

SIZE AND DISTRIBUTION OF THE CALIFORNIA BROWN PELICAN METAPOPOPULATION IN A NON-ENSO YEAR

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SUMMARY

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In 2009, the Brown Pelican *Pelecanus occidentalis* was removed from the US Endangered Species List. The California subspecies *P. o. californicus* (hereafter CABRPE) was also removed from the California state endangered species list. Three years earlier (2006), we estimated the metapopulation of CABRPE to be 70680 ± 2640 breeding pairs (mean \pm SD) and 195900 ± 7225 individuals. The largest single breeding aggregation within the range occurred among two to three islands of the San Lorenzo Archipelago, Gulf of California (GOC), totaling ~ 17225 breeding pairs, or $\sim 24.4\%$ of the metapopulation. This and the other 4 subpopulations were composed of a single “core” breeding aggregation (on one or a few adjacent islands) and many smaller but isolated colonies (a colony represents all birds on a single island). Extremely small colonies (<65 nests) made up $\sim 35.6\%$ of total colonies, but only $\sim 0.9\%$ of the total estimated numbers, corrected for detectability. Modal colony size throughout the range was much smaller (230 to 1300 breeding pairs), indicating that small, scattered colonies and sub-colonies have a function in CABRPE distributional dynamics and demography. Thus, negative single-survey data (no occupancy and small numbers) still have conservation importance as alternate or growing colonies in source-sink dynamics. Little numerical change in CABRPE in at least three decades was indicated from less precise data south of the northernmost Southern California Bight (SCB) subpopulation, but significant recent improvements in the SCB were reflected by our high estimates in 2006, supporting the USFWS delisting. At that time, we estimated the improved SCB breeding population as 11695 ± 450 pairs. However, continuing threats throughout the range, especially in the south, now include commercial fishing, tourist developments, increased human activities, and extensive/expanding aqua-cultural developments (as well as, to a lesser degree, agricultural activities). Repeated endangerment is a possibility. Continued monitoring will be important.

Key words: California Brown Pelican, *Pelecanus occidentalis californicus*, breeding pairs, El Niño, metapopulation estimate, subpopulation estimates, survey bias, distribution, breeding atlas

INTRODUCTION

Source-sink phenomena are central to Brown Pelican *Pelecanus occidentalis* (BRPE) distributional and population dynamics. For example, recruitment of Gulf of California (GOC) pelicans to the Southern California Bight (SCB) subpopulation occurred during the 1970s, when SCB numbers were declining (Anderson & Gress 1983) and when the entire species was classified as Endangered under the *Endangered Species Act* (US Fish and Wildlife Service [USFWS] 1983). SCB numbers later recovered (Anderson *et al.* 1975, Anderson & Anderson 1976, Anderson & Gress 1983), and the species was delisted in 2009 (USFWS 2009a). The recruitment of new breeders into the SCB from other colonies within the metapopulation likely enhanced that recovery.

Anderson & Gress (1983, their Fig. 2) also demonstrated that, within the SCB region itself, numbers of breeders at Anacapa Island and Isla Coronado Norte (separated by 260 km), shifted in predictable episodes, probably in response to changing food availability (Anderson *et al.* 1982), demonstrating characteristics of a metapopulation.

Anderson & King (2005) reviewed key metapopulation concepts as they applied to the American White Pelican *P. erythrorhynchos*; here, we apply those definitions to the California BRPE subspecies *P. o. californicus* (hereafter, CABRPE) (Fig. 1). The subspecies taxon, based on the American Ornithologists' Union (AOU, 1957), has also been shown to be highly useful for conservation purposes (discussions in Winker & Haig 2010).

Essentially, “metapopulation” has been defined by Newton (2004) as: “any population composed of a number of discrete and partially independent subpopulations that live in separate areas but are linked by dispersal” and by Morris & Doak (2002:375) as: “sets of discrete, largely (but not entirely) independent populations whose dynamics are driven by local extinction and recolonization via movement from other populations.” The basic units of the metapopulation for CABRPE are colonies; i.e., the total nesting pairs on individual islands, sometimes grouped into “aggregations” that comprise a clump of closely adjacent colonies.

How the presumed metapopulation structure for the CABRPE (Fig. 1) might act as a paradigm for conservation strategy and

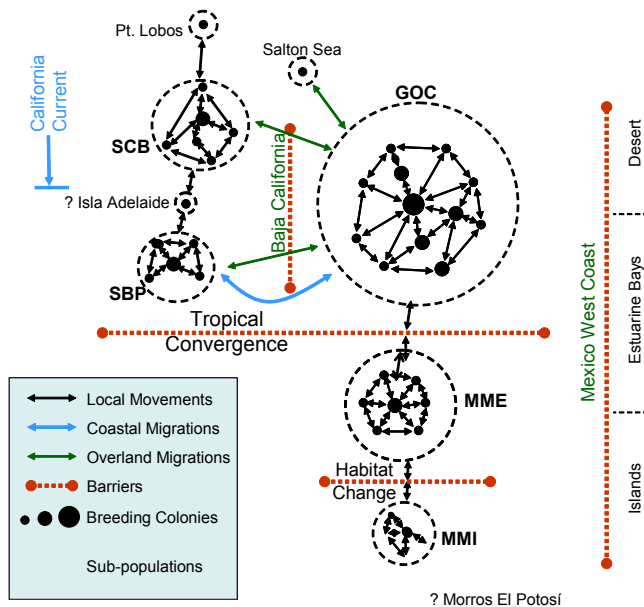


Fig 1. Hypothesized subpopulation zones within the presumed California Brown Pelican metapopulation. Short dashed lines represent subpopulation boundaries, and black patches connected by arrows (movement and dispersal) represent colonies and sub-colonies. This diagram was based on natural history as well as distributional and vulnerability characteristics that would tend to separate subpopulations into more or less independent geographic and demographic units: SCB = Southern California Bight, based mainly on confines of the California Current System (USFWS 1983); SBP = Southern Baja-Pacific, based on isolation along the southwest Baja California coast (USFWS 1983), terminated by the tropical convergence (Anderson 1983); Gulf of California (GOC) based on the confines of the Gulf of California (Anderson 1983); Mexican Mainland, Estuarine (MME) based on dominant nesting and feeding habitat (mangrove-dominated bays with vegetated islands) (GOC and MME are essentially also separated by the tropical convergence; see Anderson 1983); and Mexican Mainland, Island (MMI) = is separated from MME by a sudden change in nesting habitat and offshore oceanographic changes. The form of this diagram is adapted from the discussions of Buckley & Downer (1996). This diagram should be considered preliminary and subject to future testing and revision with demographic, genetic, morphological and ecological data. These designations entail regional management units (of course, the single colony is the most basic management unit). As conservation problems develop in the future, the management of these units may be combined or further separated.

planning (see Morrison *et al.* 2012), or how well these subdivisions fit demographic and genetic reality, require further research. But, as demonstrated by Morrison *et al.* (2012) for another endangered bird species, CABRPE population-segments, by those authors’ definition, meet their metapopulation criteria: discrete populations (subpopulations) with independent demographics that interact through dispersal or migration. Thus, we have also retained the term “subpopulation” to include CABRPE local geographical subdivisions (equivalent to ecological subpopulations) (Fig. 1).

In this paper, the subpopulations as we define them (Fig. 1) are based largely on geographical and habitat segregations of nesters, each with unique threats and conservation needs—even down to the single colony unit. The smaller-named sub-divisions within the CABRPE subpopulations are here applied as suggested by Anderson & King (2005) and Anderson *et al.* (2007). Colonies are also readily definable units within distinct administrative and management jurisdictions and therefore also useful categorizations for conservation. In this paper, we do not discuss distribution of the CABRPE outside of the breeding season, but we recognize that this aspect of distributional dynamics is important and requires separate and detailed treatment (see Jaques *et al.* 2008 and USFWS 2009b for California).

Influence of ENSO events on breeding numbers of CABRPE

Year-to-year variation in numbers of breeding Pacific seabirds, their productivity, their survival at times and their behavior are heavily affected by the El Niño/Southern Oscillation phenomenon (ENSO; Ainley *et al.* 1988), the major driver of GOC oceanographic variation (Baumgartner & Christensen 1985). It is not our intent here to review ENSO and seabird demography, but only to add perspective to findings in 2006 and to understand variability in year-to-year population size estimates. Anderson & Gress (1983) demonstrated in the SCB that differing proportions of available adults nested in any given year. This type of variation is also strongly related to ENSO effects in seabirds in the GOC, influencing adult body condition, breeding effort and reproductive success (Velarde & Ezcurra 2002, Velarde *et al.* 2004).

At Isla Piojo, GOC, Anderson *et al.* (2006) reported that numbers of breeding attempts varied between 0 and 1430 from 1969 through 2005 (mean \pm 95% CI 432 ± 114 , CV 73%, $n = 32$ y). In this same region of the GOC, Velarde & Ezcurra (2002) reviewed and reported similarly high annual variations in breeding attempts and reproductive success of other species nesting in proximity to BRPEs. In the 36-year period reported by Anderson *et al.* (2006), using standardized, normally distributed Southern Oscillation Indices (SOIs), and conversions similar to those of Velarde & Ezcurra (2002), they calculated that ENSOs of varying strength affected 38% of the breeding seasons. Based upon several colonies in the GOC studied closely since 1970 (Isla Piojo, San Lorenzo Archipelago, Puerto Refugio, Isla San Pedro Mártir and Isla San Luis [Appendix 2]), 2006 represented a year of near-maximum breeding attempts over a four-decade period (DWA, field notes; Anderson 2013). For example, in 39 years of study from 1970 to 2009 at the San Lorenzo Archipelago, GOC, 2006 population-size ranked second only to a higher population in 1988 and was statistically indistinguishable from only two other years (1970, 1993) ($n = 33$ y). At Isla Piojo, GOC, the 2006 breeding population was also an optimal nesting year for CABRPE, ranking third-highest in 37 years of data.

In the northern SCB, Anderson & Gress (1983) indicated that, from 1972 to 1979, about 20%–70% of the available adults in the region bred; the lowest proportion of breeding adults (19%) was in 1977, a “mild” ENSO year, whereas the proportion of adults in the total numbers remained relatively constant through the same period (64–86%; mean \pm 95% CI 72% \pm 8%; CV 14%). The highest proportion of adults was in 1978, when fewer young were expected because of lowered productivity the previous year. Thus, 2006 was a near-optimal nesting year for CABRPE. Elsewhere, at the southern range periphery of CABRPE, the same effect has been demonstrated. Sarmiento (1994) described year-to-year variation in breeding numbers at Isla de Pájaros, Sinaloa, Mexico: during the 1991/92 ENSO, 69 completed nests were found, in contrast to 334 active nests in 1992/93 (non-ENSO), representing a 79% reduction.

Our objectives were: (1) during a near-optimal breeding year, to obtain an estimate of total numbers of breeding pairs of the *P. o. californicus* subspecies, along with estimates of precision; (2) accurately determine the locations and sizes of breeding colonies and provide a near-complete breeding-colony atlas (appendices); and (3) define and census presumed subpopulations (Fig. 1) (also subject to a testable hypothesis using genetics). All objectives were intended to provide the USFWS with population estimates for their proposed delisting (USFWS 2006) and to provide a baseline for future evaluations.

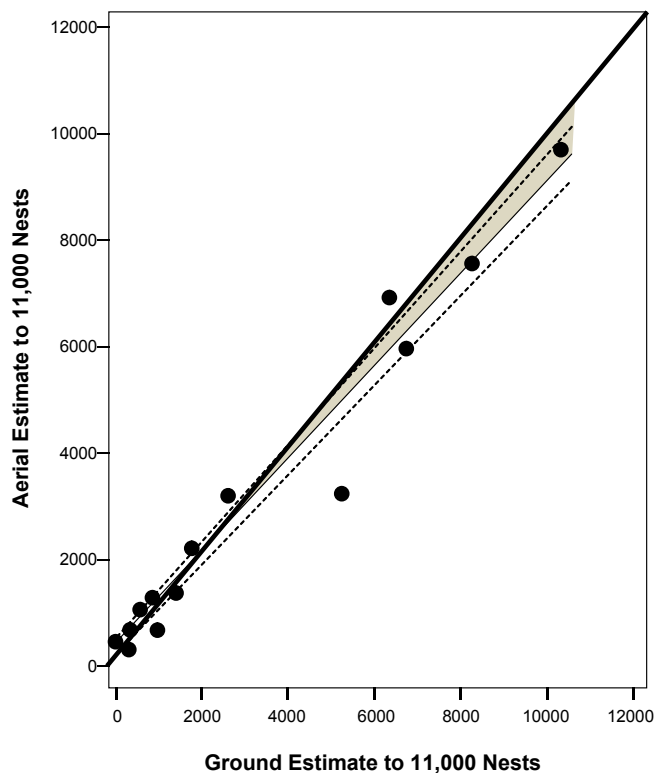


Fig. 2. Overall linear regression analysis of aerial estimates on ground-truth estimates for which (1) matched data were available and (2) phenology differences between the date of aerial survey and ground-truth surveys were small (i.e., the aerial survey was timed correctly for an accurate estimate). Dashed lines are 95% CL. The shaded portion shows how aerial estimates tend to underestimate ground-truth values.

METHODS

Study area, population units and survey methods

The total breeding range of *P. o. californicus* extends from 17°N to 36°N, or >4800 km of coastline (Palmer 1962:275). Godínez-Reyes *et al.* (2006) describe a GOC monitoring plan to determine baseline CABRPE populations (as a seabird sentinel) in the GOC Natural Protected Areas (termed here the *Salud Project*), of which this project was an early component.

The USFWS (1983) described the presumed subpopulations (used herein with minor modifications) based on distributional and ecological characteristics (Fig. 1). Additional general historical insights and distributional/numeric data were derived from Bent (1922), Grinnell (1928), Grinnell & Miller (1944), Wetmore (1945), Hutchinson (1950), AOU (1957, 1998), Palmer (1962), Gress (1970), Jehl (1973), Anderson & Anderson (1976), USFWS (1983), and Johnsgard (1993). Population estimates for various colonies have been reported by Anderson *et al.* (1976), Anderson (1983), Everett & Anderson (1991), Velarde & Anderson (1994), and Shields (2002). Overall distributional information, especially for the south, was summarized by Howell & Webb (1995) and Wilbur (1987). From a review of these references, we posed the question: could more clarification be provided on the relatively unknown southern, peripheral (extralimital) populations (Palmer 1962:275)? We surveyed the entire range through aerial survey, ground counts, information from cooperators, literature review and files of unpublished records kept by one of us (DWA) since 1970. Personal field work and accumulated records by DWA were also summarized here to record past breeding sites.

DWA, CJH, and JB surveyed the entire range in a twin-engine *Partanavia* aircraft south of the US/Mexico border during the breeding season, 23 March–1 April 2006 ($n = 68$ colonies [70.1%] overflown by us in 10 d plus $n = 9$ colonies [9.3%] not surveyed by air due to inclement weather or extreme distance with tenuous gasoline sources, for a total of 79.4% of known colonies). We supplemented aerial surveys with ground-based surveys. In the northern SCB subpopulation (Fig. 1, Fig. 2), *only* ground-based estimates were obtained by FG and ELP ($n = 8$ northernmost colonies [8.2%] without aerial surveys). In the SCB, partial aerial survey data were compared to ground-truth surveys in California and northwest Baja California to evaluate degree of phenological bias for a one-time survey of many degrees of latitude. In the GOC, DWA, CGR, KSP, and JPG conducted ground-truth surveys, but in the extreme south, colony estimates were obtained only through a literature review or cooperator information ($n = 11$ potential colonies [11.3%]).

Final estimates of active nests (breeding pairs) were based on various combinations of these methods. Three additional, outside sources of quasi-double-sampling were also used at some sites (information from cooperators, surveys of recent and historical records compiled by DWA, and literature sources where no other information existed, used to supplement our 2006 observations). However, only concurrent, matching ground-truth/aerial survey pairs were used to estimate observer error. Our final estimates (when we had matching pairs) were based upon ground-truth data, either our own or from cooperators' efforts (see Green *et al.* 2006, 2010), and compared, when comparisons could be made (Taylor & Pollard 2008: “DS method”; 27 of 59 [45.8%] 2006-occupied colonies).

Unfortunately, import and security restrictions prevented us from documenting the colonies with a large-format camera. Thus, only hand-held digital cameras were used, with limited success. In the aerial survey, there were three observers, DWA, CJH, and JB (a census-experienced biologist-pilot). All potential habitats, conditions permitting, were overflowed at 100–400 feet and examined for fresh guano deposits, bird activity and nests. If active nests were found, pelicans and all cormorants (*Phalacrocorax* sp.) were enumerated by aerial visual estimate. Nesting colonies were circled 1–4 times, each observer estimating numbers of active nests (depending on size of colony: by estimating groups of 10 or 100 for larger colonies or counting individually for smaller colonies) and photographed for later confirmation, if time and fuel-levels permitted. Before leaving the area, the three observers discussed their estimates and agreed on a final value; in the rare case that large discrepancies occurred, the colony was circled again and re-assessed. Roosting pelicans and cormorants were also overflowed, visually estimated and sometimes photographed. Age-classes as seen from the air or in photographs were categorized, if possible, as: *brown-heads* = young of the previous two seasons; or *white-heads* = likely breeding-age pelicans (see Schreiber *et al.* 1989).

Data analysis

Statistical summaries were performed using *MINITAB* 15.1 (Minitab Inc., www.minitab.com/). In estimating precision, we considered all selected ground-truth data equally and began by choosing such values as superior over aerial estimates because of their completeness. We also related aerial values to ground-truth values, where matched data were available, to calculate correction factors (CFs) for use in instances where no ground-truth data were available. For colonies lacking ground-truth data and having estimates from aerial data only, colonies of <750 nests (calculated CF = <1.06) were not corrected, but larger ones (as estimated from the air) were corrected using the CFs derived from different-sized colonies, as follows: CF = n_g/n_a (slope of regression line used to approximate CF), where n_a = numbers of nests estimated from the air, n_g = ground-truth estimates. A similar calculation was termed *aerial visibility factor* by Henny *et al.* 2008 (Fig. 2).

Overall, and given the large sample of ground-truth data, only four estimates of colony size out of 59 total colonies enumerated by air (6.8%) required correction. Most values were therefore derived from ground-truth surveys with multiple sampling. But where no other information existed, “raw” estimates from recent years (previous two years) were used. Where *only* earlier records (before previous two years) were available (11 of 59 2006-occupied/or -presumed-occupied colonies, or 18.6% of the total number of colonies seen and/or known, for example, Isla San Pedro Mártir, Appendix 3; and Isla Cedros, Appendix 2), only older estimates could be used (except, in the case of Isla San Gerónimo, a 2007 record, Appendix 1).

In our estimates of variance for the two subpopulations, SCB and SBP, we also applied the variances derived from regression analyses. Each was calculated separately and based on a regression analysis that included the total estimate for comparisons at different maximum colony sizes. We then applied these correction factors, based on the appropriate colony-size-range regression line. These estimates were as near those numbers as the available data-set allowed. Error estimates derived from the regression analyses

of n_a on n_g were applied to our values for the SCB and SBP subpopulations, assuming that if we had flown over those areas at the phenologically correct time, our precision would have been approximately the same. We feel that this assumption is reasonable, based on three similar Osprey *Pandion haliaetus* surveys conducted by CJH and DWA from 1977 to 2006, who were attempting to detect single, large nests (more tentative than *groups* of large nests; Henny *et al.* 2008). They found remarkable consistency among the three surveys.

In estimating variance for some subpopulations or for the total metapopulation, we used double-sampled data and assumed that the smaller numbers in ground-truth versus aerial comparisons were the values seen in both approaches (aerial *versus* ground, *versus* aerial-photograph, *versus* ground-truth provided by a cooperator). Our best aerial estimate of colony size was either larger or smaller than the best estimate of ground-truth, but we always accepted ground-truth as the value for our final estimate. Calculations of estimated variance (\hat{V}) were first assumed to be binomial and then calculated with the basic formula suggested by Pollock & Kendall (1987) (and essentially the same as that used by Henny *et al.* 2008), and reduced, as follows:

$$\hat{V} = \frac{(n_a)(n_g)(n_a - m)(n_g - m)}{(m)^3}$$

where m = the smallest number of the double-estimate.

These estimates were applied, however, only to two subpopulations (GOC, MME) because those were the only areas for which we had phenologically correct double-samplings. Since this was the largest sample from which to estimate variance from the entire metapopulation, the combined GOC and MME variance (considered a measure of precision) was also applied to the total metapopulation estimate. Given the large sample of ground-truth data ($n = 27$ of 59 colonies [45.8%]), the implied high levels of precision seem warranted.

All estimates of variance were converted to standard deviations (SDs) and presented in that form as measures of precision. For the southern periphery subpopulation (MMI, Fig. 1, Appendix 5) no estimates of precision were calculated.

In the survey of Ospreys by Henny *et al.* (2008), the average detection probability for Ospreys (large birds with single, large nests) was 0.5714. We assumed this value to be similar for single BRPE nests. In examining detectability, assumed to be largely a function of colony size (Pollock & Kendall 1987), we applied a linear function and projected from this value to the number of nests with an expected probability of 1. To estimate the slope of this line, we used a function of 10/11 (detection rate of colonies of ≤ 65 nests) to estimate the rate of increase to 1. Further estimation of this upper-value was also supported by using an apparent “break” seen in similar data reported by Angehr & Kushlan (2007) (two colonies with a mean 65 nests) compared to our data (four colonies with a mean >64 nests).

Personal field work and accumulated records by DWA were summarized to record locations where pelicans had bred in the past. Furthermore, the published literature was reviewed to supplement the total record (appendices).

RESULTS

Detectability of small BRPE colonies

We estimated (based on the mean colony size of all colonies >65 nests) that five small colonies would have gone undetected during our survey, or about 0.66% of all nests but 27.1% of all small (<65-nest) colonies. If colony detection was purely a function of the number of nests (Pollock & Kendall 1987), then colony detectability for BRPE aerial visual estimate surveys rapidly rises to 1. Admittedly, these detection estimates are tentative. Still, corrected for this reduced detectability, these small colonies (<65 nests) comprised only 0.21% of the total estimated nests. Angehr & Kushlan (2007) detected only two small colonies (of five and four nests) among 17 larger colonies (ranging from 62 to 1265 nests) in the Gulf of Panama. These miniscule groups may, however, be important in metapopulation dynamics, source-sink phenomena, and local breeding extirpation or establishment. We found high overall estimated aerial detectability of BRPE concentrations and colonies, usually of larger sizes (>65), which included large birds

conspicuous in, over or near occupied nesting substrate (or on the ground in large flocks), usually also “marked” with large, white patches of fresh guano from BRPEs and other associated species (in contrast to pinkish or yellowish patches, which would be from previous years). Therefore, we believe small, rare occurrences of BRPEs were inconsequential (an adjustment would add ~150 nests and only five more colonies to the overall estimate). This conclusion is supported by a previous study of 16 colonial waterbird species in Florida, which found the highest detection rate in aerial surveys for BRPE (100% detection, with the lowest false-positive rate); BRPE was the most amenable of these species to simple aerial survey (Rogers *et al.* 2005).

Sampling bias

As colony sizes become larger, our aerial estimates tended to steadily and increasingly underestimate ground-truth values (Fig. 2), a tendency commonly noted but correctable in aerial surveys. CFs from regression analyses of series with various maximum colony sizes (not all shown) were as follows: <100 nests, 1.06; <750 nests, 1.06; <2000 nests, 1.08; <7000 nests, 1.14; <11000 nests, 1.10.

TABLE 1
Estimated subpopulation and metapopulation sizes of CABRPE and their general characteristics, based on the 2006 aerial and ground-truth surveys

Designated subpopulation	Estimated size of breeding population \pm SD ^a	% of meta-population	Dominant breeding substrate	Dominant feeding habitat during breeding	No. of known colonies ^b	% of colonies occupied in 2006	Colony size		
							Mean (CV, %) ^c	Median	Range
Southern California Bight(SCB)	11 695 \pm 450 ^d	16.6	Vegetated oceanic island, ground-nests	Offshore, pelagic	14 (11)	79	1 063 (131)	250	10–4 000
Southern Baja-Pacific (SBP)	3 100 \pm 170 ^d	4.4	Oceanic desert island, ground-nests	Offshore, pelagic; estuarine	11 (5)	45	620 (121)	350	100–1 950
Gulf of California (GOC)	43 350 \pm 230 ^e	61.5	Oceanic desert island, ground-nests	Offshore, pelagic	42 (24)	57	1 806 (164)	525	5–10 625 ^f
Mexican Mainland-Estuarine (MME)	10 540 \pm 270 ^e	14.9	Estuarine island, tree/bush nests	Estuarine; some offshore, pelagic	15 (11)	73	958 (216)	90	2–6 950
Mexican Mainland-Island (MMI)	1 845	2.6	Forested oceanic island, tree/bush nests	Offshore, pelagic	15 (8)	53	231 (127)	98	25–850
Total meta-population estimate	70 680 ^f \pm 2 640 ^d	100.0			97 (64)	66	1 314 (176)	320	2–10 625

^a Best maximum estimates of numbers of breeding pairs from multiple-samplings and various data sources (cited in the text and appendices).

^b Values represent the total number of known sites with current or previously known (historical) CABRPE nesting. Numbers in parentheses represent the number of active colonies or sub-colonies in 2006. All known historical colonies would not be expected to be occupied in a single year.

^c CV = coefficient of variation.

^d SD based on regression estimates.

^e SD based on binomial distribution.

^f Value corrected for the estimated reduced detectability of small colonies (<65 nests), adding about 150 pairs and five occupied colonies to the total.

Phenology bias

Sampling biases associated with phenological differences in an aerial survey over a wide range of latitudes (~11° in 10 d) might negate a single aerial survey. For the SCB subpopulation (on northern range periphery) (Appendix 1), our flight occurred too early in 2006 to be useful by itself (which is why we depended entirely on ground-truth); but a CF of 1.5 is provided for possible future adjustments. Given the high expected year-to-year variations in the nesting cycle, even within a subpopulation (see Anderson & Gress 1983, their Fig. 4), a correction could be problematic and could necessitate multiple surveys on multiple dates, especially if only aerial visual estimate surveys are done.

CABRPE total population estimates, 2006

We estimated the entire metapopulation to be 70 700 ± 2 600 breeding pairs (rounded to the nearest 100, Table 1, Fig. 3). The largest subpopulation (Fig. 4) occurred in the Gulf of California (~43 400 breeding pairs) and included the largest single breeding aggregation, located in the Midriff Region of the GOC on the San Lorenzo Archipelago (~17 200 pairs; Table 1, Appendix 3). Age-ratio calculations from aerial survey for the entire metapopulation indicated 71.9% adults and 28.1% ± 0.33% (95% CL) immatures

(n = 71 287). Applying these age ratios to our overall estimate, the total CABRPE metapopulation in 2006 (an above-average breeding season preceded by two years of high production and survival of adult birds; DWA field notes) was >195 900 ± 7 200 individuals (rounded). This estimate does not include non-breeding individuals possibly dispersed to the north or south out of our census zone, the breeding range.

Regarding distribution, there was a decrease in large colonies (>500 nests) on both the northern or southern peripheries of the range (Fig. 5), with the largest colonies dominating the central, and to a lesser degree, northern portions of the range. Yet, median colony size was only 300 pairs, scattered widely throughout each subpopulation (Fig. 5). None of the subpopulation occupancy rates (all known colony locations versus those occupied only in 2006; Table 1) were significantly different from one another (*P* = 0.90, Chi square test), with an overall 66% occupancy rate in 2006 (Table 1).

Overall distribution and delineation of the presumed CABRPE subpopulations (Fig. 4, Table 1)

Southern California Bight (SCB). This subpopulation (Appendix 1) has been defined mainly by the bounds of the California Current System (Anderson & Gress 1983; USFWS 1983). It includes the

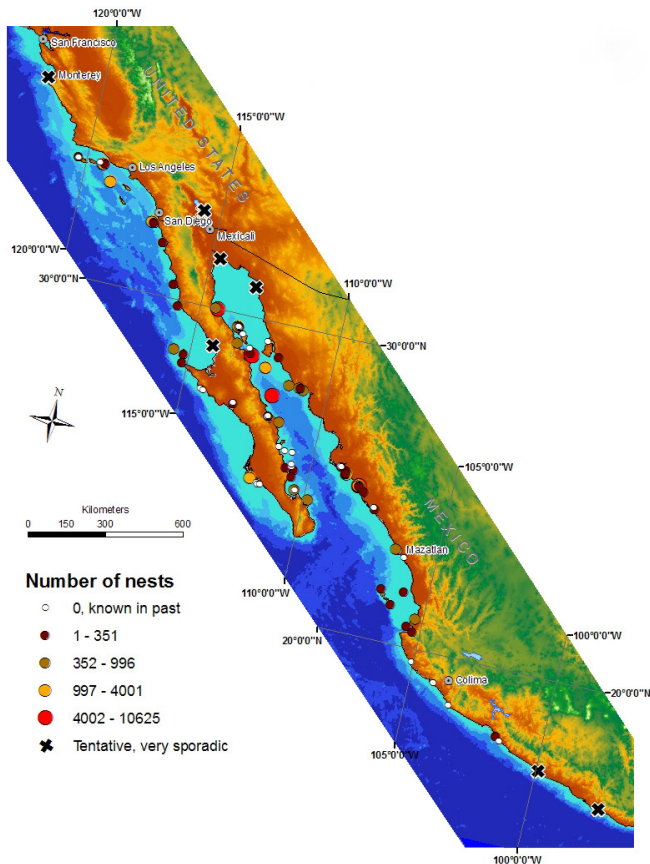


Fig. 3. The range of the California Brown Pelican (basemap data source: NGDC, USGS, ESRI, coordinate system WGS 84), 2006. Nesting colonies on the west coast of North America are plotted by size (circles of various sizes and colors) and current occupancy (small white circles indicate previous colonies, not currently occupied). Black Xs represent sporadic and largely temporary nesting colony locations, likely associated with breeding range limits.

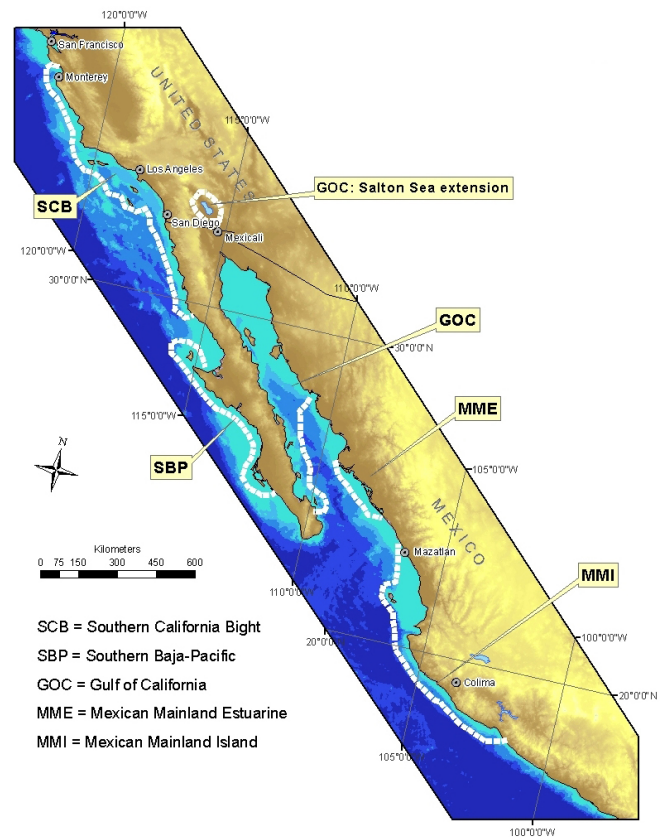


Fig. 4. Subpopulations of the California Brown Pelican and their approximate boundaries (shown by white, dashed lines and named as in Fig. 1 and Table 1).

mid- and south coast of California plus the northwest coast of Baja California south to Isla San Gerónimo, Baja California, the southernmost known BRPE nesting location for this subpopulation. Due to its remoteness and distance from gasoline supplies, Isla San Gerónimo was not surveyed from the air in 2006. However, ELP and H. Carter (pers. comm.) observed ~200 nests there in April 2007, and that value was applied to 2006. Additional details for this and the following subpopulations are provided in the appendices.

Southern Baja-Pacific (SBP). This subpopulation (Appendix 2) includes the mid- and southern Pacific coast of Baja California, from Isla San Gerónimo (29°47.5'N) to the south end of Isla Creciente (southern Magdalena Bay, 24°17.0'N). Hutchinson (1950:122-133) documented the nesting of BRPEs (but with no estimate of numbers) in this area. Another potential nesting location, Isla Adelaide (28°40.2'N, 114°16.7'W) is not listed in Appendix 2, because pelican nesting has never been confirmed, although thousands of Brandt's Cormorants *P. penicillatus* regularly nest there, with 400–500 BRPEs usually found loafing in the area (DWA field notes). Isla Adelaide was reported (Katsuo Nishikawa, pers. comm.) to have possible, sporadic, but very low numbers of nesting BRPEs. This was not confirmed in 2006, although the island is often mentioned by many of the authors cited above as an important nesting location for other seabirds. Our name for this subpopulation was changed slightly from that given by the USFWS (1983) ("Southwest Baja California Coastal Population") to avoid name-confusion with the SCB subpopulation. No nesting records of BRPEs have been reported for the outermost island groups of western Baja California, the Islas Revillagigedos (Wehtje *et al.* 1993) and Isla Guadalupe (Jehl & Everett 1985), where Howell & Cade (1954) categorized BRPE as "accidental."

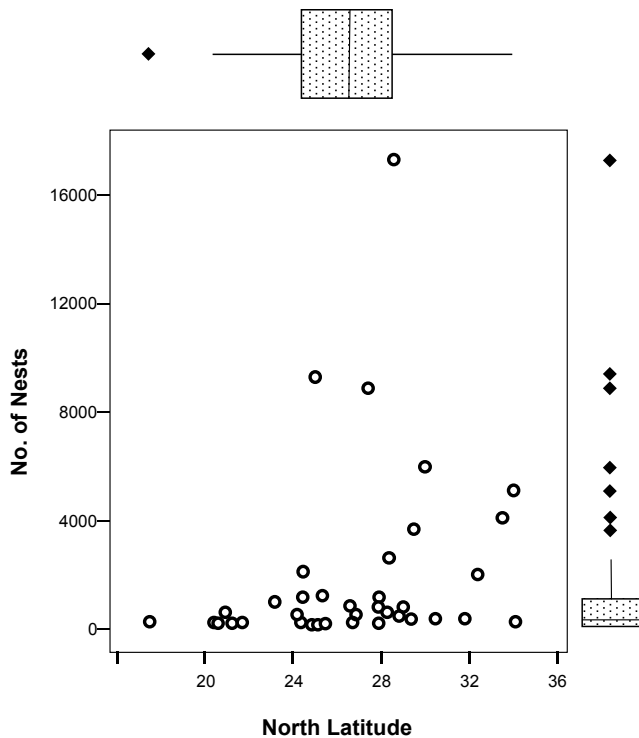


Fig. 5. North latitude versus colony size (number of breeding pairs) in 2006 survey data. On the upper box plot, Q1-Q3 = 24.5°N to 28.6°N, median = 26.7°N. On the right box plot, Q1-Q3 = 50 to 1030 breeding pairs, median = 300.

Gulf of California (GOC). This subpopulation was always regarded as the largest (USFWS 1983) and includes the Gulf of California, north from Isla Cerralvo, Baja California Sur (24°14.6'N, 109°51.4'W), and north from Punta Calaveras (near Guásimas), Sonora (27°53.4'N, 110°40.8'W) (Appendix 3). A large gap in BRPE nesting distribution occurs from the southern terminus of Bahía de Magdalena (24°20.0'N) south and around Cabo San Lucas to Isla Cerralvo (Fig. 4). Much additional information on the GOC subpopulation and the two subpopulations farther south is provided by Velarde *et al.* (2005). The largest numbers of breeding pairs on a single island, anywhere, were at Isla San Lorenzo Norte (Animás) in the GOC (Appendix 3). With colonies of closely adjacent Islas Salsipuedes, San Lorenzo Norte (Animás) and San Lorenzo Sur (together forming the San Lorenzo Archipelago), this location constituted the largest single breeding aggregation (Table 1).

Mexican Mainland, Estuarine (MME). There is another apparent gap in the distribution of BRPE nesting to the south from about Punta Calaveras (near Guásimas), Sonora, south to about Boca las Piedras at the mouth of the Río Fuerte Nuevo (25°49.1'N, 109°25.6'W, about 400 km), where a distinct change in available and utilized nesting substrate (to mangrove-dominated and vegetated, estuarine islands) occurs. Pelicans in this subpopulation typically nest mostly in mangrove canopies south to about Peninsula Quevedo (23°54.9'N, 106°58.2'W) (Appendix 4).

BRPE in this subpopulation also typically shift among the numerous bays and islands with some frequency (details in footnotes, Appendix 4). With nesting colonies shifting location, the numerous unoccupied islands of the region (where no current colonial waterbird nesting activity was seen in 2006) certainly remain important as alternate nesting or roosting locations. Why the shifts occur is not known. The isolated barrier islands, large bays and extensive mangrove habitats with large numbers of islands along Mexico's west coast are critical for many breeding (and wintering) waterbird and waterfowl species in addition to CABRPE (Carmona & Danemann 1994, Pérez-Arteaga *et al.* 2002), mostly characterized by the extensive use of mangrove-dominated wetland habitats. These habitats are currently being severely altered by mariculture activities (Naylor *et al.* 1998, Cruz-Torres 2000, DeWalt *et al.* 2002, Warne 2011) and other causes of mangrove destruction (Duke *et al.* 2007).

Mexican Mainland, Island (MMI). This subpopulation is found nesting mostly in bushes and trees on offshore islands, south of a gap starting at about Mazatlán (23°16'N, 106°28'W) and ending at about Isla Grande, Guerrero (17°40.6'N) (Appendix 5). The southern limits of this subpopulation (and the subspecies designation) are not well known. Extensive shoreline development, as well as tourist, agriculture and mariculture activity, characterizes much this area's coastline; it is highly likely, but undocumented, that larger numbers of BRPEs nested along this coastline historically.

Our MMI designation was based mostly on dominant nesting habitat association (offshore islands), distance (physical gap) from the more estuarine-inhabiting MME nesting colonies (Fig. 4), and several reliable correspondents and references. Knoder *et al.* (1980) conducted eight aerial censuses of variable coverage from the Guatemala border north into Mexico from 1971 to 1979 and reported (pers. comm. 1980) no pelican colonies to speak of south of Puerto Vallarta, Jalisco. Gonzalo Gaviño de la Torre (pers. comm. 1978) reported "little or no Brown Pelican nesting south of Isla Grande" (see also references, Appendix 5).

Extralimital populations?

Subspecies categorization in the presumed zone of overlap is likely dynamic and tentative. Knowledge of peripheral populations, south of the subpopulations described above, is scarce (other than in Panama). The BRPE subspecies that is known to breed rarely in Central America, perhaps also rarely nests in the presumed large gap between *P. o. californicus* and *P. o. carolinensis* ranges (~1500 km). For example, a colony was reported in 1971 at Laguna Chacahua, Oaxaca, Mexico (listed in Appendix 5). Jehl (1974) conducted seabird surveys within the offshore areas of this region (see Fig. 4) and commented specifically on the general rarity of BRPE. Importantly, BRPE nesting is not reported by Binford (1989), the most authoritative author on the birds of Oaxaca, and we must therefore conclude that the Laguna Chacahua record, although based on a reliable source, was no more than a sporadic, northern record for *P. o. carolinensis* (see also Thurber *et al.* 1987:128-129). Furthermore, the Laguna Chacahua area has been a National Park (*Parque Nacional Lagunas de Chacahua*) with constant annual monitoring, but no known BRPE nesting has been reported since at least the early 1970s (J.E. Mendoza pers. comm.; FG field notes). In any case, we doubt that this record at the range peripheries of both subspecies represents a regular location for nesting pelicans. Howell & Webb (1995:126), who are also recognized authorities for this region, indicate that the first “regular” BRPE colony to the south of Isla Grande (Appendix 5) is located in the Gulf of Fonseca (13°16'N, 87°42'W), near the border of El Salvador and Nicaragua. Also relevant to these observations, and informative regarding BRPE distributed into Central America, are the comments of Angehr & Kushlan (2007), stating, “ornithologically, the Gulf of Panama is the Gulf of Pelicans”; they reported ~5000 breeding pairs and an estimated total population of ~21 000 individuals. This is where a large proportion of Panama’s pelicans and other seabirds breed. But Angehr & Kushlan (2007) also suggested (and more study is needed), “we suspect that this is a taxonomically distinct population.” An “extralimital” subspecies (perhaps new to science) for this area was, as noted above, proposed by Palmer (1962) and originally also questioned by Wetmore (1945). Genetic and ecological studies from southern Mexico and Central America may provide useful information regarding isolating barriers for the two subspecies.

Nonetheless, all available information indicates that the coast of Guerrero (17°40'N) probably represents the southern nesting limits of CABRPE. Small but sporadic and rare nesting colonies may, however, occasionally be found in the zone between the two described subspecies. The two subspecies of the larger region (all of western North America), normally widely separated as breeders, commonly mix as non-breeders (see Thurber *et al.* 1987).

DISCUSSION

The San Lorenzo Archipelago contained the largest colony in the metapopulation, and no other colonies (expressed in various ways in Table 1) reached that level, with considerably smaller colonies more typical. Otherwise, each subpopulation contained at least one or two colonies or aggregates that dominated subpopulation numbers, and perhaps acted as sources: SCB: Anacapa Islands Archipelago plus Santa Barbara Island, 9000 nests (77.0% of the subpopulation); SBP: Isla Santa Margarita, 1950 nests (62.9%); GOC: San Lorenzo and San Luis Archipelagos plus Isla Tortuga, 31 485 nests (72.6%); MME: Archipelago Isla Pájaros (Bahía Santa María), 9050 nests

(85.9%); and MMI: Isla de Pájaros (Mazatlán) plus Isla la Peña, 1350 nests (73.2%) (see the appendices for details).

The range of CABRPE encompasses two Waterbird Conservation Planning Regions, 10–11 Bird Conservation Regions, and two Waterbird Planning Regions within the North American Waterbird Conservation Plan (Kushlan *et al.* 2002). In Mexico, the CABRPE range encompasses more important bird-conservation programs, for example: AICAS (*Áreas de Importancia para la Conservación de las Aves en México*) (<http://conabiweb.conabio.gob.mx/aicas/doctos/aicas.html>) and CONABIO’s *Regiones Marinas Prioritarias* (<http://www.conabio.gob.mx/conocimiento/regionalizacion/doctos/>), plus many other more local CONABIO programs, and at least seven priority sites for wildfowl conservation (Pérez-Arteaga *et al.* 2005). Many of the areas where we have defined BRPE subpopulations, especially along Mexico’s west coasts, are listed as important wetlands in Mexico (Scott & Carbonell 1986:360-372). Wetlands north to the Colorado Delta Region (see Anderson *et al.* 2003) and those few on the coasts of Baja California (see Massey & Palacios 1994) are equally important.

De la Torre (1986) aptly stated that from Nayarit and south (essentially the MMI subpopulation), offshore nesting islands are scarce; those that exist are steep, bare or vegetated, or are heavily occupied by humans and disturbed. He concluded (and we agree, see Anderson *et al.* 1976, 2006) that these smaller, more disturbed, and more scattered breeding colonies are no less important in conservation (Lesica 1995), but perhaps more vulnerable, than the larger colonies and subpopulations to the north (Table 1, Appendices 3–5). Protection is especially critical for still active nesting colonies, such as Isla de Pájaros (near Mazatlán) and Isla Peña. Based on our literature reviews (listed in Appendix 5), we are confident that subpopulation, southern range-periphery numbers in the MMI are very low—perhaps even lower than we estimate—and highly vulnerable.

We have not specifically evaluated trends in this report, but our estimate should be considered similar in size to the earlier, more crude numbers estimate by USFWS (1983:9,176), at 55 000–60 000 pairs, except that the SCB population has increased greatly in numbers since the early 1980s (Gress 1995). The USFWS (1983:11) stated, “The number of pairs breeding in the SCB from 1969 through 1981 ranged from 339 to 3 510 (average = 1 228).” Our 2006 estimate of ~11 700 pairs (Appendix 1, Table 1) indicates an increase over the earlier average numbers by almost one order of magnitude (9.5-fold). If one subtracts the SCB increase (10 470) from our 2006 metapopulation total, the remainder is about 60 200, a value remarkably close to the earlier estimate by FG and DWA in USFWS (1983). Significant historical, ecological changes in the pelagic environment (especially regarding large predatory fish and sea turtles) of the GOC ecosystem (Sagarin *et al.* 2008) began well before the late 1960s, when initial studies of seabirds by others (i.e., Banks 1963b) were first conducted. Therefore, there is little truly historical information for breeding pairs of CABRPE throughout their entire range.

We consider our 2006 estimate as a yearly near-maximum for recent history (post-1960s). Breeding numbers in other years, likely less than in 2006, are in the future likely to be naturally influenced by ENSO as well as by the unnatural effects of human economic development pressures and disturbances (e.g., Anderson & Keith 1980, Tershy *et al.* 1999, Primavera 2005), especially from the

Colorado River delta south to the southern limits of the BRPE range along Mexico's west coast. Maricultural, agricultural and tourist activities in these regions may result in substantial loss of breeding habitat, leading to decreased BRPE and other waterbird populations. The MME subpopulation and its habitat are especially dependent on the persistence of widespread estuarine/mangrove habitats and nesting islands. Much of this habitat also represents hugely significant habitat for wintering waterfowl (shorebirds, ducks, geese, coots). Further descriptions, threats and conservation (along with waterfowl census data) were reported by Saunders & Saunders (1981), Kramer & Migoya (1989), Wilson & Ryan (1997) and Pérez-Arteaga *et al.* (2002).

Our objectives were not to evaluate various census methods, but to provide and document the best estimates available from a variety of approaches and to build a reliable baseline for the future. The increasing resolution of visual satellite imagery (e.g., Taylor & Pollard 2008, Fretwell *et al.* 2012) holds promise for future surveys and monitoring.

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