Shifting Foundations and Metrics for Golden-Cheeked Warbler Recovery

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ABSTRACT Using the golden-cheeked warbler (*Setophaga chrysoparia*) as a case study, this paper discusses what lessons can be learned from the process of the emergency listing and subsequent development of the recovery plan. Are the metrics for recovery in the current warbler plan appropriate, including population size and distribution (recovery units), migration corridors, and wintering habitat? In other words, what happened, what can we learn, and what should happen (in general) in the future for development of such plans? We discuss the number of recovery units required for species persistence and estimate the number of male warblers in protected areas across the breeding range of the species, using newly published density estimates. We also discuss future monitoring strategies to estimate warbler population trends and dispersal rates. © 2012 The Wildlife Society.

KEY WORDS carrying capacity, density estimation, endangered species, golden-cheeked warbler, occupancy models, population viability analysis, recovery plan, recovery units, *Setophaga chrysoparia*.

SPECIES LISTING
The golden-cheeked warbler (*Setophaga chrysoparia*; formerly *Dendroica chrysoparia*; henceforth, warbler) was listed as federally endangered by the U.S. Fish and Wildlife Service (USFWS) in an emergency listing in 1990 (USFWS 1990). The primary threat was thought to be habitat loss and the limited breeding range of the species (i.e., central TX, USA; see Fig. 1A), but other threats were also given in the emergency listing (e.g., nest predation and nest parasitism).

RECOVERY PLAN
The recovery plan for the warbler describes the basic biology of the species, various threats, and recovery criteria that must be met in each recovery unit before the species can be down-listed (USFWS 1992). The recovery plan is currently being revised by the USFWS warbler recovery team. The recovery criteria for the current recovery plan are summarized below, copied from the Executive Summary of USFWS (1992:iv):

The warbler will be considered for delisting when (1) sufficient breeding habitat has been protected to ensure the continued existence of at least one viable, self-sustaining population in each of 8 regions outlined in the plan, (2) the potential for gene flow exists across regions between demographically self-sustaining populations needed for long-term viability, (3) sufficient and sustainable non-breeding habitat exists to support the breeding populations, (4) all existing warbler populations on public lands are protected and managed to ensure their continued existence, and (5) all of these criteria have been met for 10 consecutive years.

CONCERNS WITH METRICS OF THE RECOVERY PLAN
One of our concerns with the recovery plan (USFWS 1992) is the need for 8 recovery units or regions for the warbler, spanning the breeding range of the species in central Texas (Fig. 1). The authors were being conservative at that time because they did not know anything about genetic heterogeneity across the breeding range for the species, but we now know that there is little genetic differentiation across the range (Lindsay et al. 2008, but see Athrey et al. 2011). The original 8 recovery units were determined based on ecological and watershed characteristics (USFWS 1992). If these units are maintained in the future (there is currently no warbler-related biological basis for the boundaries), it would be better to have easily delineated boundaries, such as county lines, because there appears to be confusion on where the boundaries are for those units presented in USFWS (1992), it is difficult to accurately delineate recovery unit boundaries in commonly used spatial programs, such as ArcGIS and ERDAS IMAGINE, and topics such as population and/or urbanization changes often come in a format by county.

Population viability analyses conducted to date (USFWS 1996, Alldredge et al. 2004) imply that a viable population (i.e., one with a low probability of extinction across some time frame, e.g., 100 yr) of the warbler requires a carrying capacity, \(K\), of 3,000 breeding pairs over a 100-year time frame. This analysis assumes that a ceiling model was used.
for the population viability analysis, simulating a variety of 

$K$s, along with estimates of demographic parameters (i.e., 

annual survival and productivity). Bear in mind that a value 

of $K$ for the warbler implies that we need to provide good-

quality breeding habitat to support $K$ pairs, but not all of that 

habitat will necessarily be occupied in any given year. In fact, 

the average population size of a viable population would 

average around 1,000 pairs or fewer in any given year, 

even with $K = 3,000$ for that population (USFWS 1996). 

Assuming adequate dispersal exists within a single popula-

tion, a low probability of extinction (0.04 extinctions/100 yr) 

was estimated for a single population with a $K$ of 3,000 

breeding pairs (USFWS 1996).

We argue that the number of recovery units should 

be reduced (assuming dispersal within a unit is adequate 

for young warblers to sufficiently colonize all available 

habitat patches), especially because of the sparseness of hab-

itat in the far northern (regions 1 and 2) and far western 

(regions 4 and 7) portions of the range. Based on the goal of 

the current recovery plan of maintaining a viable population 

in each unit, the calculation of the probability of species 

persistence for 3 populations as delineated by recovery 

units is: Probability (persistence of $\geq 1$ recovery unit) = 

$1 - \text{Probability (extinction)} = 1 - (0.04)^3 = 0.999936$, where 

0.04 is the probability of extinction for a single population 

of warblers with a carrying capacity of 3,000 breeding pairs
(USFWS 1996). This calculation assumes the 3 populations are independent and results in a 99.9936% chance of the species persisting over 100 years if enough good-quality breeding habitat is protected to support 9,000 (i.e., 3 × 3,000) breeding pairs. More viable populations (e.g., 8) would result in an even higher probability of persistence, but if funding for future monitoring and protection of habitat is limited, then it would be adequate in terms of the goals of the current recovery plan to have fewer than 8 populations. Another problem with the current recovery plan concerns the statement: “If no population in a given region is a viable population by itself, then there should be at least one population in the region that is a) large enough to be demographically self-sustaining (though it can be dependent on its connection to other populations to be genetically viable) and b) has the potential for gene flow to be maintained between the population and at least one other self-sustaining population so that genetic viability is provided for” (USFWS 1992:37). This statement implies that scientists and managers will need detailed knowledge of dispersal rates in order to determine whether a non-viable population could be maintained with dispersal from a nearby viable one, but these rates may never be estimated adequately for this species to justify including mention of it in a recovery criterion. Dispersal could (possibly) be estimated with a well-designed capture-recapture study that spans the breeding range of the species (although such an intensive study would probably be very expensive). Autologistic dynamic occupancy models might also be used to provide information about connectivity (Bled et al. 2011). Regardless, it would be better to remove this criterion concerning dispersal from the recovery plan and reduce the number of recovery units or viable populations necessary, so as not to depend on dispersal rates that we may not be able to estimate. Future genetic studies may offer some way to estimate dispersal; to date, however, such studies do not produce precise estimates of dispersal, but only provide evidence that some dispersal is occurring (Rouset 2001, Koons 2010, Wen et al. 2011). Such studies also tend to be expensive.

**ABUNDANCE AS A METRIC FOR RECOVERY**

Abundance or density (no./unit area) is integral to evaluating recovery of endangered species (e.g., snail kite [*Rostrhamus sociabilis*], Martin et al. 2007; Kirtland’s warbler [*Setophaga kirtlandii*], Probst et al. 2005). Yet, it has long been known that density estimates in and of themselves are insufficient to confirm recovery of an endangered (or any other) species (Van Horne 1983, Vickery et al. 1992). Density data, per se, do not necessarily inform managers or scientists about the vital rates that cause spatial and temporal changes in density, nor do density data offer insight into the availability of habitat and features of habitat that provide food, security, and nesting sites. In step with this thinking, the listing of the warbler as endangered was driven by concerns about restricted breeding range and habitat loss that could result in low abundance and increase the risk of species extinction (Wahl et al. 1990). After the species was listed as endangered in 1990, there were programs initiated at Fort Hood Military Reservation and at Balcones Canyonlands Preserve to, in part, obtain reliable (precise and unbiased) density estimates (e.g., City of Austin [COA] 2011). One of the outcomes of that work was the finding that in habitat with <75% canopy cover of older Ashe juniper (*Juniperus ashei*, which has bark that can be stripped by warblers and used as nesting material) and oaks (*Quercus* spp.), there were low densities of territorial males (range = 0.06–0.20/ha) and low productivity of fledglings (0.0–0.5/nest) versus in habitat with >75% canopy cover of juniper and oak (density of territorial M = 0.12–0.59; fledgling productivity = 1.6–4.0; COA 2011). In this case, density of territorial male warblers may provide information about their productivity.

Given that density of territorial males (hereafter, density) has potential to inform about habitat quality and, therefore, may be useful to a recovery agenda, what are the techniques that have been used for estimating density of warblers that account for imperfect detection (sensu Nichols 1992)? Recently, distance estimators and binomial-mixture models were used to estimate density when count data were collected in a point-sampling framework (Peak 2011, Hunt et al. 2012). The estimates, made at sample-unit plot sizes of 40.5 ha and 250 ha, were then compared with spot-mapped estimates of density at the same sites. The estimates obtained from distance estimators were close to spot-mapped estimates, whereas estimates from binomial-mixture models were consistently greater than spot-mapped estimates. Common to the designs of both studies was treatment of spot-mapped estimates of density as the standard for accurately estimated density; this view is held as valid by many population biologists when male animals can be uniquely identified (which usually requires marked birds) and when density estimates are not susceptible to being an artifact of an inadequate survey effort (e.g., person-hr/ha; Verner and Ritter 1988, Verner and Milne 1990, Bibby et al. 1992).

An estimate of male abundance across the entire breeding range was recently reported by Morrison et al. (2010) and Mathewson et al. (2012), which is a commendable effort for 2 reasons: 1) no study has explicitly been designed and executed to estimate abundance across the breeding range for warblers; and 2) a range-wide abundance estimate should be useful for guiding the recovery agenda for the species. Critical to population estimation is the set of assumptions for how density is estimated (Johnson 2008). Essentially, the range-wide abundance estimate was obtained from the relationship between predicted occupancy and patch-specific counts (singing M/ha). The count data were collected using a point-sampling framework in patches of juniper–oak where the patches were randomly selected from throughout the entire breeding range and the sample points were randomly placed in patches. Observers remained at points for 5 min and recorded the number of males that were detected by sound. Occupancy (≥1 M) was measured by paired observers in patches and was predicted from (i.e., a function of) patch size, landscape composition (percent of juniper–oak habitat within 400 m of a given pixel), the statistical interaction...
between patch size and landscape composition, and additional unexplained spatial variation in occupancy that was captured by spatial random effects (Collier et al. 2012). The predicted occupancy had high accuracy (Collier et al. 2012: fig. 3). Range-wide abundance was estimated at 263,339 males (95% CI = 223,927–302,620; Mathewson et al. 2012). This estimate is higher than previous estimates (Mathewson et al. 2012: table 1). Prior predictions of range-wide abundance were the product of density estimates from a very small part of the breeding range and the estimated amount of range-wide breeding habitat.

The range-wide abundance estimate is founded in the relationship between predicted occupancy and counts. The model used to predict occupancy, a model with its own set of predictors and random effects that were listed above, was selected from a set of models that either had no predictors (null model) or predictors of patch size (linear and quadratic) and landscape composition (parameters estimated in models ranged from 1 to 4; Mathewson et al. 2012: table 2). Perhaps in the Akaike Information Criterion model-selection analysis, the predicted patch occupancy model should have been penalized with more parameter and random effect estimates than were reported in table 2 (which lists two). Nevertheless, the relationship between predicted occupancy of patches and counts in these patches was not strong (change in deviances between null model and the selected patch-occupancy model were 39.14, also see Morrison et al. 2010: figs. 3–5, 3–6, 3–7). Evidently, there is much about the occupancy–abundance relationship for warblers that is unknown. There were also discrepancies between the estimate in the peer-reviewed article (roughly 40,000 more M than in Mathewson et al. 2012) and the estimate provided in the report (Morrison et al. 2010: figs. 3–10). The discrepancy is due to the report using count data collected during 1 year, whereas the peer-reviewed article used count data collected during 2 consecutive years; in addition, different underlying models were used in the 2 analyses (H. Mathewson, Institute of Renewable Natural Resources, Texas A&M University, personal communication). Furthermore, the count data may be susceptible to ≥3 sources of error. The true population sizes may have changed between years, which could influence precision of estimates (Nichols 1992). However, it is unlikely that true population sizes change extensively between 2 consecutive years (COA 2011, Hunt et al. 2012). Ignoring year effects, if they are important, may yield biased estimates of model parameters (J. Nichols, US Geological Survey, Patuxent Wildlife Research Center, personal communication), although this oversight probably would not lead to markedly different estimates of numbers of warblers. We sometimes include more detail in models than we happen to be interested in, in order to obtain useful estimates of parameters that do interest us. If nothing else, including year effects in models in an Akaike Information Criterion model-selection analysis could be used to assess whether collecting count data during 2 years biased abundance estimates.

Another source of error is that count data are notorious for underestimating true abundance, and the extent to which count data underestimate population size typically displays spatial and temporal variation (Williams et al. 2002). Mathewson et al. (2012) purposefully chose to attempt to underestimate density to obtain a conservative (biased low) range–wide abundance estimate. Yet, the extent to which the count underestimates true patch–specific abundance is unlikely to be constant across all patches that were measured (Gates 1966, Williams et al. 2002, Martin et al. 2007, Hunt et al. 2012).

Finally, the survey protocol of Mathewson et al. (2012) was set up to avoid multiple counts of individual males. However, it is conceivable that the count data could have multiple counts of the same individuals, in which case an overestimate of abundance could have occurred. During the time that surveyors recorded the number of singing males at sample points (5 min), singing males might have traveled extensively and made it difficult to distinguish multiple males from a single male when surveyors only detected birds by sound (Peak 2011, Hunt et al. 2012).

The range-wide male abundance estimate of Mathewson et al. (2012) is eye-opening in that warbler abundance is purported to be much greater than previously supposed. It might be informative to recovery efforts if density estimates from Mathewson et al. (2012) were compared with spot-mapped estimates obtained by biologists with Fort Hood and the Balcones Canyonlands Preserve. At these locations, there were multiple plots in a number of patches where densities were estimated from marked birds in the same years during which data were collected for the analyses reported in Mathewson et al. (2012).

ADDITIONAL METRICS FOR RECOVERY

Conserving warbler habitat on protected areas is a high priority for recovery efforts, given that habitat loss is cited as the greatest threat to warblers (USFWS 1992) and range-wide breeding habitat loss is occurring on private land through urban development and land-use change (Groce et al. 2010). However, determining how much protected habitat is needed to sustain viable populations in each recovery region is not an easy task. Using satellite imagery, a number of habitat suitability maps have been created using various criteria to delineate potential warbler breeding habitat (reviewed in Morrison et al. 2010). By linking the habitat suitability map created by Diamond and True (1999) with demographic estimates, Alldredge et al. (2004) were able to simulate population viability assuming a patchy population structure for the northern region of the warbler breeding range. As stated earlier, their results suggested that each population needed to provide habitat to support a minimum of approximately 3,000 breeding pairs (i.e., carrying capacity, K) to be considered viable, which supported what was determined by USFWS (1996). However, it is important to note that we are currently in the process of building upon the work of Alldredge et al. (2004) using updated demographic parameter estimates and simulating population viability at a range-wide scale (which may yield lower estimates of breeding pairs needed for population persistence, depending on dispersal). Nevertheless, it was not until recently that range-
wide density estimates were available to estimate the number of warblers on protected lands (Mathewson et al. 2012). Using the “new model C live oak as deciduous” habitat suitability model provided by Diamond et al. (2010) and the shapefile used to estimate patch-specific warbler occupancy probabilities by Collier et al. (2012), we estimated the total amount of suitable breeding habitat and number of male warblers on protected lands (2012 conservation and recreation lands and 2010 Department of Defense lands) for each of the regions delineated by Mathewson et al. (2012) for range-wide density estimates (Fig. 1B; Table 1). The warbler suitability model provided by Diamond et al. (2010) is a 10-m × 10-m-resolution raster file that treats oaks as deciduous (e.g., habitat is not suitable if oak is >100 m from juniper or mixed juniper). The shapefile provided by Collier et al. (2012) is a 30-m × 30-m-resolution layer that identifies woodland patches (see Collier et al. 2010) within the warbler’s breeding range. Both habitat classification scenarios are considered acceptable methods to delineate warbler breeding habitat. Population estimates were then calculated by multiplying the density estimates (estimates reported in Mathewson et al. 2012) for each region; North region: low = 0.1 M/ha, $\pi = 0.15$ M/ha, high = 0.17 M/ha; Central region: low = 0.15 M/ha, $\pi = 0.19$ M/ha, high = 0.23 M/ha; South region: low = 0.26 M/ha, $\pi = 0.32$ M/ha, high = 0.37 M/ha) by the total amount of protected suitable habitat (identified by both Diamond et al. (2010) and Collier et al. (2012)). We also did this calculation for the 8 recovery units identified in USFWS (1992) [Fig. 1C; Table 2]. The numbers of male warblers within protected areas estimated using the 2 habitat suitability maps did not differ greatly between the 2 maps (Tables 1 and 2). Even when increasing the amount of protected suitable habitat per recovery unit by separating the entire warbler breeding range into 3 large regions and using the upper confidence bounds for warbler densities (Table 1), the southern region does not meet the minimum number of male warblers needed to be considered viable under both warbler habitat delineation scenarios (estimated M in the south region for habitat delineated by Diamond et al. (2010) was 1,657 and for habitat delineated by Collier et al. 2012 was 1,862). Furthermore, when separating the high population estimates between the current 8 recovery units using both warbler habitat delineation scenarios, 5 of the 8 recovery units have substantially fewer birds on protected habitat than the minimum number to be considered viable populations (Table 2). This highlights the need for additional land acquisition to take place to ensure warbler species viability (or else to reduce the no. of recovery units to <8).

If habitat acquisition (converting from private to protected) is pursued, then there will still be a need to develop criteria in order to prioritize habitat for potential acquisitions. Collier et al. (2012) provided predicted patch occupancy estimates of warblers at the range-wide scale that will undoubtedly assist with prioritizing habitat acquisitions and recovery efforts; however, habitat acquisition should not be based on patch occupancy alone. Furthermore, given that recovery efforts are generally constrained by monetary costs, it would be practical that future decision support tools for acquisition of warbler habitat incorporate current and projected land costs.

The use of population viability analyses as a tool for species recovery efforts has grown in popularity (Morris et al. 2002). Yet, estimates of demographic parameters (e.g., survival, reproduction, dispersal rates), which are needed in such analyses, are often not a high priority immediately following the listing of a species. If a population viability analysis is needed at some point in the recovery process for the species, then capture–recapture studies should be initiated as soon as possible, particularly because 3 years of capture–recapture data are needed to estimate one survival rate under the general Cormack–Jolly–Seber model structure (Pollock et al. 1990). Furthermore, ≥2 parameter estimates are required to estimate the temporal variance of a specific parameter, and precise variance estimates likely require >5 years of data (J. Nichols, personal communication).

In the case of the warbler, capture–recapture monitoring programs were initiated in 1991 and 2009, respectively, at Fort Hood and Balcones Canyonlands Preserve. However, survival estimates have previously been restricted to the male segment of the species because of the difficulty in detecting females during the breeding season (USFWS 1996, Alldredge et al. 2004). Currently, data from both recovery sites are being analyzed to calculate more precise estimates of survival. Nevertheless, most of the study areas on Fort Hood were not monitored consistently because of logistical constraints, so direct comparisons of survival rates cannot be made between study areas across all field seasons. Additionally, data to estimate demographic parameters in the southwest portion of the warbler’s breeding range are lacking. It is imperative, but often overlooked,

**Table 1.** Total estimated suitable breeding habitat and estimated population sizes by region of male golden-cheeked warblers on protected land in Texas, USA, using both the “new model C live oak as deciduous” habitat suitability model provided by Diamond et al. (2010) and the shapefile used to estimate patch-specific warbler occupancy probabilities by Collier et al. (2012) per region of the breeding range. Density estimates were provided by Mathewson et al. (2012) for each region of the warbler’s breeding range. Protected lands were 2012 conservation and recreation lands and 2010 Department of Defense lands.

<table>
<thead>
<tr>
<th>Region</th>
<th>Protected habitat (ha)</th>
<th>Population estimate</th>
<th>Protected habitat (ha)</th>
<th>Population estimate</th>
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<tr>
<td></td>
<td>Low</td>
<td>Mean</td>
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<td>11,611</td>
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to understand how demographic parameters vary spatially. This information will reduce biases associated with generalizing information gathered from a very small proportion of the species’ range to make inferences across the species’ range.

Capture–recapture studies provide a means to estimate demographic parameters, but may not be feasible across large spatial scales (Royle 2004). Still, standardized studies that implement techniques such as spot-mapping, occupancy modeling, distance sampling, or mark–recapture distance sampling may be reliable, efficient alternatives to range-wide capture–recapture studies. Currently, both Fort Hood and the Balcones Canyonlands Preserve monitor warblers by estimating warbler territory density via spot-mapping (mapping breeding bird territories within designated plots; Bibby et al. 1992). Alternatively, occupancy modeling integrates presence–absence data within a maximum likelihood framework to estimate patch occupancy and species detection probabilities (MacKenzie et al. 2002), and can incorporate spot-mapping data. Occupancy modeling has proven to be a feasible technique to extract reliable information for multiple rare species, including warblers (Jackson et al. 2006, Watson et al. 2008, Collier et al. 2010, Delaney and Leung 2010, Hunt et al. 2012). As reviewed earlier, Peak (2011) showed that distance sampling provided reasonable estimates of warbler densities when compared with spot-mapping. Laake et al. (2011) proposed the use of mark–recapture distance sampling to monitor warbler populations. Mark–recapture distance sampling integrates distance sampling with the double-observer capture–recapture methodology described by Nichols et al. (2000), and may yield more precise estimates of warbler abundance than what would be estimated by using the 2 methods (distance sampling and double-observer capture–recapture) independently (Laake et al. 2011). All methods seem to provide reliable information concerning population trends when implemented with a proper study design; however, it is important that future methods used to monitor warbler populations try to incorporate the long-term data already collected. Therefore, spot-mapping and occupancy modeling may be better methods to reliably monitor warbler populations across their range.

The warbler migrates north and south between breeding and wintering grounds following the distribution of the pine–oak forests at elevations between 1,100 m and 2,400 m (reviewed in Groce et al. 2010; see also Komar et al. 2011). Warblers spend approximately 53% of the year in the wintering habitat, which is encompassed by Mexico, Guatemala, Honduras, El Salvador, and Nicaragua. During wintering months, warblers select for pine–oak habitat in elevation (Monroe 1968, Rappole et al. 1999). Nevertheless, the availability of pine–oak wintering habitat within suitable elevation and latitudinal ranges is limited in the warbler’s wintering range (Rappole et al. 2000). The recovery plan for the warbler indicated the need for international partnerships for warbler recovery (USFWS 1992); however, no partnership has been created to date. Unless land conservation

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**Table 2.** Total estimated suitable breeding habitat and estimated population sizes per recovery unit (USFWS 1992) of male golden-cheeked warblers on protected land in Texas, USA, using both the “new model C live oak as deciduous” habitat suitability model provided by Diamond et al. (2010) and the shapefile used to estimate patch-specific warbler occupancy probabilities by Collier et al. (2012). Density estimates were provided by Mathewson et al. (2012) for each region of the warbler’s breeding range. Protected lands were 2012 conservation and recreation lands and 2010 Department of Defense lands.

<table>
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<td>232.94 South</td>
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<td>484 559</td>
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<td>44.89 Central</td>
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<td>1,053 1,217</td>
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<td>11,609</td>
<td>15,639 18,370</td>
<td>11,609 15,639 18,370</td>
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actions take place within the warbler’s wintering range and along its migratory route, conservation efforts in the United States may prove ineffective at ensuring warbler species viability.

SUMMARY

Since the warbler was first listed as endangered in 1990, we have learned a great deal about the basic biology of the warbler, about various demographic parameters estimated for the species, range-wide density and abundance, genetic heterogeneity, and population viability. We are hopeful that the warbler can be downlisted sometime in the future, although probably on a case-by-case basis for each recovery unit. Given the large numbers of male warblers estimated by Mathewson et al. (2012) across Texas, there is now much hope for the future of this species.

Concerning the future for development of such recovery plans for other threatened or endangered species, we reiterate the need to begin studies estimating demographic parameters (i.e., survival, productivity, and dispersal rates), occupancy, density, and abundance as soon as possible, given the lengthy nature of such studies and the importance they have in determining the status of the species and possibility of recovery.

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LITERATURE CITED


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