Original Article

Population Size of Island Loggerhead Shrikes on Santa Rosa and Santa Cruz Islands

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ABSTRACT Island loggerhead shrikes (*Lanius ludovicianus anthonyi*) are an endemic, genetically distinct subspecies of loggerhead shrike on California's Santa Rosa, Santa Cruz, and Santa Catalina Islands (USA). This subspecies is listed as a Species of Special Concern by the California Department of Fish and Game and has been petitioned for federal listing under the Endangered Species Act. The combination of suspected low numbers and the possibility of federal listing, prompted us to undertake a study to rigorously estimate the number of remaining individuals on Santa Rosa and Santa Cruz Islands. During the 2009 and 2010 breeding seasons, we surveyed sample units on Santa Rosa and Santa Cruz Islands using a double-observer method with independent observers to estimate joint detection probabilities (p), where we selected units under a stratified random sampling design. We estimated shrike abundance to be 169 in 2009 ($p = 0.476$) and 240 in 2010 ($p = 0.825$) for Santa Rosa Island, and 35 in 2009 ($p = 0.816$) and 42 in 2010 ($p = 0.710$) for Santa Cruz Island. These numbers, especially for Santa Rosa Island, are higher than previously reported but nevertheless are still low. Rapid vegetation change on both islands due to recent removal of nonnative herbivores may threaten the habitat and status of this subspecies and, therefore, we suggest that intensive demographic and habitat use research be initiated immediately to obtain additional information vital for the management of this subspecies. Published 2012. This article is a U.S. Government work and is in the public domain in the USA.

KEY WORDS Channel Islands, Island loggerhead shrike, *Lanius ludovicianus anthonyi*, population estimation, Santa Cruz Island, Santa Rosa Island.

Island loggerhead shrikes (*Lanius ludovicianus anthonyi*; hereafter, shrikes) are an endemic subspecies of loggerhead shrike with breeding populations on 3 of 8 of the California Channel Islands: Santa Rosa Island (SRI), Santa Cruz Island (SCI), and Santa Catalina Island (Collins 2008). In recent genetic analyses of museum specimens, Caballero and Ashley (2011) and Eggert et al. (2004) confirmed that specimens of loggerhead shrikes from the 2 northern islands of SRI and SCI are genetically similar to each other but distinct from adjacent mainland (*L. l. gambeli*) and San Clemente Island loggerhead shrikes (*L. l. mearnsi*), whereas birds from Santa Catalina Island show mixed ancestry among all 3 subspecies.

Anecdotal accounts suggest that shrikes were once common but have decreased in abundance (see references in Collins 2008, Hicks and Walter 2009). Consequently, in recent years several attempts have been made to quantify the abundance of shrikes on SRI and SCI. For example, in 1994 the National Park Service began monitoring shrikes on SRI using variable circular plots systematically placed and stratified by major vegetation types (Super et al. 1991, Coonan et al. 2011), but too few shrikes were detected for this approach to prove useful. From 1991 to 2003 The Nature Conservancy and the University of California Santa Cruz Island Reserve ran a similar program on SCI using fixed-radius plots, and a separate one-season project by Warnock and McMorran (2008) used point counts, line transects, and area searches to estimate shrike abundances on SCI. However, because of the rarity of shrikes on the island and the absence of a probability sampling framework (i.e., every unit in the sampling frame has a known, nonzero probability of being sampled, which allows for unbiased parameter estimates to be made to the entire sampling frame), none of these methods yielded rigorous estimates of shrike abundance. Finally, during the spring of 2006, 22 canyons on SRI were searched for shrikes by a single observer (Dye et al. 2009), and a survey of potential shrike habitat was conducted on SCI the same year (Hicks and Walter 2009). Unfortunately, inferences to unsampled areas were not possible because a probability sampling framework was not used. Moreover, neither Dye et al. (2009) nor Hicks

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and Walter (2009) estimated detection probabilities (i.e., the probability a shrike was detected by an observer given it was present in the survey area), which, in our experience, are usually <1, so abundance estimates were likely negatively biased.

Despite the absence of satisfactory abundance estimates for shrikes, it is believed that in 2006 there were only 15–30 shrikes on SRI and 20–30 shrikes on SCI (Walter 2006), and *L. l. anthonyi* has been listed as a Species of Special Concern by the California Department of Fish and Game. Warnock and McMorran (2008) detected 28 shrikes on SCI, but they did not sample the entire island. A formal request to federally list *L. l. anthonyi* as an endangered subspecies was submitted by Walter (2006) and in the event of listing a formal recovery plan will be required; thus, there is an urgent need to rigorously estimate the number of remaining shrikes on the islands.

In studies seeking to accurately estimate bird abundance 2 major issues must be addressed. First, areas for which abundance estimates are desired are often larger than can be adequately surveyed given available time and personnel resources. Consequently, and because avian abundance typically varies spatially over the area of interest, statistically valid sampling designs (i.e., those based on a probability sample) that allow inferences to the unsampled areas from a sample of plots (or other spatial units) are required. Second, during surveys, some birds that are present may not be detected because some individuals may be obscured by vegetation or topography during the survey, or individuals may not produce cues (e.g., calls or movements) that draw the attention of the surveyor. Consequently, methods that allow detection probabilities to be estimated from survey data must be integrated into the survey design so that counts can be adjusted for individuals not detected (Thompson 2002, Stanley and Skagen 2007).

Our objectives were 1) to obtain an accurate estimate of shrike abundance on SRI and SCI during the 2009 and 2010 breeding seasons, and 2) to compare our estimates with those calculated in other studies.

STUDY AREA

Santa Rosa Island (34°0′N, 120°07′W) and SCI (34°0′N, 199°45′W) are 2 of 8 California Channel Islands in the Pacific Ocean off the coast of Southern California, USA (Fig. 1). Santa Rosa Island was the second largest of the 4 Northern Islands (approx. 21,500 ha), had a peak elevation of 484 m, and was located approximately 53 km southwest of Santa Barbara, California. The mean annual temperature for SRI was 15–17° C, with a January mean monthly temperature range of 5–7° C and a July mean monthly temperature range of 24–28° C. Mean annual precipitation was 30–50 cm/year (Davey et al. 2007). Santa Cruz Island was the largest of the Northern Islands (approx. 25,000 ha), had a peak elevation of 750 m, and was located approximately 41 km south of Santa Barbara, California. Mean monthly temperatures for the central valley on SCI ranged from 9.4° C to 17.5° C. The average annual precipitation for the central valley (1904–2000) was 51.5 cm and the yearly totals ranged from 142.6 cm to 161.0 cm (Boyle and Laughrin 2000). Santa Rosa Island and SCI consisted of hills, mountains, and steep canyons, with vegetation cover including coastal grasslands of native perennial bunchgrasses, herbaceous annuals, and nonnative grasses that covered approximately 60% of the island (Schoenherr et al. 1999). Other common vegetation types included coastal sage scrub, coastal bluff scrub, island chaparral, island woodlands, closed-cone pine forest, riparian, and coastal beach and dune vegetation (Junak et al. 1995, Schoenherr et al. 1999).

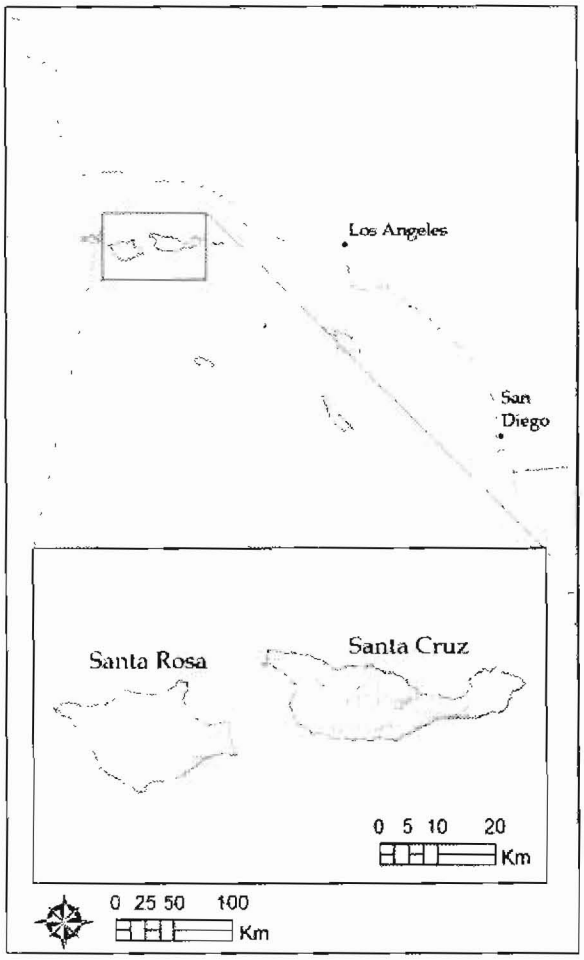

Figure 1. Map of the California Channel Islands, USA, showing the locations of Santa Rosa and Santa Cruz Islands (see inset).

METHODS

Sampling Design

As a first step in establishing our sampling design, we a priori designated certain areas of SRI and SCI as exclusion areas, which were omitted from the sampling frame. Typically, we considered exclusion areas to be too steep to survey, devoid of vegetation (e.g., high-elevation areas), or containing vegetation that was clearly unsuitable, or of very low quality, as breeding habitat for shrikes based on literature from other island and mainland studies (e.g., Gawlik and Bildstein 1995, Michaels and Cully 1998, Lynn et al. 2006*a*,

St-Louis et al. 2010). On Santa Rosa Island the only areas that qualified this way were those with dense lupine scrub (*Lupinus* spp.), and dense stands of pine (*Pinus* spp.) and eucalyptus (*Eucalyptus* spp.). On Santa Cruz Island we excluded areas dominated by dense pine, eucalyptus, and fennel (*Foeniculum vulgare*). As a consequence, 5,626 ha of SRI were considered exclusion areas, leaving 15,857 ha or 74% of SRI as part of the sampling frame (Fig. 2A), and 6,264 ha of SCI were excluded, leaving 18,824 ha or 75% as part of the of the sampling frame (Fig. 2B). Abundance estimates in this study apply only to nonexclusion areas.

Once we specified exclusion areas, we partitioned the remaining nonexclusion areas of SRI and SCI into spatial sampling units using a watershed delineation model (part of the Watershed Delineation Toolbox available for download from ESRI; Environmental Systems Research Institute, Inc., Redlands, CA; http://support.esri.com/index.cfm?fa=downloads.geoprocessing.filteredGateway&GPID=16). The watershed delineation tool uses a digital elevation model to calculate stream flow and offers the option of adjusting the sizes of the resulting watersheds. In 2009, our target watershed size was 324 ha, which was the area we thought we could effectively survey in 1 day (based on similar surveys conducted on San Clemente Island; Lynn et al. 2006*a*). In 2010, we reduced the target size of the watersheds to 192 ha, because in 2009 observers found it difficult to cover the entire unit. In both years and on both islands, we found it necessary in some cases to manually split or aggregate watersheds to

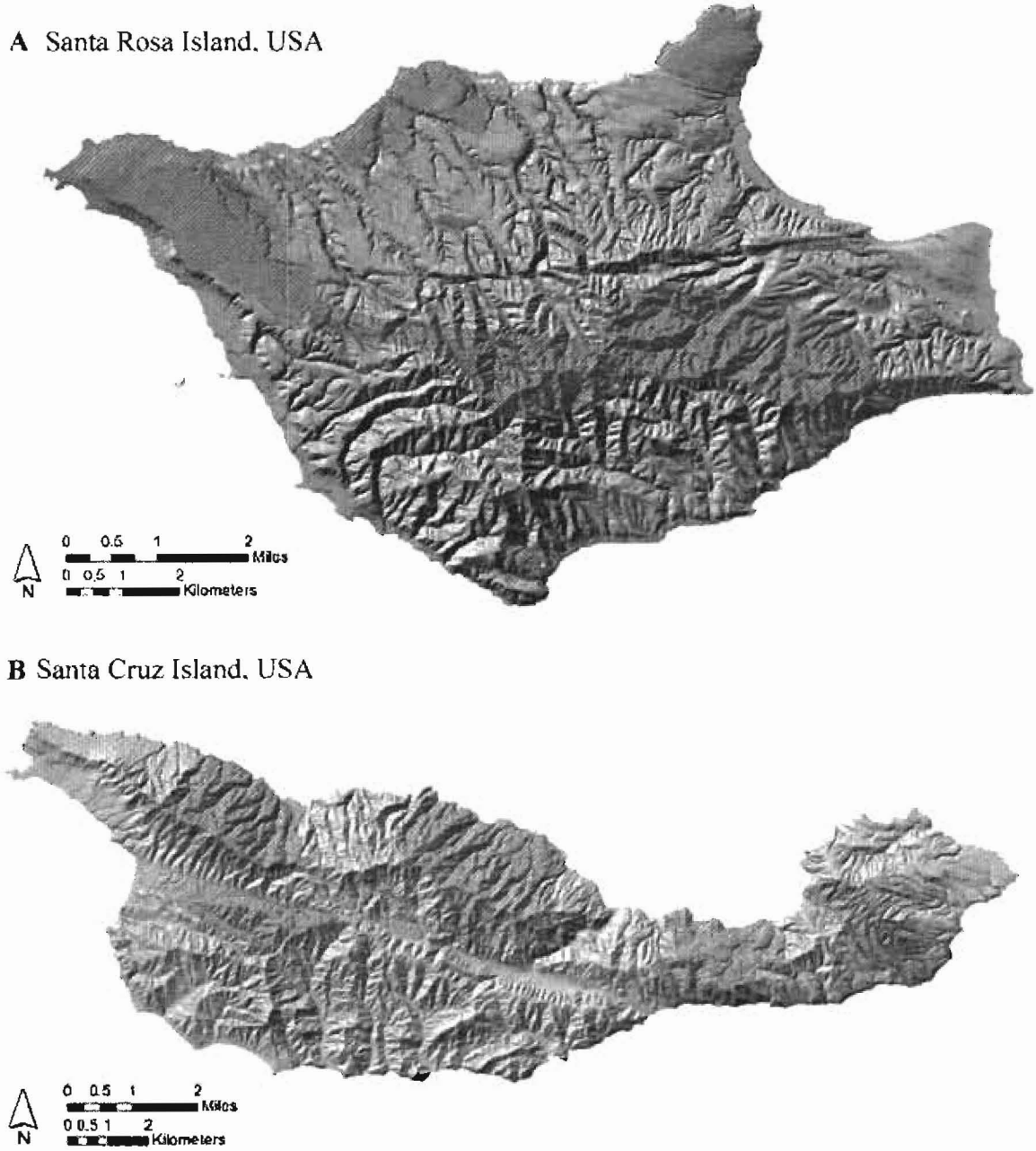

Figure 2. Maps of (A) Santa Rosa Island and (B) Santa Cruz Island, California, USA, showing the exclusion areas (red) and nonexclusion areas (gray) determined prior to sampling. Abundance estimates obtained during the 2009 and 2010 breeding seasons in this paper apply only to the nonexclusion areas.

achieve our target size. The watersheds resulting from this process comprised the set of sample units from which we drew our sample.

In this study, we sampled without replacement under a stratified random design with 2 strata. Stratum 1 consisted of sample units for which there was ≥1 historical record of a shrike sighting during the breeding season, which we defined as the period from 1 March to 30 June (based on breeding season dates for San Clemente loggerhead shrikes; Lynn et al. 2006*a*). Stratum 2 consisted of units for which no historical records of a shrike sighting during the breeding season were available. We reasoned that because the historical data on shrikes had been accumulated over several years, units without historical sightings probably had no breeding shrikes. Thus, units within strata would be expected to be relatively homogeneous with respect to shrikes detected and this would help minimize variance. We did not stratify on the basis of vegetation type because vegetation patterns were not conducive to partitioning the island into spatial sampling units of appropriate size and configuration for our surveys.

For SRI, in 2009 we assigned 33 units (approx. 9,402 ha) to Stratum 1 and 24 units (approx. 6,454 ha) to Stratum 2. We sampled 2,961 ha from Stratum 1 (approx. 31% of the area) and 1,941 ha from Stratum 2 (approx. 30% of the area). In 2010, we assigned 36 units (approx. 7,120 ha) to Stratum 1 and 48 units (approx. 8,736 ha) to Stratum 2. We sampled 3,843 ha from Stratum 1 (approx. 54% of the area) and 2,747 ha from Stratum 2 (approx. 31% of the area). For SCI, in 2009 we assigned 15 units (approx. 4,304 ha) to Stratum 1 and 56 units (approx. 14,520 ha) to Stratum 2. We sampled 3,447 ha from Stratum 1 (approx. 80% of the area) and 6,220 ha from Stratum 2 (approx. 43% of the area). In 2010, we assigned 17 units (approx. 2,826 ha) to Stratum 1 and 97 units (approx. 15,998 ha) to Stratum 2. We sampled 2,653 ha from Stratum 1 (approx. 94% of the area) and 6,320 ha from Stratum 2 (approx. 40% of the area).

Data Collection

We surveyed SRI between 23 and 27 April, 2009, and between 1 and 20 April, 2010. In both 2009 and 2010 we surveyed SCI between 6 and 24 May.

Obtaining data sufficient to estimate shrike abundance on SRI and SCI is particularly challenging for 3 reasons: 1) the area of the islands is large and many areas are remote and difficult to access, 2) the islands are topographically complex and heterogeneous, and 3) shrikes occur in very low numbers. One consequence of these factors, in our collective experience, is that point-count-based methods do poorly. This is in part because only small areas are covered by any particular point and relatively few random points can be sampled (due to remoteness), but mostly because very few shrikes are detected (Warnock and McMorran 2008). Thus, point-count methods yield insufficient data to estimate detection probabilities using distance methods (Buckland et al. 1993), double-observer methods (Nichols et al. 2000, Alldredge et al. 2006), or removal methods (Farnsworth et al. 2002). A second consequence of these factors, especially factor 2, is that transect-based methods are nearly impossible to implement because of the many steep canyons, cliffs, and mountains intersected by random transects. This problem, in combination with few shrike detections, means distance-based methods are not practical.

In our experience, and as concluded by others (Warnock and McMorran 2008), the most effective survey method for shrikes on SRI and SCI is an area search method, where some spatial unit is specified and is then searched in its entirety by one or more observers. The advantages of this method are that a large area can be searched on foot in 1 day, and topographic hazards like steep canyons can be avoided but still visually inspected for shrikes. Of course, there still remains the issue of estimating detection probabilities. Because the observer is not sampling discrete points but is instead traversing a continuous route that cannot be prespecified (due to unknown hazards or impenetrable vegetation), neither distance-based nor removal-based methods are applicable. This leaves double-observer methods, with either dependent or independent observers, as possibilities. We ruled out the use of dependent observers because it did not seem likely that the primary observer would not be influenced by cues from the secondary observer. Thus, we adapted the double-observer method with independent observers (Alldredge et al. 2006) to our area search method.

Under the double-observer method, paired observers with intermediate to advanced experience in the counting and identifying of bird species by sight and sound were provided topographic maps of their survey unit. We began surveys at approximately 0900 hours (range = 0640–1200 hours) and generally completed them by approximately 1600 hours (range = 1000–1925 hours). At the onset of the survey, observers recorded information on general weather conditions (i.e., temp, wind speed, precipitation, cloud cover) and jointly planned a route through the unit that would allow them to visually inspect all areas of the unit. During the survey, observers hiked the preplanned route separately, with the second observer trailing the first observer by ≥500 m or 30 min. Observers were instructed to avoid any communication (visual or via radio) regarding shrike detections during the survey. When a shrike was detected by an observer they recorded the method of detection (visual, vocalization, both), their Global Positioning System (GPS) location at the time of detection, the vegetation type in which the bird was detected, and they plotted the location of the bird on their topographic map. We did not record shrikes detected outside the unit boundaries, and we assumed closed populations within units. Likewise, we assumed constant observer-specific detection probabilities for the duration of the survey.

For the 2009 SRI surveys, observers assigned their shrike sightings a subjective certainty code (intended to reflect their degree of certainty that the bird they observed was indeed a shrike) of 1–5, where 1 denoted a certainty of 1–40%, 2 a certainty of 41–60%, 3 a certainty of 61–84%, 4 a certainty of 85–99%, and 5 a certainty of 100%. Low certainties (e.g., categories 1–3) were sometimes obtained when a bird was observed >100 m away, often in flight or in high wind, making the detection questionable. The certainty code,

therefore, allowed for exclusion of these records during analysis. We could not distinguish between certainties of 85% and 99% within category 4, and furthermore did not believe that a certainty of <85% constituted a reliable detection; therefore, we used an 85% certainty level as our threshold for censoring unreliable data. We applied the same rule to SRI in 2010 and SCI in both years, although in those surveys we recorded percent certainty rather than the certainty codes.

Once surveys were completed, observers recorded ending weather conditions, shaded their topographic map to show areas they were not able to visually survey, and reconciled the shrikes they detected with each other. The goal of reconciliation was to compare the GPS locations and timing of detections so as to determine which shrikes were detected by only one observer or by both observers, and to summarize the data in the form of a capture history (i.e., a history of {10} means a shrike was detected by observer A but not observer B, {01} means a shrike was detected by observer B but not observer A, {11} means both observers detected the shrike). Based on previous surveys shrikes appeared to be rare and at low densities on SRI and SCI, and they are territorial during the breeding season; thus, we assume that observers accurately reconciled their sightings. We note that similar island-wide surveys conducted by the U.S. Department of the Navy, in which observers reconcile detections immediately following the survey, have been used successfully for the San Clemente Island loggerhead shrike for many years (M. Booker, U.S. Navy, personal communication).

Abundance Estimation

Prior to analyzing shrike detection data we determined the proportion of each sample unit surveyed by both observers (in some cases weather, topography, or time constraints prevented portions of a unit from being surveyed): or, more specifically, the intersection of the set of points visually surveyed by each observer. In cases where one observer surveyed an area but the other did not, we discarded any shrike detections in that area. Discarding detections was necessary because the double-observer method requires that a shrike be available for detection by each observer, independently, which is clearly not the case if one observer did not survey the area in which a shrike was detected by the other observer.

We estimated detection probabilities for shrikes under the independent-observer method using the closed-population model M_0 from Otis et al. (1978), as implemented by Huggins (1991). We used this model because 1) the number of shrikes detected was generally low; 2) for most sample units no shrikes were detected; and 3) most observers sampled only 2 sample units. Thus, we simply did not have enough data to support more complex models. If we denote shrike detection probabilities on unit i for observers 1 and 2 by p_{i1} and p_{i2}, we get for the estimated joint detection probability on the unit $\hat{p}_i = 1 - (1 - \hat{p}_{i1})(1 - \hat{p}_{i2})$. However, under model M_0 it is assumed that detection probabilities are homogeneous within and among observers. Thus, $p_{i1} = p_{i2} = p^*$ and the estimated joint detection probability simplifies to $\hat{p} = 1 - (1 - \hat{p}^*)^2$. Consequently, we estimated shrike abundance for the ith sample unit (n_i) as $\hat{n}_i = c_i/\hat{p}$, where c_i is the number of distinct shrikes counted in unit i. We obtained numerical estimates for p^* and its variance using Program MARK (White and Burnham 1999).

We obtained abundance and variance estimates for the sample units using ratio estimators and other standard formulae for stratified random sampling (Cochran 1977, Scheaffer et al. 1986). If we let N denote the number of shrikes in the sampling frame, then our ratio estimate of N is

$$\hat{N} = \sum_{h=1}^{2} \left(\frac{\sum_{i=1}^{m_h} \hat{n}_{hi}}{\sum_{i=1}^{m_h} A_{hi}} \right) A_h, \tag{1}$$

where $\hat{n}_{hi}$ is the estimated abundance of shrikes in the ith unit of stratum h ($h = 1, 2$), A_{hi} is the total sampled area in ha of unit i of stratum h, A_h is the total area in ha of stratum h, and m_h is the number of units sampled in stratum h.

To obtain our estimated variance for $\hat{N}$, we start by rewriting $\hat{n}_{hi}$ in (Equation 1) as $\hat{n}_{hi} = c_{hi}/\hat{p}_h$. Substituting this into (Equation 1), we get $\hat{N} = \sum_{h=1}^{2} \left(\sum_{i=1}^{m_h} (c_{hi}/\hat{p}_h) / \sum_{i=1}^{m_h} A_{hi} \right) A_h$, which can be rewritten as

$$\hat{N} = \sum_{h=1}^{2} \frac{1}{\hat{p}_h} \left(\frac{\sum_{i=1}^{m_h} c_{hi}}{\sum_{i=1}^{m_h} A_{hi}} \right) A_h = \sum_{h=1}^{2} \frac{r_h A_h}{\hat{p}_h}, \tag{2}$$

where $r_h = \left(\sum_{i=1}^{m_h} c_{hi} / \sum_{i=1}^{m_h} A_{hi} \right)$. The estimated variance of $\hat{N}$, which we denote $\hat{V}(\hat{N})$, is obtained by applying the delta method (Seber 1982) to the right-hand side of Equation (2), which, under an independence assumption, yields

$$\hat{V}(\hat{N}) = \sum_{h=1}^{2} A_h^2 \left[\frac{r_h^2}{\hat{p}_h^4} \hat{V}(\hat{p}_h) + \frac{1}{\hat{p}_h^2} \hat{V}(r_h) \right], \tag{3}$$

where $\hat{V}(\hat{p}_h)$ is obtained from Program MARK and $\hat{V}(r_h) = (1 - f_h) \frac{1}{A_h^2} \left[\sum_{i=1}^{m_h} (c_{hi} - r_h A_{hi})^2 / m_h(m_h - 1) \right]$ (Scheaffer et al. 1986:131). In this latter expression, $(1 - f_h)$ is the finite population correction factor for stratum h, and $f_h = \sum_{i=1}^{m_h} A_{hi}/A_h$. The estimated standard error of $\hat{N}$ is $\sqrt{\hat{V}(\hat{N})}$. A 95% confidence interval was computed as $\hat{N} \pm 1.96 \times SE(\hat{N})$.

RESULTS

We detected shrikes throughout the day during surveys. After pooling detection data across islands and years, we found the frequency distribution of detections peaked at 1300 hours (Fig. 3A). However, when we rescaled these frequencies by the number of hours of survey effort during each interval, the peak disappeared; these results suggested that shrike detections were distributed uniformly throughout the day except for peaks at 0800 hours and 1800 hours (Fig. 3B).

On SRI and SCI in 2009 and 2010, the number of shrikes we detected per stratum was generally small and led to

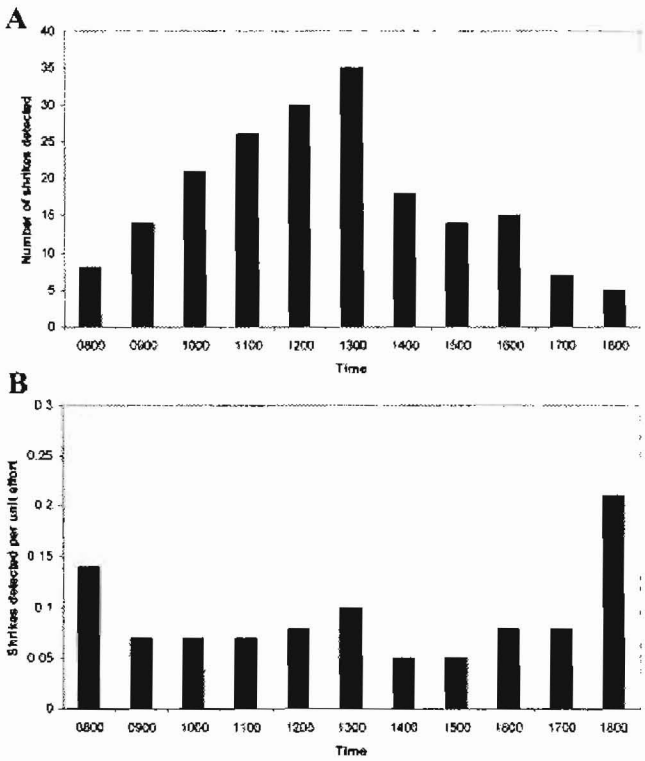

Figure 3. Histograms for (A) the time-specific number of island loggerhead shrikes (*Lanius ludovicianus anthonyi*) detected during surveys, and (B) the same data rescaled by hours of survey effort. We pooled data for Santa Rosa and Santa Cruz Islands, California, USA, over the 2009 and 2010 breeding seasons.

problems with parameter estimation. Hence, within each island × year combination, we pooled data across strata in order to estimate detection probabilities.

Santa Rosa Island

We recorded 29 shrike observations on SRI in 2009. Two of these observations we censored due to low certainty levels and 2 we censored because only one of the observers surveyed the area where the birds were detected, thereby violating a basic assumption of the double-observer method. Of the 25 observations remaining, 4 were detections by both observers and 21 were detections by only one observer. Eighteen observations were from Stratum 1, and 7 were from Stratum 2. Using these data, we estimated a detection probability of 0.276 (SE = 0.109; 95% CI = 0.116–0.526), which yielded a joint detection probability (i.e., the probability of being detected by ≥1 observer) of 0.476 (SE = 0.158; 95% CI = 0.208–0.758) and an abundance estimate of 169.1 shrikes (SE = 53.5; Table 1).

We recorded 81 shrike observations on SRI in 2010, 3 of which we censored due to low certainty levels, leaving 78 observations for abundance estimation; 32 were detections by both observers and 46 were detections by only one observer. Thirty-eight observations were from Stratum 1 and 40 were from Stratum 2. Using these data, we estimated a detection probability of 0.582 (SE = 0.056; 95% CI = 0.470–0.686), which yielded a joint detection probability of 0.825 (SE = 0.047; 95% CI = 0.714–0.899) and an abundance estimate of 239.6 shrikes (SE = 47.4) in 2010 (Table 1).

Table 1. Estimated abundance (95% CI) of island loggerhead shrikes (*Lanius ludovicianus anthonyi*) during the 2009 and 2010 breeding seasons in the nonexclusion areas of Santa Rosa and Santa Cruz Islands, California, USA. Estimates were obtained using a double-observer approach (independent observers) under a stratified random sampling design.

	2009		2010	
Study area	Estimate	95% CI	Estimate	95% CI
Santa Rosa Island	169.1	64.2–274.0	239.6	146.7–332.4
Santa Cruz Island	34.9	16.5–53.4	42.4	23.3–61.5
Pooled estimates	204.0	97.5–310.6	282.0	187.2–376.8

Santa Cruz Island

We recorded 22 shrike observations on SCI in 2009, 7 of which we censored due to low certainty levels, leaving 15 observations for abundance estimation. Of these, 6 were detections by both observers and 9 were detections by only one observer. Six observations were from Stratum 1, and 9 were from Stratum 2. Using these data, we estimated a detection probability of 0.571 (SE = 0.129; 95% CI = 0.322–0.789), which yielded a joint detection probability of 0.816 (SE = 0.111; 95% CI = 0.511–0.950) and an abundance estimate of 34.9 shrikes (SE = 9.4; Table 1).

We recorded 22 shrike observations on SCI in 2010, 2 of which we censored due to low certainty levels. Thus, we considered 20 observations reliable for purposes of abundance estimation. Of these 20 observations, 6 were detections by both observers and 14 were detections by only one observer. Fourteen observations were from Stratum 1, and 6 were from Stratum 2. Using these data, we estimated a detection probability of 0.462 (SE = 0.121; 95% CI = 0.248–0.690), which yielded a joint detection probability of 0.710 (SE = 0.131; 95% CI = 0.414–0.895) and an abundance estimate of 42.4 shrikes (SE = 9.7) in 2010 (Table 1).

Pooled Estimates

Combining shrike abundance estimates for nonexclusion areas over both islands yielded a pooled estimate of 204.0 (SE = 54.3) for 2009, and 282.0 (SE = 48.4) for 2010 (Table 1).

DISCUSSION

Lanius ludovicianus anthonyi occurred historically on the 4 Northern Channel Islands of Santa Rosa, Santa Cruz, Anacapa, and San Miguel, as well as Santa Barbara and Santa Catalina, but now occurs only on Santa Rosa, Santa Cruz, and Santa Catalina (Collins 2008, Hicks and Walter 2009). Grinnell and Miller (1944, summarized in Collins 2008) noted that this subspecies was "fairly common" on Santa Cruz Island and "less numerous" on San Miguel and Anacapa at the time, and other field ecologists (summarized in Collins 2008:279) listed the shrikes as "much in evidence" on Santa Rosa Island in 1927. However, from the early 1950s forward, the shrike distribution appeared to narrow. For example, Miller (1951) stated that shrikes on Santa Cruz

Island occurred regularly, but only in particular areas on the island (summarized in Hicks and Walter 2009). Island shrikes nested on San Miguel Island until 1976 and on Anacapa Island until 1978, but have not been observed breeding since (Collins 2008). Shrike numbers on Santa Catalina Island also appear to have declined over the past 40 years (Collins 2008). The range contraction of island shrikes over the past 60–70 years underscores the urgent need to understand their population status on the islands where they remain.

During the spring and summer (25 Apr–8 Jul) of 2006, Dye et al. (2009) surveyed approximately 6,450 ha of shrike breeding habitat in 22 canyons on SRI. They detected 37 adults, 3 fledglings, 28 juveniles, and 1 of an unknown age group. Considering only the adults, this translates into approximately 0.006 shrikes/ha. The sampling design used by these authors did not permit island-wide abundance estimates. By comparison, during the spring (1 Apr–27 Apr) of 2009 and 2010, we surveyed all cover types in a sample of the nonexclusion areas on SRI and detected 0.005 shrikes/ha in 2009 and 0.012 shrikes/ha in 2010. Thus, with respect to shrikes/ha, our 2009 results are similar to Dye et al. (2009), whereas the 2010 results were approximately 2 times larger. One possible explanation for the large jump in shrikes/ha in 2010 is that detection probabilities increased from 0.476 in 2009 to 0.825 in 2010. Nevertheless, despite the 2010 increase in shrikes/ha, our abundance estimates for 2009 and 2010 did not differ statistically (i.e., the estimate from 2009 is included in the 95% CI for 2010, and vice versa; Table 1).

During the spring (25 Mar–15 May) of 2006, Hicks and Walter (2009) surveyed approximately 10,000 ha of potential shrike habitat (described as grassland on gentle slopes close to a water source) on SCI. They detected only 12 shrikes, which translates into approximately 0.001 shrikes/ha. By comparison, during the spring (6 May–24 May) of 2009 and 2010, we surveyed all cover types in a sample of the nonexclusion areas on SCI and detected 0.002 shrikes/ha in 2009 and 0.002 shrikes/ha in 2010. Thus, our numbers were similar to those of Hicks and Walter (2009). Based on their counts, and extrapolating to areas not sampled, Hicks and Walter speculated there were between 24 and 36 shrikes on SCI. In a 2008 study, Warnock and McMorran (2008) counted 22–28 shrikes on SCI, though only a portion of the island was surveyed. In both of these studies, the abundance estimates and counts generally fall within the 95% CIs estimated from our study and, when one considers that in neither study was there a correction for detectabilities <1, their estimates are relatively close to our estimates of 35 shrikes in 2009 and 42 shrikes in 2010. Thus, it is possible that shrikes may be maintaining a stable population on the island, albeit at very low levels.

During the spring (6 May–15 Jun) of 1996, approximately 5,820 ha of Santa Catalina Island was surveyed for shrikes and 11 adults were detected, indicating that approximately 0.002 shrikes/ha were detected (C. de la Rosa, Catalina Island Conservancy, unpublished data). If one were to make the strong assumption that detection probabilities were similar for shrike surveys conducted on all 3 islands, then our data suggest that densities of shrikes are similar on Santa Catalina Island and SCI, but that densities are from 3 to 7 times higher on SRI. Note, we are not suggesting that detection probabilities were actually equal among islands. However, in the absence of abundance estimates incorporating detection probabilities, these data are the best available for making inter-island comparisons.

In the petition to federally list shrikes as an endangered species, Walter (2006) provided population estimates of 15–30 birds for SRI, 20–30 birds for SCI, and 10–25 birds for Santa Catalina Island, to yield between 45 and 85 birds for all 3 islands. In contrast, we estimated that on SRI and SCI alone there were 204 birds in 2009 and 282 birds in 2010. Adding Walter's estimated 10–25 birds for Santa Catalina, we calculate an estimate of between 214 and 307 shrikes for all 3 islands, or a maximum of 107–154 breeding pairs. Thus, our findings indicate that island shrikes are more numerous than Walter suggested. However, these numbers are still low with respect to maintaining viable populations of this subspecies.

Clearly the major difference in our numbers and those provided by Walter (2006) lies in our abundance estimates for SRI. Our data suggest shrike abundance on SRI is much greater than has previously been realized. The larger numbers of shrikes on SRI, when compared to SCI and Santa Catalina Island, suggests shrike habitat on SRI might currently be of better quality for shrikes than on the other 2 islands. This may, in part, be due to the eradication of sheep from the island in the 1930s. Although it is impossible to know the full extent of the damage caused by sheep on SRI, it is likely that the native plant communities began to recover following eradication of the sheep. More recently the vegetation on SRI has been undergoing additional significant changes. In 1992, all pigs were removed from SRI, and in 1998, all cattle were removed; subsequent monitoring has documented increased frequency and cover of native perennial species within tree and shrub dominated communities and within riparian areas. Further vegetation changes will likely occur with the complete removal of Roosevelt elk (*Cervus canadensis roosevelti*) and mule deer (*Odocoileus hemionus*) from the island by 31 December 2011. It is expected that decreased herbivory will stimulate increased plant growth. Similarly, on SCI sheep were removed from the island by 1988, and by 2006, feral pigs were completely removed. Removal of these species has resulted in significant and dramatic vegetative growth in recent decades, especially in regard to shrub and grassland cover.

It is unclear at this point how vegetative changes on SRI and SCI will influence habitat availability for shrikes on those islands. Although vegetative restoration is generally a beneficial outcome, it may not be beneficial to shrikes, which generally are known to use areas with short vegetation (e.g., Yosef and Grubb 1993, Collins 2008, St-Louis et al. 2010). For example, Lynn et al. (2006*a*) summarized breeding-season habitat use from 1998 to 2005 for San Clemente loggerhead shrikes, and found that occupied shrike territories had lower percent vegetative cover than

unoccupied territories. They suggested that a desirable management technique for enhancing foraging and nesting habitat use was to remove stands of tall, thick grass. An experimental study by Yosef and Grubb (1993) demonstrated that loggerhead shrikes in Florida, USA, actually caught as much food in areas with tall vegetation as in areas with mowed vegetation; however, the shrikes' metabolic expenditures were higher in tall growth than short because they had to hover more to forage successfully. Thus, these authors proposed that shrikes' net energy gain was probably lower in tall vegetation, overall.

Predicting shrike responses to vegetative restoration on SRI and SCI is not straightforward, however, because loggerhead shrikes also are known to use shrub and tree perches for foraging and nesting. For example, Lynn et al. (2006*a*) found that numbers of trees and shrubs were greater in occupied territories than in unoccupied territories, and many studies have noted that loggerhead shrike abundances, foraging rates, and numbers of nests increase where there are more shrubs and trees available (see summary in Lynn et al. 2006*b*). Thus, although vegetative changes on the islands may include the development of thick patches of tall grass that could impact foraging, eventual growth of shrubs and trees in these same areas may also facilitate foraging and nesting. Another unknown is how vegetative changes will affect populations of shrike nest predators, such as the island fox (*Urocyon littoralis santarosae*).

Based on our results, and similar to recommendations provided by Collins (2008), we think shrike numbers on SRI and SCI are low enough at the present time to warrant additional research into demographics (especially nesting suecess) and habitat needs. For example, concerns about possible impacts of island scrub jays (*Aphelocoma insularis*) on shrikes on SCI (e.g., Miller 1951) necessitate an examination of interactions between them, and reproductive data (especially chick and fledgling survival rates) are lacking for *L. l. anthonyi*. Considering the rapid changes in vegetation occurring on both islands, and the fact that even more dramatic changes are expected to occur on SRI once elk and deer are removed, it is imperative to know what management actions can be taken to ensure perpetuation of this subspecies. Likewise, we think that rigorous and statistically defensible surveys of shrike abundance on Santa Catalina Island, like those that we report for SRI and SCI, are needed to further clarify and resolve the range-wide status of this subspecies.

MANAGEMENT IMPLICATIONS

Management of shrikes on SRI and SCI presents an unusual problem. Whereas efforts to remove nonnative herbivores (e.g., deer, elk) are desirable from the standpoint of restoring vegetation on the islands to a more natural state, the vegetation changes expected could negatively affect shrike numbers, which our data indicate are already low. In particular, we expect numbers of trees and shrubs to increase with a simultaneous increase in the height and density of grasses. Because shrikes hunt from perches, increased numbers of trees and shrubs—up to a point—may benefit shrikes. However, shrikes also prefer foraging in areas with bare ground where their prey are more visible, so increased grass height and density may be detrimental to shrikes. Clearly the tandem goals of restoring island vegetation while also maintaining shrike numbers suggests management of shrikes on the islands should include the collection of current demographic information on the breeding population of shrikes, and should especially identify factors contributing to the decline of the subspecies such as habitat change and predation. Our abundance estimates represent a baseline against which land managers should evaluate the effectiveness of their management efforts, and the interannual variability in abundance estimable from our data should be used to develop target abundance levels for each island that account for environmental and demographic stochasticity.

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