest (P<0.001), scrub mangrove
(P<0.001), and tropical hardwoods (P=0.004) (Fig. 26). The reduced model explained 13.5% of the observed variation in counts of Black-whiskered Vireos. Yellow Warbler abundance was related to only one variable: the extent of black-mangrove forest (P<0.001) (Fig. 27). The model with this single term explained 76% of the observed variation in Yellow Warbler counts. Prairie Warbler abundance was associated with variation in the extent of black-mangrove forest (P < 0.001), tropical hardwoods (P< 0.001), and maximum canopy height (P<0.001) (Fig. 28). The model with these terms explained 28% of the variation in counts of Prairie Warblers.

For three of the species in this analysis – White-crowned Pigeon, Gray Kingbird, and Yellow Warbler – the habitat relationships described above reflect in part their narrow distribution. That is, features more common at the northern end of the study area (e.g., tall canopied stands of red mangrove) may appear to be selected against simply because those features are not found in the Keys or on the islands of Florida Bay. To address this issue, we also conducted the analyses considering only those points within the Keys and Florida Bay. White-crowned Pigeon abundance was related to red-mangrove forest (P<0.001) and black-mangrove forest (P<0.001), but the nature of the relationships differed from those described in the analysis of all points in the study area (Fig. 29). The model applied to the reduced data set explained a similar amount of variance in White-crowned Pigeon abundance (29%). We could not estimate the relationship between White-crowned Pigeons and canopy height or extent of scrub mangrove due to the small sample size of the restricted data set. For Gray Kingbirds, none of the predictor variables were significant when only the reduced data set was analyzed – all terms were smoothed to zero, producing a null model. For Yellow Warblers, the analysis of the reduced data set produced the same result – abundance increased as a non-linear function of the extent of black-mangrove forest – as generated from the analysis of the complete data set.

A long-term monitoring protocol for mangrove landbirds.

Optimal number of points to survey

For a moderately abundant species (Black-whiskered Vireos, 0.36 birds per point), surveying points every 5 years was generally more efficient than annual or biennial sampling (Fig. 30). Achieving 80% power to detect a 50% decline over 20 years, at α = 0.05, required sampling approximately 50 points annually, 100 points biennially, or 150 points every 5 years. Over a 5-year period, annual and biennial sampling at this level required 1.7 times as much effort as quinquennial sampling.
Optimal seasonal timing of surveys

The six species of landbirds limited primarily to mangroves showed similar detection rates over the course of our surveys (Fig. 31). Detections were low in late April, rose through May and early June, and began declining in late June. One-week moving averages for White-crowned Pigeons and Yellow Warblers were flat across the season. Black-whiskered Vireos exhibited 3 peaks in detection rate, with two smaller peaks occurring in early May and mid-June, and one larger amplitude peak occurring in late May. The moving average for detection rate of Prairie Warblers was flat throughout much of May, and declined slowly through June. Moving averages of detection rates for Mangrove Cuckoos and Gray Kingbirds were highest in mid- to late May, with Gray Kingbirds showing a slightly earlier peak than Mangrove Cuckoos, and then declined modestly throughout the remainder of the survey season.

Use of playbacks to increase detectability of Mangrove Cuckoo

We conducted 42 playback and 21 control experiments in 2010 and 25 playback and 22 control experiments in 2011. We surveyed control points between 9 May and 23 June in 2010 and 7 May and June 15 in 2011. We surveyed experimental points between 9 May and 26 June in 2010 and 7 May and 22 June in 2011. The median date on which control points were surveyed was 27 May and 16 May in 2010 and 2011, respectively. The median date on which experimental points were surveyed was 9 June and 23 May in 2010 and 2011, respectively. We detected Mangrove Cuckoos at 14 of the 67 treatment points (20.9%) and at 1 of the 44 control points (2.3%). This difference was highly significant: using a playback (P=0.01) increased the likelihood of detecting a Mangrove Cuckoo. We never recorded a response by >1 bird at any point; males and females are not distinguishable by sight, so we do not know the sex of birds responding to the playback. The average time of response was 131 s (range: 20 s – 267 s). All of the birds detected during the experiment were first detected within 50 m of the observer. Of the other predictors, only the presence of Mangrove Cuckoos at the point in past years was significantly related to the probability of detecting a Mangrove Cuckoo (P = 0.05). Detections of cuckoos during other surveys conducted in the same year was unrelated to the probability of detecting a cuckoo during the experiment (P = 0.5) as was the date on which the experiment was conducted (P = 0.74).
DISCUSSION

Abundance and distribution of mangrove landbirds.

Patterns of abundance of mangrove landbirds.

The assemblage of mangrove landbirds was dominated numerically by a group of 4 species that are both abundant (>1 individual per point) and widespread in their distribution: Red-bellied Woodpecker, Northern Cardinal, Prairie Warbler, and White-eyed Vireo. Of this group, only the Prairie Warbler is endemic to mangroves; the remaining 3 species are common in a variety of habitats throughout the eastern United States. A second group was composed of species found at slightly lower average abundances (c.a. 0.4-0.8 individuals/point). This group included Great Crested Flycatcher, Pileated Woodpecker, American Crow, and Black-whiskered Vireo, which were widespread and moderately abundant throughout the study area, and Red-winged Blackbird and White-crowned Pigeon, which were more localized in their distribution but very abundant where present. Of the species in this group, only Black-whiskered Vireo and White-crowned Pigeon depend on mangroves for habitat. The remaining species in the mangrove landbird assemblage were relatively uncommon (<0.2 individuals/point). With the exception of Yellow Warblers and Gray Kingbirds, which had very limited ranges but were abundant when present, the species in this group were never found in great numbers. Some were probably only occasional or transient inhabitants of the mangroves, such as Blue Jays and Downy Woodpeckers. Others were more closely associated with adjacent habitat types, such as Carolina Wrens, which were generally found further inland at the transition from mangroves to freshwater marsh.

The most enigmatic of the uncommon members of the assemblage was the Mangrove Cuckoo, which, despite being dependent on mangroves for habitat, was both local and inconsistent in its distribution (i.e., it was rarely counted at the same point in different years) and present only in low numbers where it occurred. Passive surveys undercounted the true number of this species, and our effort to account for detectability probably did not address undercounting because the removal models (or, for that matter, distance sampling) do not account for birds present but unavailable for detection (i.e., individuals that did not vocalize during the count). This suggests that the estimates of abundance presented in this report are substantial underestimates of the true abundance of Mangrove Cuckoos.

Patterns of distribution of mangrove landbirds.

The relatively narrow strip of Florida’s southern coastline covered by mangroves marks both a northward and southward front: several temperate species reach their southern range limit in the
mangroves of peninsular Florida, and several West Indian species reach their northern range limit in the Keys and islands of Florida Bay. The former group includes Carolina Wren, White-eyed Vireo, Eastern Towhee, Pileated Woodpecker, and American Crow, and in the latter are Yellow Warbler and White-crowned Pigeon (we detected Gray Kingbird only in the Keys, but it is also known to occur as far north as the Carolinas). Mourning Doves may represent both situations. Birds in the Keys may belong to northernmost populations of *Z. m. macroura* and individuals in the Ten Thousands Islands region may belong to the widespread temperate subspecies *Z. m. carolinensis* (Aldrich and Duvall 1958). In the case of Yellow Warbler, and perhaps Mourning Doves, the observed distribution probably reflects the recent history of colonization from the south. There are no obvious habitat factors limiting the northward spread of Yellow Warblers, and so over time we might expect to see Yellow Warblers breeding beyond Florida Bay. Species like Pileated Woodpecker and White-crowned Pigeon are probably limited by the distribution of key habitat features. In the case of Pileated Woodpeckers, the presence of large snags in which to nest likely acts as a limiting factor. White-crowned Pigeons, on the other hand, are likely limited to Florida Bay and the Keys by the joint distribution of predator-free islands and poisonwood (*Metopium toxiferum*), a key food species. Why the remaining species appear to hit a north or south range limit in the mangroves is unclear, but range limits might generally reflect the gradual retreat of temperate species and the slow expansion of tropical species.

Red-bellied Woodpecker, Northern Cardinal, and Prairie Warbler were essentially ubiquitous and not obviously limited by habitat or larger biogeographic processes (e.g., ongoing range expansion). Members of this group were detected in all regions of the study area and at >75% of all survey points, and were typically found at relatively high density. No obvious distributional breaks were apparent for Great Crested Flycatcher, Black-whiskered Vireo, Red-shouldered Hawk, and Mangrove Cuckoo, either, although species in this group were found at somewhat lower densities and at far fewer points. Given the difficulty of detecting Mangrove Cuckoos, we are hesitant to draw conclusions about the distribution of this species in the mangroves, as continued surveys would likely reveal a very different pattern of occupancy. Finally, both Common Yellowthroat and Red-winged Blackbird were found throughout the study area but at only a small percentage of survey points. Distributions of both species overlapped extensively, and were likely driven by their shared preference for shrubby marshes.

**Habitat associations of mangrove landbirds.**

In previous reports, we have noted that existing, spatially explicit data were inadequate for quantifying habitat associations of mangrove landbirds. For example, the Florida Land Cover data classified most of our points into a single class, mixed-mangrove forest. Not only does this preclude any
meaningful analysis of the relationship between bird abundance and vegetation type, but it also appears to be incorrect at the scale of our measurements; in direct field sampling, we found that most of our points were dominated by red-mangrove forest. This does not indicate that the Florida Land Cover data are erroneous, but rather highlights the problem of downscaling remotely sensed data. In addition to these methodological challenges, use of the land-cover types to examine habitat associations may be undesirable for other reasons. For example, the Florida Land Cover database did not distinguish among some mangrove forests that are substantially different when viewed from the ground. The database classifies some of the short red mangrove that fringe the Florida Bay islands as dwarf mangrove, presumably because of their short stature, even though these narrow patches of fringing mangrove are quite different ecologically from the extensive stands of stunted red mangrove that occur further inland and that are more closely aligned with conventional definitions of the dwarf mangrove forest type (e.g., Lugo and Snedaker 1974).

Given these problems, we conducted field sampling of vegetation conditions in 2011 to identify bird-vegetation relationships. The advantage of this approach was that it allowed us to address some of the concerns about the accuracy of the Florida Land Cover data for the scale at which our study was conducted. The disadvantage of developing models based on field-sampled vegetation cover was that it did not allow us to predict abundance of mangrove landbirds at points that we did not sample because we have no information on vegetation cover outside of our sample points. These concerns were unwarranted because the resulting models were relatively uninformative. The models were of little use in elucidating relationships between birds and vegetation structure because the results were unduly influenced by the distribution of uncommon cover types and because the results were scale-dependent and tended to reflect broad differences in vegetation structure among different regions of the study area.

As an example of the effect of uncommon vegetation types on the relationships, two species, Black-whiskered Vireo and Prairie Warbler, showed negative relationships with tropical hardwoods. However, this relationship almost certainly reflected a quirk in our data set, whereby the only locations with substantial cover by hardwoods were inland sites near the marsh ecotone in the Canepatch area of Everglades National Park. It is likely that Black-whiskered Vireo and Prairie Warbler avoid these inland sites for other reasons (e.g., extensive areas of open marsh offer few nesting and foraging opportunities), not because they avoid tropical hardwoods per se. Indeed, Black-whiskered Vireos are relatively abundant in hardwood hammocks in the Keys. As such, we suspect that the relationship we have described is spurious both for Black-whiskered Vireo and Prairie Warbler.
The problem of potential confounding between large-scale differences in vegetation cover and geographic range limits of birds was more widespread. For example, White-crowned Pigeon, Gray Kingbird, and Yellow Warbler all showed positive relationships with three habitat variables that exhibited a distinct latitudinal gradient: black-mangrove forest, canopy height, and extent of scrub mangrove. Black-mangrove forest was most extensive on the islands of Florida Bay, scrub mangrove was most extensive in the Keys, and canopy height was lowest in the Keys. White-crowned Pigeon, Gray Kingbird, and Yellow Warbler all have distributions centered on Florida Bay and the Keys, raising the question of whether the relationships with the habitat variables were simply a reflection of the geographic distribution of the birds. In two cases, the relationships derived from an analysis of the complete data set were quite different from those derived from the analysis of points within the Keys and Florida Bay (that is, excluding points beyond the current range limits of the three bird species). For White-crowned Pigeon, the relationship with black-mangrove forest switched sign, from positive at the larger scale to negative when considering only those points within their range. White-crowned Pigeons showed a positive relationship with buttonwood when considering all points within the study area, and no relationship when the analysis was limited to the Keys and Florida Bay. The relationship with red-mangrove forest was consistent regardless of the scale of analysis. For Gray Kingbird, none of the variables were significant predictors of abundance when the analysis was limited to the Keys and islands of Florida Bay, whereas this species showed significant relationships with black-mangrove forest, canopy height, and scrub mangrove when modeling was applied to points from across the study area. The habitat models for White-crowned Pigeons and Gray Kingbirds that were based on all of the points in the study area are therefore unreliable, and probably reflected latitudinal differences in vegetation structure that corresponded with the distribution of both species. Only Yellow Warblers showed a consistent relationship regardless of scale, suggesting that this species does seek out black-mangrove forests.

Given the difficulties that we had in creating habitat models with vegetation cover as a predictor, we suggest that future investigations focus on more intensive descriptions of habitat use, for example by delineating individual home ranges and identifying those features of vegetation structure and composition that distinguish home ranges from unoccupied areas. In the meantime, our qualitative descriptions of distribution may prove useful in identifying broad patterns and generating hypotheses about habitat use among mangrove landbirds. In Appendix C, we provide a brief description of our findings for each mangrove landbird.
A long-term monitoring protocol for mangrove landbirds.

We considered four general elements important in designing a long-term monitoring protocol: sample design, or how survey points are identified; sample size; timing of sampling; and analysis of data, in particular how to account for imperfect detection of birds. We also addressed a fifth issue specific to this study, namely whether detection rates of Mangrove Cuckoos could be increased through the use of recorded vocalizations.

Designing a sampling network

We found that a GRTS sampling design has several useful features. This approach generates a spatially balanced sampling network while maintaining a non-zero probability of inclusion for every possible point, thus allowing for straightforward estimates of variance while avoiding the potential problems of purely random sampling designs (e.g., spatial clustering). This method also allows for additional points to be incorporated at a later date, which may be useful if the scope or intensity of monitoring changes. However, implementing GRTS was not without its difficulties. The most obvious shortcoming is that the GRTS sample included many points that did not meet our accessibility rule. This required drawing a much larger GRTS sample than needed (1,000 points, when only approximately 10% could be used) and manually discarding all those that were not accessible. This problem could be overcome by better defining the sampling frame within the GIS so that the grid of possible points only included accessible areas. However, doing so would require spatially explicit information that is currently unavailable. For example, we had no way to limit the sampling frame to navigable creeks <50 m wide, as existing data do not identify the width of water features. Thus, many of the points drawn in the GRTS sample were along large watercourses, which we did not wish to sample or along creeks so small that they could not be accessed by kayak. Conversely, many navigable creeks were not identified in existing spatial data sets, and thus would not have been included in the sampling frame if, for example, we had delineated the sampling frame based on creeks identified in available GIS coverages. In sum, for the GRTS approach to be efficient, the sampling frame must be thoroughly and carefully defined; if all possible limitations on the sampling frame cannot be defined a priori, then extensive post-processing screening of the GRTS sample will be required.

In defining the appropriate sampling frame, long-term monitoring programs, such as that being piloted in this study, need to balance the importance of statistical rigor with the realities of data collection. In the case of mangrove forests, the most significant problem is that many areas are simply not accessible except by extraordinary means. For example, we found that accessing a single point on some of the small islands in the lower Keys required 3-4 hours, which precludes doing more than a
single survey in a morning. Thus, although it is possible to sample randomly from all mangrove forests in south Florida, doing so comes at the expense of sample size. As such, assuming that resources are not unlimited, there is a trade-off between the extent of the sampling frame – that is, the area about which we wish to draw inference – and the power to detect trends in population size or to describe habitat associations. Before any long-term monitoring program is adopted, potential users of the data should agree on a suitable sampling frame. In other words, is it sufficient to sample accessible areas, or must we take extraordinary measures to reach inaccessible locations away from creeks or trails? To help answer this question, it may be useful to invest in additional research to determine the extent to which counts conducted along accessible features produce biased estimates of abundance and distribution. Concerns about the nature of the sampling frame might be mitigated if it was determined that counts from points along creeks or trails produced similar estimates of abundance as counts conducted at points in the mangrove interior.

**Optimal number of points to survey**

Of the various monitoring scenarios that we considered, the most efficient approach was to sample every 5 years. We estimated that sampling 150 points every 5 years would yield approximately 80% power to detect a 50% decline over 20 years at $\alpha = 0.05$; sampling 200 points every 5 years would yield approximately 90% power to detect this trend. In contrast, annual and biennial sampling required approximately 1.7 times as much effort. Under either annual or biennial sampling, achieving the desired power required conducting approximately 250 points over a 5-year period. This is fewer points than currently surveyed, which suggests that a subset of the points that we have established would make a suitable sampling frame for a long-term monitoring effort.

However, the power analysis assumes that every point has a non-zero probability of yielding at least 1 individual of the focal species. For many of the species in this study, a random sample of 150 points – or worse, 50 points – would likely include a non-negligible number of points where the species never occurred. This scenario would result in a realized power much lower than predicted by the power analysis. The results of the power analysis should therefore be viewed as the minimum number needed within a properly defined sampling frame. A poorly designed sampling frame will require many more points than predicted here.

Our power analysis was based on a trend that started with an average abundance equal to that observed for Black-whiskered Vireos from 2008-2011. As such, a monitoring program based around the findings of this analysis should be suitable for detecting trends in most of the mangrove-dependent species in Florida, assuming that the sampling frame is appropriately described. Trends in abundance of
Mangrove Cuckoos, however, would almost certainly not be estimated with sufficient precision under a monitoring program designed around more commonly encountered species. A monitoring program capable of detecting trends in this species, given its rarity and spotty distribution, would require either adopting methods to increase detectability or surveying a large number of points and perhaps adopting a more refined sampling frame so as to avoid sampling areas of non-habitat. Until we have a better understanding of the habitat needs of this species, we would not recommend designing a range-wide monitoring program around Mangrove Cuckoos. Instead, more intensive monitoring in the few areas where it appears to be more common, such as the Ten Thousand Islands area or Biscayne National Park, might provide some early warning of population declines. For example, Lloyd and Doyle (2011) were able to document significant declines in Mangrove Cuckoos over an 8-year period at Ten Thousand Islands National Wildlife Refuge based on surveys conducted at 101 points. Using call-playback surveys across a broader area might also be useful, as long as doing so is compatible with generating estimates of abundance for other focal species.

Optimal seasonal timing of surveys

In designing this study, we relied on anecdotal information provided by Terry Doyle (USFWS) to define the appropriate time of year in which to conduct surveys. In his experience, surveys conducted between 1 May and 15 June were most useful because singing rates were high, wintering birds and most passage migrants had departed, and migratory breeders like Black-whiskered Vireos had returned. Our results support this conclusion. For some species, detections were relatively consistent over the course of the season; for others, detections tended to peak in mid-May to early June. As such, surveys conducted from 1 May to 15 June would encompass the apparent peak of singing activity in the mangroves. If a longer survey season were required, extending the season later into June would be preferable to starting earlier in April, as counts were generally low in late April but remained relatively high into the end of June.

Analyzing survey data

Identifying the best sampling methodology is difficult because much of our assessment is, by necessity, based on theoretical considerations or our own opinion. We recommend continued use of removal models (or similar methods) and do not recommend that future survey and monitoring efforts take the time to estimate the distance to each detection. Removal models did not require problematic assumptions about the spatial distribution of individuals relative to the observer (as is the case with distance sampling) and generated estimates of detectability that in our opinion were reasonable,
although much of our confidence in the results of the removal models is based largely on their close
correlation with uncorrected estimates of abundance. We recognize the circular nature of this
argument (i.e., that a method producing estimates close to uncorrected values is preferable).
Nonetheless, it reflects our general concern that existing methods for estimating detectability are
burdened with assumptions that are difficult to test and, if not met, may result in biased estimates.

A ten-minute count period worked well with the removal models. We found, averaging across
species, that nearly 75% of detections were recorded in the first 5 minutes of a survey. This suggests
that the duration of the survey period could be reduced from 10 minutes to 5 minutes with a
proportionally smaller loss of sample size. However, the benefit of doing so is likely small because travel
time accounts for the majority of field time. Most locations in the mangroves are difficult to access and
doing so requires substantial effort and time. On average, the travel time between subsequent points is
10 minutes, so that 20 minutes elapses from the start of one survey to the beginning of the next. Given
a 2-hour survey period, reducing the survey duration to 5 minutes would allow 2 extra points to be
surveyed; in other words, a halving of survey time would yield a 25% increase in the number of points
that could be sampled. In many cases, the gain in number of points sampled would be null; for example,
travel time between points on islands in Florida Bay is as long as 40 minutes, and so any reduction in
survey duration is unlikely to allow any more points to be visited on a given day. Unless points are
closely spaced and easy to access, we suggest maintaining a ten-minute count period.

In previous reports, we have used distance sampling to estimate abundance and correct for
imperfect detectability of birds during our surveys. Although the analysis of distance data is relatively
straightforward, and the method is well supported in certain situations, it relies heavily on the
assumption that individual birds are distributed at random in regards to the observer. This assumption
can be met by placing survey points randomly. In this survey, points were not placed randomly, but
instead were placed along features accessible by boat or foot, primarily tidal creeks and hiking trails. If
vegetation structure changes along the edge of creeks or trails, which it almost certainly does, then birds
may not be distributed at random in regards to the observer and estimates produced by distance
sampling will be biased. Similarly, birds will not be distributed at random in regards to the observer
when large areas of open water are present in the survey area, because individuals that are detected
will generally be recorded when they are singing from a perch within the mangroves. Distance sampling
is therefore poorly suited to our study area on theoretical grounds. From a purely subjective standpoint,
the estimates of detectability generated by distance sampling seemed too low, and the resulting
estimates of density too high. Of course, we have no way of knowing the actual detectability of
different species in the mangrove, but we have difficulty believing that a relatively loud and vocal species such as White-eyed Vireo would be detected only 19% of the time that it sings, or that Northern Cardinals and Red-bellied Woodpeckers were detected less than half of the time that they were present and singing.

Determining how best to address the problem of imperfect detectability presents a significant challenge. As pointed out by Johnson (2008), every method used to correct for variation in detectability relies on a number of assumptions, few of which can be evaluated. As a consequence, estimates of abundance that have been corrected for detectability are not necessarily closer to the true value of abundance than are uncorrected counts; indeed, in some cases, corrected counts may be more biased than uncorrected counts (Efford and Dawson 2009). Unfortunately, in extensive studies such as ours, we rarely know the true value of abundance, and thus have no way to evaluate whether our efforts to correct for detectability are improving our estimates. Nonetheless, we suggest that future surveys employ removal models, or some similar approach, and avoid distance sampling unless a strong argument can be made to counter the associated theoretical problems. We also recommend that future surveys report unadjusted counts, which will allow comparisons among studies using different methods and, ultimately, may be the best use of the data (Grant et al. 2010).

Use of playbacks to increase detectability of Mangrove Cuckoos

The final issue regarding sampling methodology concerns the low detectability of Mangrove Cuckoos. As an apparently rare and declining (Lloyd and Doyle 2011) species, survey and monitoring efforts need to adequately describe the abundance and distribution of Mangrove Cuckoos. Mangrove Cuckoos are easy to detect when they vocalize – they are relatively loud and their calls are unmistakable – but they seem to vocalize infrequently. When they do not vocalize they are difficult to detect because, like most birds in the mangroves, they spend most of their time within the dense vegetation and so are seen only rarely. It is unlikely that correcting for detectability using removal models – or any other approach, for that matter – addresses the undercounting, because the detectability correction does not account for individuals that are present but not available for detection. Our experiment with the use of recorded playbacks indicated that broadcasting vocalizations produced a substantial gain in detection rate, presumably because the recordings stimulated individuals present, but otherwise undetectable, to vocalize in response. Assuming that our control and treatment points contained an equal number of individuals, which we attempted to ensure by randomizing application of the treatment, playbacks increased detection rate by ten-fold. As such, any future survey or monitoring effort should incorporate playbacks because doing so will produce more accurate estimates of abundance and distribution. One
risk of doing so is that the use of a recorded playback may also change vocalization patterns in other species, which would require that all future results be adjusted for detectability in order to allow comparisons with our data set. We do not know whether other species increase or decrease their rate of vocalization when exposed to playbacks of Mangrove Cuckoos, but it would be worthwhile to evaluate this possibility before adopting playbacks into any future, multi-species survey and monitoring effort.

Summary

Although we have identified several technical challenges in implementing an efficient monitoring program for mangrove birds, the greatest obstacle may be the lack of institutional sponsors and associated funding. A comprehensive, dedicated program for monitoring mangrove landbirds will require personnel to maintain and update a database, conduct surveys, analyze data, and prepare reports and publications. A coordinator would likely be necessary to ensure data quality and provide outreach so that potential users are aware of the data and able to access it. Ultimately, implementing a long-term monitoring program for mangrove landbirds will require a consistent source of funding, whether that funding comes in the form of administration approval for agency biologists to conduct the work or more directly in the form of grants or contracts.

At least two options for keeping tabs on populations of mangrove landbirds exist even if a dedicated, range-wide monitoring program cannot be implemented in the short-term. First, individual administrative units (e.g., Ten Thousand Island National Wildlife Refuge or Biscayne National Park) with the interest and resources can pursue monitoring on a smaller scale. As evidence by Lloyd and Doyle (2011), even local and small-scale monitoring programs can be useful in providing early warning of problems that may eventually reach populations throughout the mangroves. The advantage of this approach is that it allows some level of monitoring even if a range-wide program proves unviable and allows for more local control over the objectives of the effort (e.g., monitoring can be tailored to examine the response of populations to local management actions). The potential disadvantages are that it can lead to data that are not comparable between different sites and can result in many small, independent efforts, none of which have sufficient power to detect biologically important trends range-wide. If individual agencies or organizations pursue small-scale monitoring efforts, we encourage them at the very least to adopt a unified approach to conducting surveys. Based on our findings, we recommend using counts of at least 6 minutes duration, with detections tallied into 1-minute intervals. Recording detections for 6 minutes yields 75% of the detections gleaned from 10-minute counts at 60% of the effort, an important consideration if time is limited. However, any gain in efficiency achieved by
reducing survey duration may be entirely negligible when travel time to and among points is large. Intervals can be merged during analysis, but retaining 1-minute intervals provides maximum flexibility.

A second approach is to use existing monitoring tools to the fullest extent possible to address the need for long-term monitoring. The Breeding Bird Survey (BBS), a roadside survey of North American avifauna managed by the U.S. Geological Survey (USGS), has been a valuable monitoring tool for many breeding birds in Florida. Each 24.5 mile randomly placed BBS route consists of 50 3-minute counts, each 0.5 mile apart. Surveys are conducted by volunteers once a year during the breeding season (1 May-15 June in Florida), and birds seen or heard within the 3 minutes at each point count location are totaled on a data sheet. The BBS is an unlimited distance point count survey. Detailed information on survey methods, observer training, and data submission and use can be found at http://www.pwrc.usgs.gov/bbs/. Florida currently has 92 BBS routes. Of these, 20 are nonrandom routes (‘900” series) established to monitor birds on public lands. The USGS does not include data from 900 series routes in its annual trend analysis; however, data are managed and available for other users. Nonrandom routes can be established by providing the USGS stop descriptions, stop coordinates, a route map, and assurance that the route will be consistently run. Nonrandom routes can consist of disjunct sections and include portions conducted by boat (personal communication, Keith L. Pardieck, BBS National Coordinator). Once the route is established, the Florida BBS coordinator can assign an observer.

Roadside surveys of avian populations may not adequately sample some habitats (Betts et al. 2007) and this is especially true of mangrove and coastal hardwood hammock in South Florida. Only 4 active BBS routes in Monroe and Miami-Dade counties (#s 25-135, 25-180, 25-283, and 25-907) include portions of these habitats. Four other routes located in coastal South Florida were discontinued primarily because increased traffic made surveys unsafe. We recommend that additional nonrandom 900 series BBS routes be established where possible, particularly in Collier County, to better monitor avifauna in mangrove and hardwood hammock on a regional scale. The flexibility of nonrandom routes would facilitate surveys in these relatively inaccessible habitats. If necessary, remote sensing (Landsat land-cover files and GIS applications) could be used to delineate a sample area and select potential routes or arrays of points on public lands in coastal southern Florida. Managers of public land or volunteers could be recruited to conduct surveys.

Despite the challenges that face any effort to craft and implement a long-term monitoring program for mangrove landbirds, there are many good reasons to do so. Florida’s mangroves host a unique avifauna. Many of the breeding species occur nowhere else in North America. Despite the
reputation for elusiveness and rarity associated with this group of birds, only Mangrove Cuckoos appear truly uncommon. Some of the mangrove-dependent species are widespread and abundant, and most are abundant in at least some parts of the mangroves. Nonetheless, recent declines of Mangrove Cuckoos and Black-whiskered Vireos in the Ten Thousand Islands area (Lloyd and Doyle 2011) should warn against any complacency that comes with a better understanding of the distribution and abundance of mangrove landbirds. Substantial downward changes in population size appear to be underway, at least in the northern part of the mangroves, but whether these changes are cyclical or reflect the deterministic outcome of ongoing environmental changes is unknown. In either case, some form of population monitoring is needed both to understand the scope and extent of the problems facing mangrove landbirds and as a necessary prelude to implementing conservation efforts. Information gleaned thus far and presented in this report can be used as a first step towards that goal.

**Research and management priorities.**

- Establish and implement a multi-agency strategy to monitor trends in abundance and distribution of mangrove landbirds.
- Determine the extent of bias associated with conducting surveys along trails and creeks, for example by comparing abundance estimates with estimates generated from points placed at varying distances from a vegetation edge.
- Document habitat associations of mangrove landbirds using individual-based study designs.
- Predict climate-change effects on mangrove landbirds using habitat- or niche-based models and develop mitigation strategies as needed.
- Determine whether the use of playbacks to increase detectability of Mangrove Cuckoo changes detectability of other focal species.
- Document natural history (e.g., nesting ecology, seasonal movements, and molt patterns) of poorly known species, especially Mangrove Cuckoo.
Table 1. Probability of detection \((p; \pm S.E.)\); average number counted per survey \((\pm S.E.)\); estimated average abundance \((\pm S.E.)\), corrected for detectability; estimated average density \((\text{no. birds/ha}; \pm S.E)\), corrected for detectability; and percent of points present.

<table>
<thead>
<tr>
<th>Species</th>
<th>(p)</th>
<th>Average count</th>
<th>Abundance</th>
<th>Density</th>
<th>% present</th>
</tr>
</thead>
<tbody>
<tr>
<td>Red-bellied Woodpecker</td>
<td>0.78 (0.03)</td>
<td>1.66 (0.04)</td>
<td>2.13 (0.05)</td>
<td>0.30 (0.010)</td>
<td>96.9</td>
</tr>
<tr>
<td>Northern Cardinal</td>
<td>0.80 (0.04)</td>
<td>1.36 (0.05)</td>
<td>1.69 (0.05)</td>
<td>0.23 (0.009)</td>
<td>92.4</td>
</tr>
<tr>
<td>Prairie Warbler</td>
<td>0.75 (0.05)</td>
<td>1.12 (0.03)</td>
<td>1.49 (0.06)</td>
<td>0.23 (0.013)</td>
<td>83.2</td>
</tr>
<tr>
<td>White-eyed Vireo</td>
<td>0.89 (0.02)</td>
<td>1.05 (0.03)</td>
<td>1.18 (0.03)</td>
<td>0.19 (0.004)</td>
<td>75.1</td>
</tr>
<tr>
<td>Red-winged Blackbird</td>
<td>0.83 (0.04)</td>
<td>0.72 (0.04)</td>
<td>0.86 (0.04)</td>
<td>0.11 (0.005)</td>
<td>36.1</td>
</tr>
<tr>
<td>Great Crested Flycatcher</td>
<td>0.69 (0.08)</td>
<td>0.54 (0.04)</td>
<td>0.78 (0.05)</td>
<td>0.11 (0.010)</td>
<td>68.6</td>
</tr>
<tr>
<td>American Crow</td>
<td>0.67 (0.11)</td>
<td>0.44 (0.03)</td>
<td>0.66 (0.02)</td>
<td>0.05 (0.007)</td>
<td>44.8</td>
</tr>
<tr>
<td>Pileated Woodpecker</td>
<td>0.75 (0.05)</td>
<td>0.39 (0.02)</td>
<td>0.52 (0.04)</td>
<td>0.04 (0.003)</td>
<td>56.9</td>
</tr>
<tr>
<td>Black-whiskered Vireo</td>
<td>0.66 (0.13)</td>
<td>0.31 (0.03)</td>
<td>0.46 (0.03)</td>
<td>0.07 (0.011)</td>
<td>41.2</td>
</tr>
<tr>
<td>White-crowned Pigeon</td>
<td>0.72 (0.10)</td>
<td>0.33 (0.03)</td>
<td>0.46 (0.01)</td>
<td>0.06 (0.006)</td>
<td>21.3</td>
</tr>
<tr>
<td>Keys &amp; Florida Bay</td>
<td></td>
<td>1.50 (0.04)</td>
<td>1.89 (0.13)</td>
<td>0.29 (0.041)</td>
<td>83.3</td>
</tr>
<tr>
<td>Common Yellowthroat</td>
<td>0.71 (0.12)</td>
<td>0.07 (0.02)</td>
<td>0.10 (0.02)</td>
<td>0.01 (0.002)</td>
<td>13.2</td>
</tr>
<tr>
<td>Red-shouldered Hawk</td>
<td>0.76 (0.11)</td>
<td>0.13 (0.01)</td>
<td>0.17 (0.02)</td>
<td>0.01 (0.002)</td>
<td>31.7</td>
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<tr>
<td>Yellow Warbler</td>
<td>0.69 (0.17)</td>
<td>0.10 (0.01)</td>
<td>0.15 (0.01)</td>
<td>0.02 (0.003)</td>
<td>11.2</td>
</tr>
<tr>
<td>Keys &amp; Florida Bay</td>
<td></td>
<td>0.59 (0.04)</td>
<td>0.64 (0.08)</td>
<td>0.08 (0.021)</td>
<td>38.8</td>
</tr>
<tr>
<td>Mangrove Cuckoo</td>
<td>0.58 (0.14)</td>
<td>0.09 (0.01)</td>
<td>0.15 (0.02)</td>
<td>0.03 (0.005)</td>
<td>24.9</td>
</tr>
<tr>
<td>Gray Kingbird</td>
<td>0.73 (0.10)</td>
<td>0.15 (0.02)</td>
<td>0.20 (0.01)</td>
<td>0.03 (0.003)</td>
<td>19.1</td>
</tr>
<tr>
<td>Keys &amp; Florida Bay</td>
<td></td>
<td>0.63 (0.05)</td>
<td>0.85 (0.06)</td>
<td>0.15 (0.021)</td>
<td>83.3</td>
</tr>
<tr>
<td>Carolina Wren</td>
<td>0.83 (0.08)</td>
<td>0.12 (0.01)</td>
<td>0.14 (0.01)</td>
<td>0.02 (0.001)</td>
<td>27.2</td>
</tr>
<tr>
<td>Mourning Dove</td>
<td>0.87 (0.09)</td>
<td>0.10 (0.01)</td>
<td>0.11 (0.01)</td>
<td>0.01 (0.001)</td>
<td>16.0</td>
</tr>
<tr>
<td>Keys &amp; Florida Bay</td>
<td></td>
<td>0.31 (0.05)</td>
<td>0.36 (0.03)</td>
<td>0.05 (0.001)</td>
<td>44.6</td>
</tr>
<tr>
<td>Common Grackle</td>
<td>-</td>
<td>0.06 (0.01)</td>
<td>-</td>
<td>-</td>
<td>14.0</td>
</tr>
<tr>
<td>All landbirds combined(^1)</td>
<td>0.76 (0.02)</td>
<td>8.98 (0.05)</td>
<td>11.81 (0.02)</td>
<td>1.67 (0.036)</td>
<td></td>
</tr>
</tbody>
</table>

\(^1\) Includes the 18 species listed in this table, plus the additional 13 mangrove landbirds with too few detections to estimate density.
Table 2. Estimated abundance (number of birds in the 150-m radius circle around a survey point) based on distance sampling conducted at points surveyed between 2008 and 2011.

<table>
<thead>
<tr>
<th>Species</th>
<th>Average abundance</th>
<th>95% confidence interval</th>
<th>Probability of detection</th>
<th>95% confidence interval</th>
</tr>
</thead>
<tbody>
<tr>
<td>White-eyed Vireo</td>
<td>5.04*a</td>
<td>4.76-5.33</td>
<td>0.19</td>
<td>0.18-0.20</td>
</tr>
<tr>
<td>Prairie Warbler</td>
<td>3.76*a</td>
<td>3.34-4.12</td>
<td>0.27</td>
<td>0.25-0.28</td>
</tr>
<tr>
<td>Red-bellied Woodpecker</td>
<td>3.20</td>
<td>2.84-3.20</td>
<td>0.42</td>
<td>0.41-0.44</td>
</tr>
<tr>
<td>Northern Cardinal</td>
<td>2.63*a</td>
<td>2.49-2.70</td>
<td>0.44</td>
<td>0.42-0.46</td>
</tr>
<tr>
<td>White-crowned Pigeon</td>
<td>0.12</td>
<td>0.09-0.15</td>
<td>0.26</td>
<td>0.23-0.29</td>
</tr>
<tr>
<td>Keys &amp; Florida Bay</td>
<td>2.06</td>
<td>1.56-2.70</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Great Crested Flycatcher</td>
<td>1.35*a</td>
<td>1.28-1.49</td>
<td>0.38</td>
<td>0.33-0.43</td>
</tr>
<tr>
<td>Black-whiskered Vireo</td>
<td>1.14</td>
<td>0.92-1.35</td>
<td>0.26</td>
<td>0.23-0.30</td>
</tr>
<tr>
<td>Gray Kingbird</td>
<td>0.06</td>
<td>0.04-0.08</td>
<td>0.27</td>
<td>0.21-0.35</td>
</tr>
<tr>
<td>Keys &amp; Florida Bay</td>
<td>0.99</td>
<td>0.71-1.42</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Yellow Warbler</td>
<td>0.04</td>
<td>0.03-0.06</td>
<td>0.33</td>
<td>0.25-0.43</td>
</tr>
<tr>
<td>Keys &amp; Florida Bay</td>
<td>0.78*a</td>
<td>0.64-0.92</td>
<td></td>
<td></td>
</tr>
<tr>
<td>American Crow</td>
<td>0.36</td>
<td>0.28-0.43</td>
<td>0.72</td>
<td>0.57-0.90</td>
</tr>
<tr>
<td>Common Yellowthroat</td>
<td>0.36</td>
<td>0.21-0.50</td>
<td>0.20</td>
<td>0.15-0.26</td>
</tr>
<tr>
<td>Carolina Wren</td>
<td>0.24</td>
<td>0.14-0.36</td>
<td>0.28</td>
<td>0.20-0.38</td>
</tr>
<tr>
<td>Pileated Woodpecker</td>
<td>0.23*a</td>
<td>0.21-0.28</td>
<td>0.74</td>
<td>0.68-0.81</td>
</tr>
<tr>
<td>Common Grackle</td>
<td>0.23</td>
<td>0.14-0.36</td>
<td>0.19</td>
<td>0.13-0.27</td>
</tr>
<tr>
<td>Mourning Dove</td>
<td>0.14</td>
<td>0.07-0.21</td>
<td>0.46</td>
<td>0.34-0.62</td>
</tr>
<tr>
<td>Mangrove Cuckoo</td>
<td>0.11</td>
<td>0.07-0.18</td>
<td>0.51</td>
<td>0.35-0.76</td>
</tr>
</tbody>
</table>

*a Abundance estimate was generated from a model showing a significant lack of fit to the data.
Table 3. Average number of individuals of selected mangrove landbird species (excluding those with <100 detections) detected per point from 2008-2011, excluding points at which the species was never detected. Data were collected in a 150-m radius around 354 points in mangrove vegetation in southern Florida.

<table>
<thead>
<tr>
<th>Species</th>
<th>Average no. detected per point</th>
<th>95% confidence interval</th>
</tr>
</thead>
<tbody>
<tr>
<td>Red-winged Blackbird</td>
<td>1.94</td>
<td>1.78-2.09</td>
</tr>
<tr>
<td>Red-bellied Woodpecker</td>
<td>1.81</td>
<td>1.61-2.01</td>
</tr>
<tr>
<td>White-eyed Vireo</td>
<td>1.47</td>
<td>1.37-1.60</td>
</tr>
<tr>
<td>Northern Cardinal</td>
<td>1.41</td>
<td>1.33-1.49</td>
</tr>
<tr>
<td>Prairie Warbler</td>
<td>1.36</td>
<td>1.25-1.46</td>
</tr>
<tr>
<td>Yellow Warbler</td>
<td>1.32</td>
<td>0.97-1.67</td>
</tr>
<tr>
<td>White-crowned Pigeon</td>
<td>1.30</td>
<td>1.02-1.55</td>
</tr>
<tr>
<td>Great Crested Flycatcher</td>
<td>0.83</td>
<td>0.76-0.90</td>
</tr>
<tr>
<td>Black-whiskered Vireo</td>
<td>0.83</td>
<td>0.72-0.93</td>
</tr>
<tr>
<td>American Crow</td>
<td>0.82</td>
<td>0.73-0.91</td>
</tr>
<tr>
<td>Pileated Woodpecker</td>
<td>0.82</td>
<td>0.76-0.89</td>
</tr>
<tr>
<td>Gray Kingbird</td>
<td>0.74</td>
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<tr>
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<td>Mourning Dove</td>
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<td>0.41-0.62</td>
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<td>Carolina Wren</td>
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<td>Common Grackle</td>
<td>0.39</td>
<td>0.27-0.51</td>
</tr>
<tr>
<td>Mangrove Cuckoo</td>
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<td>0.33-0.41</td>
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<tr>
<td>Red-shouldered Hawk</td>
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Table 4. Average number (and 95% confidence intervals) of individuals of mangrove landbirds detected per survey point from 2008-2011 within each of the 5 administrative areas included in the study. Confidence intervals were estimated using a non-parametric bootstrap routine. Areas are organized from northernmost (TTINWR) to southernmost (FKNWR).

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<th>BNP</th>
<th>FDEP</th>
<th>FKNWR</th>
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<td>1.02</td>
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<td>(0.83-1.21)</td>
<td>(0.94-1.71)</td>
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<td>(0.04-0.23)</td>
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<td>(0.02-0.15)</td>
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<td>0.19</td>
<td>0.03</td>
<td>0.06</td>
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<td>0.02</td>
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a Ten Thousand Islands National Wildlife Refuge  
b Everglades National Park  
c Biscayne National Park  
d Florida Department of Environmental Protection, specifically John Pennekamp Coral Reef State Park  
e Great White Heron National Wildlife Refuge and National Key Deer Refuge, collectively the Florida Keys National Wildlife Refuges
FIGURES
Figure 1. Extent of mangrove vegetation surveyed for mangrove landbirds from 2008-2011.
Figure 2. Location of points surveyed in Everglades National Park for mangrove landbirds in 2008-2011.
Figure 3. Location of points surveyed in Ten Thousand Islands National Wildlife Refuge (inclusive of Rookery Bay National Estuarine Research Reserve, and including parts of Collier-Seminole State Park and Fakahatchee Strand Preserve State Park) for mangrove landbirds in 2008-2011.
Figure 4. Location of points surveyed in Great White Heron National Wildlife Refuge and National Key Deer Refuge for mangrove landbirds in 2008-2011.
Figure 5. Location of points surveyed at Biscayne National Park and John Pennekamp Coral Reef State Park for mangrove landbirds in 2008-2011.
Figure 6. Accumulation of detections as a function of time elapsed during a survey. The dotted line shows the average accumulation rate of all species; the solid black lines are 95% confidence limits. The solid gray line shows the accumulation rate for Mangrove Cuckoo.
Figure 7. Surveys points where at least one Red-bellied Woodpeckers was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 8. Surveys points where at least one Northern Cardinal was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 9. Surveys points where at least one Prairie Warbler was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 10. Surveys points where at least one Black-whiskered Vireo was detected (blue circles) and survey points where none were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 11. Surveys points where at least one Great Crested Flycatcher was detected (blue circles) and survey points where none were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 12. Surveys points where at least one Mangrove Cuckoo was detected (blue circles) and survey points at which no detections were recorded (red circles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 13. Surveys points where at least one Red-winged Blackbird was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 14. Surveys points where at least one White-crowned Pigeon was detected (blue circles) and survey points where none were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 15. Surveys points where at least one Gray Kingbird was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 16. Surveys points where at least one Yellow Warbler was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 17. Surveys points where at least one Common Yellowthroat was detected (blue circles) and survey points where none were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 18. Surveys points where at least one Common Grackle was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 19. Surveys points where at least one White-eyed Vireo was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 20. Surveys points where at least one American Crow was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 21. Surveys points where at least one Carolina Wren was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 22. Surveys points where at least one Pileated Woodpecker was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2010. Green shading is extent of mangroves.
Figure 23. Surveys points where at least one Mourning Dove was detected (blue circles) and survey points at which no detections were recorded (red triangles) during 10-minute point-count surveys conducted from 2008-2011. Green shading is extent of mangroves.
Figure 24. Vegetation features that were significantly related to variation in the number of White-crowned Pigeons counted at points in the mangroves of south Florida during 2008-2011. Solid lines are the mean response; dotted lines are 95% confidence limits.
Figure 25. Vegetation features that were significantly related to variation in the number of Gray Kingbirds counted at points in the mangroves of south Florida during 2008-2011. Solid lines are the mean response; dotted lines are the 95% confidence limits.
Figure 26. Vegetation features that were significantly related to variation in the number of Black-whiskered Vireos counted at points in the mangroves of south Florida during 2008-2011. Solid lines are the mean response; dotted lines are the 95% confidence limits.
Figure 27. Vegetation features that were significantly related to variation in the number of Yellow Warblers counted at points in the mangroves of south Florida during 2008-2011. Solid line is the mean response; dotted lines are the 95% confidence limits.
Figure 28. Vegetation features that were significantly related to variation in the number of Prairie Warblers counted at points in the mangroves of south Florida during 2008-2011. Solid lines show the mean response; dotted lines are the 95% confidence limits.
Figure 29. Vegetation features that were significantly related to variation in the number of White-crowned Pigeons counted at points in the mangroves of Florida Bay and the Florida Keys during 2008-2011. Solid lines show the mean response; dotted lines are 95% confidence limits.
Figure 30. Statistical power to detect a 50% decline in abundance over a 20-year period as a function of sampling frequency (every year, every other year, or every 5 years) and the number of survey points sampled per survey event.
Figure 31. Seasonal patterns of detection for 5 species of mangrove landbirds in Florida. Blue lines show number of detections per survey and black lines are the 1-week moving average of the total number of detections per survey.
LITERATURE CITED


Appendix A. Total number of individuals counted, by species.
**Appendix A.** Number of individuals detected, by species, during breeding-season point-count surveys conducted between 2008 and 2011 at 347 locations in mangroves of Florida. Species identified as mangrove landbirds were the focus of this study.

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<th>Common Name</th>
<th>Scientific Name</th>
<th>Total counted</th>
<th>Landbird?</th>
</tr>
</thead>
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<td>2371</td>
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<td>White-eyed Vireo</td>
<td><em>Vireo griseus</em></td>
<td>1501</td>
<td>Yes</td>
</tr>
<tr>
<td>Red-winged Blackbird</td>
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<td>-----------------------------</td>
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Appendix A. Continued.

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<th>Common Name</th>
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<td>Wood Stork</td>
<td><em>Mycteria americana</em></td>
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</table>

a These raptors were not included in analyses because most individuals were detected flying over survey points, and so cannot be clearly associated with any particular survey location, and because their wide-ranging nature raises the risk of counting the same individual multiple times at different survey locations.

b These landbirds were not included in analyses because they use the mangroves only during the winter or as a stopover site during northward migration.

These resident landbirds were not included in analyses because their use of the mangroves is likely incidental, based on habitat requirements (e.g., ground-nesting Common Nighthawks and Northern Bobwhite are unlikely to be residents in the mangroves).
Appendix B. Representative photographs of the study area.
Photograph 1. Tall, closed-canopy red-mangrove (*Rhizophora mangle*) forest on Pumpkin Key in the Ten Thousand Islands region of southwest Florida. Trees in the foreground are approximately 6 m in height; the taller trees in the background are approximately 15 m in height.
Photograph 2. A black-mangrove (*Avicennia germinans*) dominated shrubland on Lower Sugarloaf Key, Florida. Canopy height is approximately 2 m.
Photograph 3. Dry, extra-tidal mangrove shrubland, dominated by black mangrove (*Avicennia germinans*), on Big Torch Key, Florida. Canopy height is less than 2 m; the taller trees in the foreground are approximately 4 m.
Photograph 4. The ecotone between freshwater marsh and mangrove forest in the Canepatch area of Everglades National Park, Florida.
Photograph 5. Tropical-hardwood hammock in the Canepatch area of Everglades National Park, Florida.
Photograph 6. A large canopy gap in a tall, mixed-mangrove forest along Broad River, Everglades National Park, Florida.
Photograph 7. Open-canopied, black-mangrove woodland next to the Snake Bight trail in Everglades National Park, Florida. These forests are not subject to daily tidal inundation and the understory is primarily *Batis maritima*.
Photograph 8. A fringe of relatively short (3 – 4 m) red mangrove (*Rhizophora mangle*) along a tidal creek near Card Sound, Florida. This is a very common forest type along smaller creeks throughout Florida’s mangroves.
Photograph 9. Short red-mangrove (*Rhizophora mangle*) forest at Hell’s Bay, with paurotis palm (*Acoelorrhaphe wrightii*) rising in the background.
Appendix C. Species accounts.

**Red-shouldered Hawk.** Red-shouldered Hawks were detected throughout the mangroves, including on the small islands of Florida Bay, although we detected them only rarely in the Keys (two detections of lone individuals in the lower Keys, and a handful on Key Largo). They showed little evidence of a preference for one mangrove type over another.

**White-crowned Pigeon.** In Florida, White-crowned Pigeons breed primarily on the small mangrove islands of Florida Bay, less often on the mainline Florida Keys, and forage and roost in hardwood forests on the lower peninsula and mainline Keys. To avoid disturbing nesting individuals, we eliminated any survey points that fell on islands with known nesting colonies, and so many of the birds detected during our surveys were likely found while foraging or roosting. Indeed, we found White-crowned Pigeons at highest densities in the lower Keys and on upper Key Largo, both of which have areas of upland hardwood forest. We found them at lower densities in John Pennekamp Coral Reef State Park, on the mangrove islands of Florida Bay, and along the south coast of the peninsula along the Snake Bight and Rowdy Bend trails in Everglades National Park. Birds detected along the south coast of the peninsula were likely using the hardwoods that occur along Snake Bight and, to a lesser extent, Rowdy Bend. Individual birds were detected at inland sites in Everglades National Park at West Lake and in Hell’s Bay. One notable record occurred far inland at the Canepatch area of Everglades National Park, which falls at the ecotone between mangrove and freshwater marsh but which also supports relatively large patches of hardwood hammock. Stevenson and Anderson (1994) suggested that a breeding population may once have occurred in the Ten Thousand Islands region, but we recorded no detections at any of the 108 points that we surveyed in that region (Terry Doyle also failed to record any detections of White-crowned Pigeons when surveying these same points from 1999-2008).

**Eurasian Collared-Dove.** Eurasian Collared-Dove is a recent arrival to Florida, colonizing the state in the late 1970s or early 1980s from populations established earlier in the Bahamas (Woolfenden and Robertson 2006). This species appears to use mangroves incidentally and was detected only in mangroves adjacent to developed areas. Individuals were detected at two points in the narrow fringe of mangroves along Biscayne Bay in Biscayne National Park, at two points on Big Pine Key, at one point on Big Torch Key, at two points in the Card Sound area, and at two points in John Pennekamp Coral Reef State Park. Given its habitat of foraging for seeds on the ground, this species is unlikely to be found in mangroves except along edges or in dry, extra-tidal forests.
**Mourning Dove.** Two subspecies of Mourning Dove occur within the study area: *Zenaida macroura carolinensis*, the widespread form of the eastern United States, and *Z. m. macroura*, a West Indian form with a North American distribution apparently limited to the Florida Keys (Aldrich and Duvall 1958). Stevenson and Anderson (1994) reported that this species was rare or absent in the Keys until the 1940s or 1950s, but that more recently it had increased in numbers throughout the Keys. Our findings corroborate this suggestion: Mourning Doves reached their greatest abundance in the Keys, with both density and the frequency of occurrence declining to the north. We found a large gap in the distribution of this species, beginning on the islands of Florida Bay and continuing into most of Biscayne National Park and Everglades National Park, including the southwest coast. In the Ten Thousand Islands region, Mourning Doves were again found in relatively high densities.

We found Mourning Doves at points surrounded by extensive mangrove forest, unlike Eurasian-Collared Doves, which were found primarily at points adjacent to uplands or developed areas. For example, this species was detected regularly in the Ten Thousand Islands region, which is largely undeveloped and supports large stands of mangrove forest. However, Mourning Doves in this region avoided most of the offshore keys, instead using mangrove forests further inland along the tidal rivers that drain into the various inland bays. Thus, individuals may be using the mangroves for nesting but travelling farther afield to find open, dry areas suitable for foraging. A similar segregation of nesting and foraging habitat might explain the use of mangrove forests in the Keys, where most patches of mangroves were within a relatively short distance of areas more suitable for a ground-foraging bird. As a caveat, we have no direct evidence that Mourning Doves nest in mangroves.

The split distribution that we observed – higher densities in the south and north of the study area, but absent from the middle – might also reflect the current distribution of the two subspecies that occur in Florida. Without body-size measurements, we are unable to determine whether the individuals in the Keys and the Ten Thousand Islands region represent *macroura* and *carolinensis*, respectively, but data in Aldrich and Duvall (1958) and Stevenson and Anderson (1994) would suggest that this is the case. If the Keys population is composed of individuals of *macroura*, then the high densities of individuals in that region may reflect a growing population of this recent colonist. The absence of individuals on apparently suitable islands in Florida Bay and along the extra-tidal buttonwood and black-mangrove forests of the south coast of the Peninsula may then reflect a temporary gap as the Keys population gradually extends its range northward. To the north, the population in the Ten Thousand...
Islands region may be composed of individuals of *carolinensis*, perhaps less suited to the increasingly tropical (and perhaps arid) conditions further south.

**Common Ground-Dove.** Neither Stevenson and Anderson (1994) nor Robertson and Kushlan (1974) identify mangroves as habitat for this species, although Common Ground-Doves nest in mangroves in other parts of their range (e.g., Costa Rica, Barrantes [1998]; but not in mangroves on Little Cayman, Diamond [1980]). In Florida, Common Ground-Doves are a permanent resident throughout the state. We found them scattered throughout every region of our study area, from Lower Sugarloaf Key, through the islands of Florida Bay, along the southwest coast, and into the Ten Thousand Islands. Individuals were always detected as pairs or lone individuals, and at only two points – one in the fringing forests of Biscayne Bay and one on Lower Sugarloaf Key – did we detect individuals in more than one year. The lack of consistent detections at any survey point complicates interpretation of our data, as does the extended breeding season of this species (Stevenson and Anderson 1994). Were the individuals that we detected breeding at the survey location, or simply wandering individuals using the mangroves in a transient fashion?

**Yellow-billed Cuckoo.** Yellow-billed Cuckoos breed throughout Florida, including in forest edges in the Keys (Stevenson and Anderson 1994). We recorded no detections of this species until 2011, when we detected 29 individuals at 19 different survey points, clustered in the Keys and the southwest coast of Everglades National Park. Detections were noted only in late May and June of 2011. Given the apparently extended migration period of this species (Stevenson and Anderson 1994), coupled with the lack of any previous records during our surveys, we suspect that all of the detections of Yellow-billed Cuckoos reflected late, northward migrants.

**Mangrove Cuckoo.** Robertson and Woolfenden (1992) and Stevenson and Anderson (1994) both argued that the species was a permanent resident based on increasing numbers of winter sightings beginning in the mid-1950s. Robertson and Kushlan (1974), Robertson and Woolfenden (1992), and Woolfenden at al. (2006) further suggested that this species had expanded its range in recent years, especially into interior locations of the southern peninsula.

We detected Mangrove Cuckoos throughout the study area, with the exception of the islands of Florida Bay, the dry buttonwood forests of the south coast of the peninsula, and the interior mangrove shrublands of Everglades National Park (e.g., Hell’s Bay). This species was never detected in great
numbers, with the highest densities occurring in the Ten Thousand Islands region and on the northern end of Key Largo near Card Sound. We found intermediate densities of Mangrove Cuckoos in the lower Keys; in the tall, riverine forests along the southwest coast; and along the edge of Biscayne Bay. Biscayne Bay appears to support a relatively robust population; although densities were not especially high, 19 of the 25 points surveyed in those forests supported at least one Mangrove Cuckoo during the course of our surveys. Outside of the Ten Thousand Islands region, the only other locations with consistent detections of Mangrove Cuckoos were along Broad River (detections at 7 of 11 survey points) and Harney River (6 of 15 survey points yielded Mangrove Cuckoos).

**Red-bellied Woodpecker.** Red-bellied Woodpeckers are a year-round resident throughout the state of Florida (Stevenson and Anderson 1994). Red-bellied Woodpeckers were abundant and ubiquitous during the course of our study and were found in every type of mangrove forest that we surveyed. The only apparent limiting factor for this species is the availability of trees large enough to support a nest cavity.

**Downy Woodpecker.** Downy Woodpeckers are locally common in interior forests of Florida, but have not been reported from mangroves. This species either maintains a small, isolated breeding population in the mangroves of the southwest coast of the peninsula, or is an incidental visitor in this same region. We detected a total of 7 individuals (one pair and 5 singletons) at 6 points, one of which was along Wood River, just south of the junction of Wood River and Broad River, and the remaining 6 of which were in the Devil’s Fingers area of Everglades National Park. Two of the detections came in 2009, and the other 5 were recorded in 2011. Interestingly, Tufted Titmice, which commonly co-occur with Downy Woodpeckers in the inland forests of southern Florida (e.g., Lloyd and Slater 2011), were once reported to occur along Wood River (Robertson and Kushlan 1974, Stevenson and Anderson 1994), although none were detected during our surveys.

**Northern Flicker.** Northern Flickers are breeding residents throughout the state, including the Keys (Stevenson and Anderson 1994, Bancroft et al. 1995). In winter, resident birds are supplemented by migrants from northerly breeding populations. We detected only a handful of Northern Flickers, with 9 of the 12 observations occurring in the narrow strip of mangroves that border Biscayne Bay. The mangrove forests along Biscayne Bay are adjacent to several roads and variety of other vegetation types.
(e.g., developed areas and agricultural land) that were probably more suitable for this species, which forages primarily on the ground.

**Pileated Woodpecker.** Pileated Woodpeckers are a fairly common resident of Florida and are found in a variety of forest types, including mangroves (Stevenson and Anderson 1994). Robertson and Woolfenden (1992), Stevenson and Anderson (1994), Woolfenden and Robertson (2006), and Cox (2006) all noted a southward decline in the abundance of this species, with relatively few detections in the Keys, especially the lower Keys.

We found Pileated Woodpeckers in every region of the study area, including on two of the small islands of Florida Bay (Joe Kemp Key and Murray Key). Based on estimated densities, the core areas for this species were in the Ten Thousands Islands region and the riverine forests of the southwest coast. We detected single individuals or pairs occasionally throughout the Keys as far south as Big Torch Key. Changes in the prevalence of large trees and abundance of snags likely explain the pattern of distribution of Pileated Woodpeckers in the mangroves. Mangrove forests in the Ten Thousand Island region and along the large rivers of the southwest coast contained many tall, large-diameter trees and snags and thus probably allowed for a greater density of nesting pairs. Further south, especially in the Keys, we noted fewer trees large enough to support nesting cavities for this species.

**Great Crested Flycatcher.** Great Crested Flycatchers breed throughout the state, although rarely in the Keys, and winter in the southern portion of the state (Robertson and Woolfenden 1992). Nests are built in cavities; according to Stevenson and Anderson (1994:425), “almost any cavity or any height will serve”. Great Crested Flycatchers were widespread and found at intermediate densities through much of the study area. The distribution of this species was fairly even, with pockets of high density found in all regions except for Biscayne Bay and the upper Keys. Contrary to the situation described by Robertson and Woolfenden (1992) and Stevenson and Anderson (1994), Great Crested Flycatchers were detected regularly and frequently in the lower Keys, especially on Big Torch Key. This may reflect a continuation of the southward advance of this species, first described by Robertson and Kushlan (1974). The paucity of records of Great Crested Flycatchers in the dense, relatively short forests of the upper Keys and Biscayne Bay may reflect a lack of foraging and nesting habitat in these areas. Bancroft et al. (1995) suggested that the distribution of Great Crested Flycatchers in the Keys might be driven by the availability of nest sites. Indeed, Red-bellied Woodpeckers, which are probably the most important
excavator of cavities used for nesting by Great Crested Flycatchers, showed a similar pattern of
distribution wherein densities were lowest along Biscayne Bay and in the upper Keys.

**Gray Kingbird.** Gray Kingbirds are breeding migrants found in coastal areas, and in some inland cities,
throughout the state (Stevenson and Anderson 1994, Cox 2006). They have a prolonged migration
period, with some fall departures occurring as late as December, only several months before the earliest
northward-moving individuals begin arriving in the state (Stevenson and Anderson 1994). Overall, Gray
Kingbirds were found at relatively low densities, and even with their limited range they never occurred
in great numbers (average density within the portion of the study area occupied = 0.10 individuals per
ha [95% CI = 0.07 – 0.12]). Although small numbers of Gray Kingbirds breed much further north, Gray
Kingbirds were largely restricted to the Keys and the islands of Florida Bay. The two detections at inland
points, both at Lane Bay in Everglades National Park, were of a pair and an individual seen on the same
day in mid-May, 2009, and may have been migrating individuals. Gray Kingbirds – like Great Crested
Flycatchers and Red-bellied Woodpeckers – were absent from Biscayne Bay despite occurring in high
densities nearby on the north end of Key Largo. Why the fringing forests along Biscayne Bay are avoided
by Gray Kingbirds is unclear, but is presumably related to forest structure; given that this species nests in
city parking lots in nearby Homestead, it is unlikely that the lack of records along Biscayne Bay reflects
the extent of human-related disturbance in these forests. Gray Kingbirds were also absent from the tall,
mostly closed-canopy, riverine forests along the southwest coast and in the Ten Thousand Islands
region.

**White-eyed Vireo.** Two weakly differentiated subspecies of White-eyed Vireo – the widespread *Vireo
griseus griseus* and *V. g. maynardi* in the Keys and south coast of the peninsula – breed in Florida, and a
third may winter in or migrate through the state (*V. g. noveboracensis*) (Stevenson and Anderson 1994).
White-eyed Vireos nest in a variety of vegetation types, with the common thread being the presence of
a dense understory of shrubs. Along the coast of Florida, White-eyed Vireos are associated with stands
of red mangrove, which presumably share structural characteristics with the shrubby woodlands that
this species occupied inland and further north (Stevenson and Anderson 1994). White-eyed Vireos were
abundant and widespread in the mangroves. Densities were greatest in the Ten Thousand Islands
region, but we also detected large numbers of individuals at inland sites in Everglades National Park,
including at Hell’s Bay, Lane Bay, and Canepatch. Based on the map in Stevenson and Anderson (1994),
individuals in the Ten Thousands Islands region would belong to *griseus*, whereas individuals further
south in Everglades National Park would belong to *maynardi*. Further south, densities declined substantially, and we detected individuals only sporadically and in small numbers in the lower Keys. Dense, shrubby stands of mangrove were common throughout the Keys, and on the islands of Florida Bay, and so the scarcity of White-eyed Vireos in the southern part of the region is somewhat puzzling and not obviously related to the availability of habitat. White-eyed Vireos in Florida Bay and the Keys are at the far southern edge of their breeding range, and so the low densities that we observed may reflect a lack of adaptation to the environmental conditions – perhaps the increased aridity – present in much of the Keys.

**Black-whiskered Vireo.** Black-whiskered Vireos are a West Indian species thought to have colonized Florida in the 1800s (Stevenson and Anderson 1994). Outside of Florida, they are found in a variety of forests; in the state, they are limited to mangrove forests and tropical hammocks. Robertson and Woolfenden (1992) and Woolfenden and Robertson (2006) argued for a northward and inland range expansion among Black-whiskered Vireos in recent decades. We found little evidence of extensive use of inland mangroves, but otherwise Black-whiskered Vireos were evenly distributed in most regions of the study area. Behind the Florida Prairie Warbler, Black-whiskered Vireo was the second most abundant species of the mangrove specialists. The only regions with consistently low densities of Black-whiskered Vireos were Biscayne Bay and the upper Keys.

**Blue Jay.** Robertson (1955) and Robertson and Kushlan (1974) suggested that Blue Jays in southern Florida had adapted to the loss of their primary habitat – pine rocklands – by becoming birds of forest edge and suburbia. Perhaps accompanying this shift in habitat utilization, Blue Jays were also reported to have expanded their range in the past century to include parts of the Keys thought to have been previously unoccupied (Robertson and Woolfenden 1992, Woolfenden and Robertson 2006). Not noted as an inhabitant of mangroves, we detected 49 Blue Jays at 19 different survey points, 3 in the Ten Thousand Islands Region and 16 in Biscayne Bay and the upper Keys (near Card Sound and in John Pennekamp Coral Reef State Park). At 5 of these points – 2 at Biscayne Bay and 3 in John Pennekamp Coral Reef State Park – individuals were detected in 2 or more years. All of the other detections were of lone individuals or pairs detected on a single occasion. With the exception of the points in the Ten Thousand Islands Region, all of the mangroves in which we detected Blue Jays had areas of commercial, residential, or agricultural development nearby. Given this, and the mostly irregular nature of the
detections, we believe that Blue Jays are either occasional, incidental visitors to the mangroves or a very uncommon nesting species in mangroves adjacent to suburban development.

**American Crow.** American Crows nest during the dry season, typically January through March, and so our surveys fell well outside the breeding season of this species (Stevenson and Anderson 1994). American Crows are considered largely a species of inland habitats, except for an association, presumably predatory, with nesting colonies of waterbirds in Florida Bay (Robertson and Kushlan 1974). Historically, American Crows were considered absent from the Keys (Robertson and Kushlan 1974), although more recently they have been described as occasional stragglers to the region (Robertson and Woolfenden 1992, Stevenson and Anderson 1994). We found American Crow to be an abundant member of the mangrove landbird assemblage, but with a fairly patchy distribution. American Crows reached their greatest densities in inland areas along the south coast, especially along the Snake Bight and Rowdy Bend trails, Hell’s Bay, and Lane Bay. American Crows occur regularly along the main road through Everglades National Park, usually feeding on roadkill, and so some of the elevated densities that we observed in surrounding areas may reflect the subsidies inadvertently provided by the road and the cluster of development in Flamingo. American Crows were detected frequently, albeit in lower numbers, in the large forests of the southwest coast and in the Ten Thousand Islands region. As reported by previous authors, American Crows were largely absent from the Keys and Biscayne Bay. We detected lone individuals at 5 different points, with the southernmost record coming from Lower Sugarloaf Key.

**Fish Crow.** Considered the “city crow” of southern Florida, Fish Crows are common throughout much of northern and central Florida but are restricted to the coasts and cities of south Florida (Stevenson and Anderson 1994). Robertson and Woolfenden (1992) reported that Fish Crows were an irregular visitor to the Keys, but noted the possibility that a small resident population might exist in the lower Keys. We detected only 6 Fish Crows; 3 at 2 different points in John Pennekamp Coral Reef State Park, 2 at 2 different points along Biscayne Bay, and 1 on Clive Key in Florida Bay. Unlike American Crows, Fish Crows breed during the period in which we conducted our surveys, so the records reported here could reflect breeding individuals. However, absent evidence of breeding, we consider Fish Crows to be an occasional visitor to and possible rare breeder in the mangroves of south Florida.
**Carolina Wren.** Found virtually anywhere in the eastern United States where dense, shrubby vegetation exists, Carolina Wren is common throughout much of Florida (Robertson and Woolfenden 1992, Stevenson and Anderson 1994). Carolina Wrens are uncommon in the southern peninsula and the Keys, apparently breeding south of Key Largo only on rare occasions (Robertson and Woolfenden 1992).

Our findings portray Carolina Wren as a continental species reaching the edge of its range in the mangroves; they were neither widespread nor abundant even where present, and had a distribution limited primarily to the northern half of the study area. In the Ten Thousand Islands region, where this species reached its greatest density, individuals were absent from the keys and were found only at inland locations along the major tidal rivers (e.g., Blackwater River). We detected 4 individuals at 3 points on Key Largo, but found no individuals at any points further south in the Keys.

**Northern Mockingbird.** We detected only 38 Northern Mockingbirds (Appendix A) at 15 different points. Individuals were detected at six points around Biscayne Bay, seven points around Card Sound, at one point on Cudjoe Key, and at one point on Big Pine Key. Similar patterns of distribution were noted for other species associated with settled areas (e.g., Eurasian Collared-Dove), and so we assume that the presence of Northern Mockingbirds in the mangroves is largely incidental.

**Common Yellowthroat.** Common Yellowthroats breed throughout North America in areas with low, dense vegetation. Resident breeding populations in Florida are supplemented by winter migrants which begin arriving in September and which linger as late as May (Stevenson and Anderson 1994). During the winter, Common Yellowthroats are found throughout the state, including the Keys, but breeding populations are thought to occur no further south than southern mainland (Stevenson and Anderson 1994). Common Yellowthroats were neither abundant nor widespread in the mangroves. They exhibited a unique pattern of distribution that probably reflects their closer association with habitats other than mangroves. Pockets of high density were noted at several inland locations in Everglades National Park – not surprising given the association that this species has with freshwater marsh – and in the lower Keys. Detections of lone individuals were recorded at points scattered throughout all other regions of the study area with the exception of the Ten Thousand Islands area, where none were detected. The presence of Common Yellowthroats in the lower Keys is somewhat unexpected, given the presumed lack of breeding populations south of the mainland (e.g., Stevenson and Anderson 1994). However, reported migration dates for wintering individuals extend well into May (Stevenson and Anderson 1994), and so we cannot rule out the possibility that the individuals we detected in the Keys
were winter migrants. Indeed, most of the detections from the Keys were from late April or early May, and later surveys at these same points yielded no detections. For example, on 30 April 2009, 5 individuals were counted at a point on Horseshoe Key in Great White Heron National Wildlife Refuge; 5 weeks later none were counted at this point. However, another point on Horseshoe Key yielded 2 individuals on 30 April 2009 and one individual on 26 May 2009. The persistence of at least one individual through the spring is suggestive of an attempt at breeding. Overall, however, Common Yellowthroats appear to be only an occasional and opportunistic inhabitant of the mangroves, with a distribution limited largely to mangroves found near inland marshes.

**Yellow Warbler.** Cuban Yellow Warblers, presumably colonists from either the Bahamas or Cuba, were first discovered in the state when a pair was found nesting in 1941 in Great White Heron National Wildlife Refuge (Greene 1942). At least in Florida, they nest exclusively in mangroves (Stevenson and Anderson 1994, Prather and Cruz 1995). Since their discovery, Yellow Warblers have spread up the Keys and northward through Florida Bay. According to unpublished sources cited by Robertson and Woolfenden (1992) and Stevenson and Anderson (1994), the breeding range extends into Collier County on the Gulf side and to the north end of Biscayne Bay on the Atlantic side of the peninsula. However, we found no evidence that Cuban Yellow Warblers have extended their range to include the mainland.

Because of their limited distribution within the study area, overall density of Cuban Yellow Warblers was relatively low. Within the Keys and the islands of Florida Bay, however, we found them at much higher densities: 0.65 individuals per ha (95% CI = 0.60 – 0.70). Indeed, when present, they were one of the most abundant species detected. Cuban Yellow Warblers reached their greatest densities on the small islands of Florida Bay. We detected this species on every surveyed island in Florida Bay with the exception of Big Key, which was surveyed twice in 2011. We also found Cuban Yellow Warblers at high density at some points in the lower Keys, although they were also absent from many points on the mainline keys, including Cudjoe Key and Big Torch Key. Further north, we found them at low density in the short red mangrove forests at John Pennekamp Coral Reef State Park and around Card Sound. We recorded only one individual on the mainland, a singing male detected during a single survey along the Snake Bight Trail in Everglades National Park in 2009. Yellow Warblers were not detected in Biscayne Bay, the southwest coast of Everglades National Park, and the Ten Thousand Islands region. Individuals probably wander north of Florida Bay on occasion, but we found no evidence that this species has expanded its regular breeding range to the extent suggested by Robertson and Woolfenden (1992) or Stevenson and Anderson (1994). Bird surveys of the 108 points that we surveyed in the Ten Thousand
Islands region have been conducted annually since 1999, and Yellow Warblers have never been detected during that time (Terry Doyle, USFWS, unpublished data). Based on our data, Cuban Yellow Warblers are limited primarily, if not exclusively, to the Keys and to the islands of Florida Bay; records of breeding individuals further northward would be of note and should be carefully examined to rule out the possibility that individuals are migrants belonging to one of the more northerly subspecies in the aëstiva group, which are uncommon spring migrants (Stevenson 1957, Stevenson and Anderson 1994).

**Prairie Warbler.** The endemic subspecies of Prairie Warbler that inhabits Florida’s mangrove was first described by Howell (1930). Believed to have diverged in the late Pleistocene (c.a. 70,000 years ago; Buerkle [1999]) from a common ancestor shared with the nominate subspecies, *Setophaga discolor discolor*, Florida Prairie Warblers differ from their northern cousins both morphologically – they are larger and have longer patches of white on the outermost 3 retrices (Buerkle 2000) – and behaviorally, in that they are non-migratory. The recorded range of Florida Prairie Warbler extends further north than any of the other mangrove specialists, from Pasco County on the Gulf side to Volusia County on the Atlantic coast (Robertson and Woolfenden 1992, Stevenson and Anderson 1994). Nesting occurs primarily, but perhaps not exclusively, in coastal mangrove forests (Stevenson and Anderson 1994, Prather and Cruz 1995).

Florida Prairie Warblers were by far the most common and widespread mangrove specialist in our surveys. They occurred in every region of the study area without any apparent regard for variation in the structure of the mangroves. They were as common in the tall mixed forests of the Ten Thousand Islands region as they were on the small islands of Florida Bay or in the short-statured red mangrove shrublands of the lower Keys. The only areas where they were relatively uncommon were inland sites near the marsh/mangrove ecotone.

**Eastern Towhee.** Eastern Towhees are a relatively common species in a variety of upland forests in Florida (Stevenson and Anderson 1994). Robertson and Kushlan (1974), citing unpublished observations, suggested a small population might exist in the mangroves of the southwest coast. Our data support this observation. We detected a relatively small number of individuals of this species, but all 55 detections came at the same 9 points in Hell’s Bay, Everglades National Park. Multiple individuals were present at each of these points in every year of the study, suggesting the presence of a very small but persistent breeding population. Eastern Towhees are a common breeding resident of the pine rocklands at Long Pine Key, Everglades National Park, which is <25 km distant. Although the short red
mangroves that predominate in Hell’s Bay may be structurally similar to the shrubby forest edges occupied elsewhere by this species, the presence of persistent stand water would seem to pose a challenge for a ground-foraging species like Eastern Towhee. Additional observations on habitat use and behavior in this population would be of great interest.

**Northern Cardinal.** In Florida, Northern Cardinals are common throughout the state, including the Keys (Robinson and Woolfenden 1992). In the present study, Northern Cardinal was the second most abundant and frequently encountered species in the mangroves. Northern Cardinals inhabited almost every region of the study area and showed no obvious preference for one vegetation type over another. The islands of Florida Bay most distant from the mainland were the only areas where we did not detect any individuals.

**Red-winged Blackbird.** A gregarious species of marsh, prairie, and forest edge, Red-winged Blackbirds are abundant in suitable habitat throughout Florida (Stevenson and Anderson 1994). We found Red-winged Blackbirds at high densities when they occurred, which is to be expected given the semi-colonial nesting habits of this species. Densities were greatest in the Keys and on the islands of Florida Bay, all of which were inhabited save for Big Key. The open, prairie-like depressions at the center of many of the islands in Florida Bay apparently offer attractive breeding habitat for this species. Red-winged Blackbirds were far less common on the mainland, reaching intermediate densities at the most inland sites and in the narrow forests along Biscayne Bay. They were absent from the extensive forests along the southwest coast and much of the Ten Thousand Islands region, with the exception of a few points at the ecotone between freshwater marsh and the mangroves. The use of mangroves by Red-winged Blackbirds appears to be restricted to the drier, shorter, and more open forests and shrublands of the Keys and to areas with extensive forest edge, especially where freshwater marsh or open prairie abuts mangrove forest. Anecdotal evidence in support of this pattern is evident from the J.N. “Ding” Darling National Wildlife Refuge in Sanibel, where a once robust breeding population disappeared after a large area of freshwater marsh was allowed to revert to mangrove forest (Joyce Palmer, USFWS, personal communication).

**Common Grackle.** Within their range, Common Grackles breed nearly anywhere that offers suitable nest sites, from the boreal forest of northern Canada through the pine forests and suburban plantings of south Florida. Although not noted in Robertson and Kushlan (1974) as a resident of mangroves, we
found Common Grackles throughout the study area, albeit at low densities and with a rather patchy distribution. Detections of Common Grackles were concentrated in the lower Keys and on the islands of Florida Bay, presumably because mangroves in these areas are patchier and offer more edge habitat. As with Red-winged Blackbirds, Common Grackles were not detected in the large, extensive mangrove forests of the southwest coast.

**Brown-headed Cowbird.** Brown-headed Cowbirds are a recent addition to the avifauna of Florida, and since their arrival have spread southward throughout the peninsula (Robertson and Woolfenden 1992, Woolfenden and Robertson 2006). Breeding has not been confirmed on the Keys (Robertson and Woolfenden 1992). We detected relatively few Brown-headed Cowbirds, although they were spread over most of the study area. Six detections occurred at 5 different points along Biscayne Bay. Single detections were noted at Rowdy Bend; Clive, Frank, and Palm keys in Florida Bay; Turtle and Tripod keys in the Ten Thousand Islands region; and, perhaps most significantly, at 2 points on north Key Largo near Card Sound.