*J. Field Ornithol.* 85(4):421–429, 2014 DOI: 10.1111/jofo.12081

# Demographic trends of Brown Pelicans in Louisiana before and after the Deepwater Horizon oil spill

Scott T. Walter,[1,4,‡] Michael R. Carloss,[2,†] Thomas J. Hess,[3,§] and Paul L. Leberg[1]

[1]*Department of Biology, University of Louisiana at Lafayette, P. O. Box 42451, Lafayette, Louisiana 70504-2451, USA*

[2]*Louisiana Department of Wildlife and Fisheries, 2000 Quail Drive, Baton Rouge, Louisiana 70808, USA*

[3]*Louisiana Department of Wildlife and Fisheries, Rockefeller Refuge, 5476 Grand Cheniere Hwy, Grand Cheniere, Louisiana 70643, USA*

Received 14 April 2013; accepted 5 September 2014

ABSTRACT. Marine oil spills may have extensive and deleterious effects on coastal waterbirds, but pre-spill data sets are often not available for making comparisons of demographics to the period following a spill. The 2010 Deepwater Horizon oil spill allowed us to compare Brown Pelican (*Pelecanus occidentalis*) demographics during pre- and post-spill years. We banded 1114 pelicans on Louisiana barrier islands from 2007 to 2009, tracked their distribution via band re-sighting surveys from 2008 to 2011, and conducted age-structure surveys. Across Louisiana coastal islands in 2011, we detected 7% of pelicans that had been oiled during the 2010 spill and released following rehabilitation. Similarly, 6% of pelicans (not oiled) banded at the same release site in 2007 were observed across coastal islands 1 yr after banding. We observed variation in proportions of pelicans that were 1, 2, and 3 or more years old among years (2008–2011) and across islands, but little variation could readily be assigned to spill-related mortality. These Brown Pelican demographic trends one year following the Deepwater Horizon oil spill are contrary to other assessments of the impacts of oil contamination on marine birds. However, additional research is required to evaluate potential long-term population trends.

RESUMEN. **Las tendencias demográficas del Pelicano pardo, antes y después del derrame de petróleo Deepwater Horizon**

Los derrames de petróleo en el mar pueden tener amplios y efectos perjudiciales sobre las aves acuáticas en las zonas costeras, pero frecuentemente los conjuntos de datos que se refieren al tiempo antes del derrame no están disponibles para hacer comparaciones demográficas con el periodo que sigue el derrame. En el 2010, el derrame de petróleo Deepwater Horizon nos ha permitido comparar la demografía del Pelicano pardo (*Pelecanus occidentalis*) durante los años, antes y después, del derrame. Entre el 2007 al 2009, colocamos anillos en 1114 pelicanos en las islas de barrera en Luisiana. Luego hemos rastreado la distribución de estos pelicanos a través de re-estudios de avistamiento desde el 2008 al 2011, y realizamos encuestas demográficas sobre la estructura de la edad. En las islas costeras de Luisiana en el 2011, detectamos el 7% de pelicanos que habían sido empetrolados durante el derrame en el 2010, pero que fueron llevados a rehabilitación y liberados cuando terminaron su recuperación. Del mismo modo, el 6% de los pelicanos (no empetrolados) que fueron anillados en el mismo lugar de liberación (en el 2007) fueron observados en las islas costeras, un año después de ser anillados. Observamos variaciones en los porcentajes de pelicanos que tenían uno, dos, tres o más años de edad, entre los anos (2008 – 2011) y a través de las islas. Poca variación podría ser asignada a la mortalidad relacionada con el derrame de petróleo. Estos estudios demográficos sobre el Pelicano pardo, hechos un año después del derrame de petróleo Deepwater Horizon, son contrarias a otras evaluaciones sobre los impactos de la contaminación de petróleo sobre las aves marinas. Sin embargo, se requiere investigación adicional para evaluar las tendencias potenciales a largo plazo de la población del Pelicano pardo.

*Key words:* age, banding, barrier islands, Gulf of Mexico, movement, *Pelecanus occidentalis*

During oil spills, marine birds are commonly contaminated (Piatt et al. 1990, Belanger et al. 2010) and exposure to polyaromatic hydrocarbon oil compounds can cause a range of deleterious health effects, and sometimes kill birds (Balseiro et al. 2005, Alonso-Alvarez et al. 2007, Chen and Denison 2011). Because of these effects, oil spills may shift regional population demographics if the number of contaminated individuals is large (Irons et al. 2000, Votier et al. 2008). For example, following the 1989 Exxon Valdez oil spill in Alaska, post-spill densities of a variety of waterbird species

[4]Corresponding author. scott.t.walter@gmail.com

[†]Current address: Ducks Unlimited, Inc., 218 Rue Beauregard, Suite K, Lafayette, Louisiana 70508, USA.

[‡]Current address: Department of Ecology and Evolutionary Biology, Tulane University, 6823 St. Charles Ave., 400 Lindy Boggs, New Orleans, Louisiana 70118, USA.

[§]Deceased.

were lower in the oiled region than in unoiled regions from 1 to 9 yr following the spill (Irons et al. 2000). Furthermore, recruitment of juvenile waterbirds during years with oil spills was nearly twice that in non-oil-spill years in Wales (Votier et al. 2008).

One method of monitoring the potential impacts of oil contamination on waterbirds is to evaluate demographic trends over time (Nevins and Carter 2003). Depending on the location and timing of oil spills, there may be differential contamination of different age classes or sexes (Votier et al. 2005, Humple et al. 2011). When spills occur near breeding sites, adults foraging in contaminated waters can become coated in oil and may subsequently contaminate eggs or chicks at their nests (Parnell et al. 1984). In North Carolina, hatching rates were lower for oiled Brown Pelican (*Pelecanus occidentalis*) eggs (Parnell et al. 1984). Furthermore, ingestion of crude oil reduced growth rates of nestling Herring Gulls (*Larus argentatus*) and Black Guillemots (*Cepphus grille*) in lab and field experiments in Maine (Peakall et al. 1981) and, compared to the preceding 5 yr, survival of nestling Pigeon Guillemots (*Cepphus columba*) was lower for 1–7 yr in areas impacted by the Exxon Valdez spill in Alaska (Golet et al. 2002). Given these effects on hatching rates and nestling survival, the age structure of regional populations in years subsequent to a spill may be skewed toward more adults and fewer young birds as a result of reduced recruitment of young.

In 2010, the Deepwater Horizon oil spill occurred 80 km off the coast of Louisiana (Crone and Tolstoy 2010). From 20 April to 15 July, >636 million liters (4 million barrels) of oil spilled and contaminated more than 1100 km of coastline, primarily in Louisiana (Camilli et al. 2010, Chen and Denison 2011). As a result, oil carbon entered the marine food web (i.e., plankton, Graham et al. 2010; small fish, Whitehead et al. 2011) and a wide range of marine aquatic invertebrates, fish, birds, and mammals were exposed to the oil (Belanger et al. 2010, Henkel et al. 2012). Because the oil spill occurred during the breeding season for many colonial-nesting seabirds, some breeding adults were also contaminated (United States Fish and Wildlife Service [USFWS] 2011). For example, Brown Pelicans breed on coastal islands in Louisiana from April through August, and >10,000 Brown Pelicans were found oiled in 2010 by the Deepwater Horizon spill (USFWS 2011).

To ameliorate the deleterious effects of oiling, oiled birds are sometimes captured, cleaned, medically treated as needed, then released. However, previous studies have provided conflicting results concerning survivorship of treated birds after such rehabilitation. For example, although average life expectancy of Common Guillemots (*Una aalge*) following oiling and cleaning was 9.6 d (Sharp 1996), de-oiled Cape Gannets (*Morus capensis*) experienced survival rates comparable to those of unoiled individuals 1 yr after cleaning (Altwegg et al. 2008). Although there have been few attempts to rehabilitate oiled Brown Pelicans, no adults oiled and cleaned in California displayed breeding activity during the subsequent 2 yr of monitoring (Anderson et al. 1996). During the Deepwater Horizon oil spill, 612 contaminated Brown Pelicans were captured, rehabilitated, and released at several Gulf coast locations, including at a non-contaminated colony site on Rabbit Island, Louisiana (Selman et al. 2012, E. Miller, unpubl. data; Fig. 1).

Because oil contamination may differentially affect the survival and age structure of seabirds and little is known about the effects of oiling and rehabilitation on immature Brown Pelicans, we collected band re-sighting and age structure data in 2011 to compare with data from surveys conducted from 2007 to 2010 (Walter et al. 2013) to determine if regional age and distribution patterns shifted following the 2010 Deepwater Horizon oil spill. We used data from these pre- and post-oil-spill years to (1) evaluate potential differences in re-sightings of pelicans banded before the spill to sightings of pelicans rehabilitated and released after the spill, and (2) compare age structure trends from years prior to the spill to the year after the spill, and between islands located near the spill and one located outside the region of the coast that experienced the spill.

## METHODS

**Study sites.** In 2011, we conducted surveys for banded birds along the Louisiana coast on Rabbit, Raccoon, Whiskey, Trinity, Wine, Shallow Bayou, Grand Isle, Queen Bess, Bird II, and Mangrove Islands (Fig. 1). We selected these islands because we had pre-oil spill band

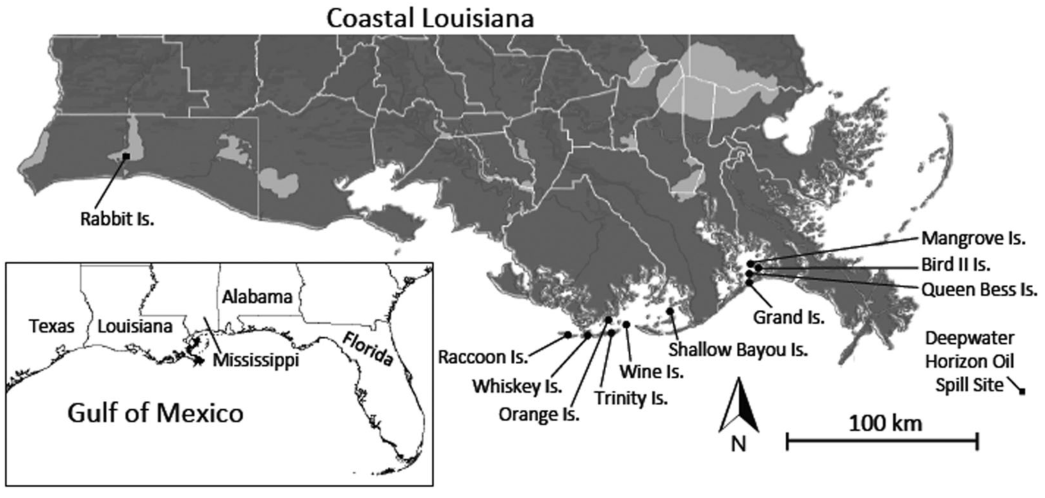


Fig. 1. Location of Louisiana and other coastal states along the Gulf of Mexico (inset), islands in Louisiana used for Brown Pelican research from 2007 to 2011, and the location of the 2010 Deepwater Horizon oil spill.

re-sighting data for some of them (Raccoon, Whiskey, Trinity, and Wine), and the other islands were nearby and accessible given our logistical constraints. All but Rabbit Island received light to heavy oiling as a result of the 2010 Deepwater Horizon oil spill (Michel et al. 2013). Rabbit Island was not surveyed for shoreline oiling, but no oil was reported on monitored shorelines >150 km from the island, suggesting that Rabbit Island was not contaminated during the spill (Michel et al. 2013). Rabbit Island was also considered to be sufficiently far from offshore oil that it was a safe location to release pelicans.

During 2010, we could only work on sites until oil washed ashore. Raccoon Island was oiled on 8 July 2010, and we observed pelicans and other seabirds with oiled feathers. Oil apparently washed ashore at other study sites (Whiskey, Trinity, and Wine Islands) about the same time, but we were asked by personnel from the Louisiana Department of Wildlife and Fisheries to discontinue our sampling at that time and so cannot directly report on oiling levels. Shoreline oiling on Raccoon Island was considered light to moderate in 2010, with oiling levels increasing on islands further to the east (Michel et al. 2013). In 2011, oiling levels at our study sites in southeastern Louisiana varied from trace amounts at Raccoon Island to light to moderate levels on islands to the east (Michel et al. 2013), and there was again no evidence of oil on shorelines near Rabbit Island (Michel et al. 2013). Brown Pelicans were observed resting (i.e., loafing) on all islands in our study, but only nested on Rabbit, Raccoon, Wine, Shallow Bayou, Queen Bess, Bird II, and Mangrove Islands. Pelicans loafed on all the islands we studied regularly starting in 2008, but nested only on Raccoon (2008–2011) and Wine (2008–2010; see Walter et al. 2013) Islands. We have not been able to document the full history of the nesting colonies on islands added to our study in 2011, but all had active colonies in 2011 and at least Rabbit, Shallow Bayou, and Queen Bess Islands had supported colonies for several years (Louisiana Department of Wildlife and Fisheries, unpubl. data).

**Pelican banding and distribution surveys.** For a previous study, we banded Brown Pelican chicks from colonies at Raccoon ($N$ = 575) and Wine ($N$ = 539) Islands each year from 2007 to 2009 (Fig. 1, Walter et al. 2013). We also banded 63 pelican chicks at Rabbit Island in Lake Calcasieu in western Louisiana in 2007 (Fig. 1, Walter et al. 2013). This island is a unique colony site because of its inland location. Each pelican received a U.S. Geological Survey band and an alphanumeric color band. Finally, 182 Brown Pelicans oiled during the Deepwater Horizon spill in 2010 were captured, rehabilitated, and subsequently released at Rabbit Island, Louisiana, between 5 August and 10 September 2010 (E. Miller, unpubl. data; Fig. 1). These individuals (91% were hatch-year birds) received alphanumeric colored leg bands to allow identification (Selman et al. 2012).

To track the movement and distribution patterns of banded Brown Pelicans, we conducted band re-sighting surveys. In prior research from 2008 to 2010, we conducted 200 band re-sighting surveys across Raccoon, Whiskey, Trinity, and Wine Islands that comprise the Isles Dernieres Refuge, and conducted 15 additional surveys on nearby Orange Island in 2010 (Fig. 1; Walter et al. 2013). From 2008 to 2010, surveys were conducted between 26 March and 22 July, and no re-sighting surveys were conducted on Rabbit Island during these years due to logistical constraints. From 18 May to 28 August 2011, we conducted 244 band re-sighting surveys on 10 islands, including Rabbit, Raccoon, Whiskey, Trinity, Wine, Shallow Bayou, Grand Isle, Queen Bess, Bird II, and Mangrove Islands (Fig. 1). During each island visit, we used spotting scopes (20 × 60 power) to scan for banded pelicans at all areas where pelicans were present. Because the legs of pelicans, and therefore potential bands, were typically obstructed when individuals were in vegetation or on nests, our observations were restricted to individuals loafing on beaches (so we did not know if they were breeding or not). We conducted surveys from 07:00 to 18:00 and each survey lasted 1–4 h, sufficient time to scan all pelicans loafing on an island. When a band was identified, we recorded its number and color, the island where detected, and date.

**Age-structure surveys and analyses.** In addition to island-wide surveys for banded pelicans, to assess age structure, we also randomly selected one or two distinct clusters of pelicans (irrespective of whether they were banded or not) during island visits from 07:00 to 18:00. For each cluster, we tallied individuals into the age categories 1, 2, or ≥ 3 yr old based on plumage characteristics (Shields 2002). When two clusters were surveyed on an island on the same day, two surveyors conducted counts at close to the same time on opposite ends of the island to ensure that individual pelicans were counted only once. We conducted 146 surveys from 19 May to 28 June 2011 ($N$ = 4131 individuals) on Rabbit, Raccoon, Whiskey, Trinity, Wine, Shallow Bayou, Grand Isle, Queen Bess, Bird II, and Mangrove Islands (Fig. 1). To perform a temporal comparison of age-structure data collected in 2011, we used data from Walter et al. (2013) from 236 age-structure surveys conducted on Raccoon, Whiskey, Trinity, Wine, and Orange Islands from 26 March to 10 July 2008 to 2010 ($N$ = 7963 individuals).

Although aging 1-, 2-, and 3-yr-old pelicans by plumage characteristics is reasonably accurate (we correctly aged 89% of 231 pelicans surveyed in a previous study; Walter et al. 2013), we used an age-adjustment factor for pelican count data from age-structure surveys. Walter et al. (2013) calculated this adjustment factor by comparing field-estimated ages (based on plumage characteristics) of banded pelicans to their actual ages ascertained later from banding records. With this adjustment factor, we calculated proportions of improperly estimated ages in the field for each age in our age-structure data, and then subtracted the incorrect proportion from each category and added it to the proper age category. For example, if banding records showed that 5% of the pelicans identified in the field as 2-yr-olds were actually 1-yr olds, we would, for our age-structure data, subtract 5% of the count of 2-yr olds and add that count to the 1-yr-old category, and so on for other age categories. This adjustment provided us with more accurate counts of pelicans across age categories. We applied this technique to our raw age structure data in both Walter et al. (2013) and this study. Thus, for counts of 1-yr-olds, we subtracted 16.82% and added it to the 2-yr-old count. For 2-yr-old tallies, we added 4.35% to the 1-yr-old age class and 6.52% to the ≥ 3-yr-old count, and for ≥ 3-yr-old counts, we subtracted 2.56% and added the value to the 2-yr-old count (Walter et al. 2013).

To determine possible relationships between the variables island, age class, and year and the counts of different age classes tallied on Raccoon, Whiskey, Trinity, and Wine Islands from 2008 to 2011, we used log-linear categorical modeling (SAS Institute 2008; PROC CATMOD) to compare all response combination frequencies in the fully saturated and reduced models. Because islands varied in oil exposure in 2010 (none at Rabbit, light to moderate at Raccoon, and moderate to heavy at the other islands) and 2011 (none at Rabbit, trace at Raccoon, and light to moderate at the other islands), and because it was not clear if the occurrence or absence of Brown Pelican breeding colonies on the islands affected age-structure patterns, we also used log-linear categorical modeling to determine if there was an interaction between the variables island and age class that influenced age-structure data at our 10 survey islands in 2011. Because we had access to only a single sample location outside the region of oil exposure, we treated all islands as separate sampling units. This allowed us to determine if the proportion of age classes on Rabbit Island fell outside the range observed across exposed sites.

## RESULTS

**Pelican distribution.** Across our 10 study sites in 2011, we observed 54 of the 1114 individuals (4.8%) banded on Raccoon (575 birds) and Wine (539 birds) Islands from 2007 to 2009. Most pelicans (83.3%) were observed at the same island where we banded them as chicks. In 2011, we also observed 12 of 182 (6.6 ± 1.8 [SE]%) of the rehabilitated pelicans released 1 yr earlier at Rabbit Island in 2010.

**Age structure.** We estimated the age of 12,094 Brown Pelicans from 2008 to 2011 across all study islands, with an average of 34 individuals per cluster (range = 10–223). After adjusting for age-estimation error, across all years and islands, we observed 1.6 times more ≥3-yr-old pelicans ($N$ = 7392) than 1- or 2-yr olds ($N$ = 4702). From 2008 to 2011, counts of birds in each age class were dependent on interactions between island and year, as suggested by the model that included the three-way interaction in log-linear categorical modeling ($\chi^2_{18} = 44.3$, $P < 0.001$, $N = 9456$). Proportions of 1-yr olds were similar across years on individual islands, but differed among islands; Raccoon Island had the lowest proportions (Fig. 2). Furthermore, Wine Island had a smaller proportion of 1-yr olds in 2010 than 2008. Two-yr-old proportions were also similar across years and islands (Fig. 2). Finally, proportions of pelicans 3 or more yr old varied across years and islands (Fig. 2). Raccoon Island had the highest proportion of this age class, yet values were lowest in 2008 compared to subsequent years. Also, proportions of pelicans ≥3-yr old were lower in 2008 than in 2009 and 2010.


Fig. 2. Average proportions of 1-, 2-, and ≥3-yr-old Brown Pelicans counted across surveys on Raccoon ($N$ = 85), Whiskey ($N$ = 63), Trinity ($N$ = 61), and Wine ($N$ = 80) Islands, Louisiana, from 2008 to 2011. We counted distinct clusters of pelicans during surveys (mean = 35 birds per cluster; range = 10–136), and 9456 total individuals were counted.

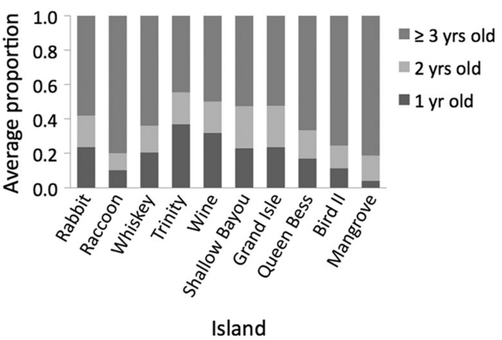


Fig. 3. Average proportions of 1-, 2-, and ≥3-yr-old Brown Pelicans determined from surveys conducted on Rabbit ($N$ = 12), Raccoon ($N$ = 20), Whiskey ($N$ = 14), Trinity ($N$ = 19), Wine ($N$ = 17), Shallow Bayou ($N$ = 20), Grand Isle ($N$ = 5), Queen Bess ($N$ = 17), Bird II ($N$ = 12), and Mangrove ($N$ = 10) Islands in coastal Louisiana in 2011. All islands had pelican nesting colonies except Whiskey, Trinity, and Grand Isle Islands. Rabbit Island received no shoreline oiling. Raccoon Island had light to moderate oiling in 2010 and very light oiling in 2011, whereas other islands had moderate to heavy oiling in 2010 and light to moderate oiling in 2011. We counted distinct clusters of pelicans during surveys (mean = 32 birds per cluster; range = 10–223), and 4131 total individuals were counted.

Counts of pelicans 1-, 2-, and ≥3-yr old across our 10 study sites in 2011 were dependent on an age and island interaction ($\chi^2_{18} = 217.1$, $P < 0.001$, $N = 9456$), as demonstrated by varying age proportions across islands (Fig. 3).

Trinity and Wine Islands had the highest proportion of 1-yr olds and Raccoon Island the smallest, but many of the remaining sites had comparable 1-yr-old proportions to these islands. For 2-yr olds, all islands appeared to have similar proportions. Finally, Raccoon, Whiskey, Queen Bess, Bird II, and Mangrove Islands had higher proportions of pelicans ≥3 yr old than did Rabbit, Trinity, Wine, Shallow Bayou, and Grand Isle Islands. The proportions of pelicans in the three age classes on Rabbit Island, which was not oiled, and Raccoon Island, which was lightly to moderately oiled in 2010 and, along with Wine Island, only very lightly oiled in 2011(based on Michel et al. 2013) fell within the range observed on islands with greater levels of oil exposure.

## DISCUSSION

At 1 yr post-release or banding, we found comparable short-term abundance and distribution patterns for rehabilitated and non-oiled Brown Pelicans. In 2011, we observed 7% of the rehabilitated pelicans released in 2010, similar to the 6% ($N = 4/63$) of pelican chicks banded in 2007 and observed in 2008 (Walter et al. 2013). The lack of difference in band re-sighting suggests survivorship of rehabilitated pelicans was similar to natural mortality, suggesting that rehabilitation was relatively successful. One caveat concerning these observations, however, is that the results reported by Walter et al. (2013) were based on the first-year survival of hatch-year birds. Banded birds released at Rabbit island in 2010 were of mixed ages, but 92% were hatch-year birds.

Beyond survivorship of rehabilitated Brown Pelicans, successful reintegration into the breeding population is what determines the efficacy of cleaning treatments. Anderson et al. (1996) found that rehabilitated Brown Pelicans did not join breeding colonies for 2 yr following treatment. One year following rehabilitation and release, we observed rehabilitated Brown Pelicans at five islands, including four with active breeding colonies. However, we do not know if these pelicans have bred on these islands in subsequent years.

If oil had affected chick production or survival in the Isles Dernieres during 2010, we might have expected to observe lower proportions of 1-yr-old birds in 2011 than in previous years. Likewise, if oil had effected adult survival, we might have expected to see increases in the proportions of 2-yr-old birds in 2011 because they would not have been as close to breeding sites in 2010 as older birds and would not have faced the energetic demands of chick rearing. Although we observed significant variation among years and islands, none of the patterns we observed could be easily explained by declines in fecundity or age-specific mortality in 2010. For example, although there was variation in the proportions of age classes among years, proportions observed in 2011 were with the range observed between 2008 and 2010. Effects of oil on fecundity and age-specific mortality could be mitigated by movements of birds into the study area from other sites, but band sightings suggest pelicans exhibit significant fidelity to natal sites (Walter et al. 2013). Care should be taken in interpretation of our result as a lack of effect on demography because there are combinations of spill-related reductions in fecundity and age-specific mortality that could result in no changes in age structure. Also, oil spills can have unexpected effects on seabird demography. For example, in Wales, recruitment of Common Guillemots (*Uria aalge*) during years immediately following oil spills was nearly double that in non-oil-spill years, perhaps in response to decreased competition due to adult mortality from oil exposure (Votier et al. 2008). This sort of unexpected response makes long-term monitoring of both reproduction and age-structure important to understanding the consequences of oil spills for seabird populations (Votier et al. 2008).

If oil exposure caused differences in fecundity, age-specific mortality, or emigration among islands, we would have expected to see differences in age structure in 2011. We observed significant variation in age structures among the islands, but these differences are not easily explained by oil exposure. The two western-most islands, Rabbit Island with no shoreline oiling and Raccoon Island with light to moderate shoreline oiling in 2010 and trace amounts in 2011, had proportions of juveniles and birds 2-yr-old and older that fell within the range of those observed on islands further to the east. Care must be taken in interpreting our results because oiling of shorelines may not equate directly to exposure to oil by foraging birds. Brown Pelicans typically forage within 20 km of their nesting colonies, but foraging up to 75 km from nesting colonies has been reported (Briggs et al. 1981). Thus, oil observed on an island may not reflect the oil levels at foraging sites. With that caveat in mind, along with the difficulty in quantifying oil slick area and duration in open seas, there is a rough relationship between shoreline oiling and slicks in nearby waters. Using NOAA's Environmental Response Management Application (http://gomex.erma.noaa.gov), there were no oil slicks within 25 km of Rabbit Island, and no locations within 160 km that experienced cumulative oil exposure of more than 1 d. Although the waters adjacent to Raccoon Island experienced cumulative oil slicks lasting 2–5 d, birds would have had to fly 40 km to reach areas of longer cumulative exposure. Areas of cumulative exposure of 6–10 d were located within 20 km of breeding colonies located east of Raccoon Island. Thus, pelicans from Rabbit Island, and perhaps Raccoon Island, may have experienced less oil at loafing and foraging sites than pelicans at breeding colonies located further to the east, but this difference in oil exposure does not easily explain inter-island differences in age structure.

Although shifts in seabird demographic trends have been reported after oil spills at other locations (e.g., Spies et al. 1996, Votier et al. 2008), age and distribution trends for Brown Pelicans in our study were similar prior to and during the year after the Deepwater Horizon oil spill. However, because residual oil remains in an ecosystem well after a spill, longer-term effects on Brown Pelicans are still possible. For instance, hydrocarbons may enter the food chain via uptake by fish, for example, oil was found in salmon (*Oncorhynchus* spp.) tissue during the year of the Exxon Valdez spill (Spies et al. 1996). Also, oil carbon from the Deepwater Horizon spill was assimilated into plankton (Graham et al. 2010) as well as Gulf killifish (*Fundulus grandis*) during the first 4 mo of the spill (Whitehead et al. 2011). Following other oil spills several times smaller than the Deepwater Horizon spill, impacts on populations of some species of piscivorous waterbirds continued to be documented for several years. For instance, for 9 yr following the Exxon Valdez spill, densities of five taxa of marine birds were lower at oiled than non-oiled locations, and persistent oil in the environment was suggested to have slowed recovery of seabird populations (Irons et al. 2000). Irons et al. (2000) attributed some of these long-term effects on density to oil-related changes in mortality and fecundity, and others to movement to other areas. Following the Prestige spill in Spain, European Shags (*Phalacrocorax aristotelis*) switched from their primary demersal and benthic fish diet to one with higher proportions of pelagic and semi-pelagic fish species, probably due to changes in preferred prey availability as a result of the spill (Moreno et al. 2013). Because oil may spread over 750 km from spill sites (Peterson et al. 2003), petroleum hydrocarbon residues can remain in soil substrates for over 30 yr (Reddy et al. 2002) and oil may become incorporated into the food web, additional research on the possible long-term impacts of the Deepwater Horizon oil spill on Brown Pelicans in the northern Gulf of Mexico will be needed beyond our 1-yr post-spill assessment.

Our study was limited to Brown Pelicans located >200 km from the Deepwater Horizon spill site. Results may differ for birds at sites closer to the spill site, such as the Breton National Wildlife Refuge. However, we are unaware of data on oiling levels and pelican demographics in areas closer to the spill site. This lack of data illustrates the importance of widespread and continued studies to assess population metrics within the context of natural and anthropogenic disturbances (Mallory et al. 2010).

## ACKNOWLEDGMENTS

We thank K. Patton, G. Bazer, E. Rosenberg, J. Wood, F. Nebenburgh, M. Smith, J. Bosler, A. White, B. Olinger, A. Dugas, L. Renoux, K. Riley, J. Weibe, S. Longest, T. Cochran, and M. Luent who assisted with data collection and research coordination. Particularly helpful support was provided by D. Fuller, and rehabilitation data were provided by E. Miller. The Louisiana Department of Wildlife and Fisheries and the U.S. Fish and Wildlife Service funded various aspects of this project. Finally, we thank D. Johnson, M. Hester, J. Albert, C. Green, and four anonymous reviewers for suggestions that improved this research.

## LITERATURE CITED

Alonso-Alvarez, C., I. Munilla, M. López-Alonso, and A. Velando. 2007. Sublethal toxicity of the *Prestige* oil spill on Yellow-legged Gulls. Environment International 33: 773–781.

Altwegg, R., R. J. M. Crawford, L. G. Underhill, and A. J. Williams. 2008. Long-term survival of de-oiled Cape Gannets *Morus capensis* after the Castillo de Bellver oil spill in 1983. Biological Conservation 141: 1924–1929.

Anderson, D. W., F. Gress, and D. M. Fry. 1996. Survival and dispersal of oiled Brown Pelicans after rehabilitation and release. Marine Pollution Bulletin 32: 711–718.

Balseiro, A., A. Espí, I. Márquez, V. Pérez, M. C. Ferreras, J. F. G. Marín, and J. M. Prieto. 2005.Pathological features in marine birds affected by the Prestige's oil spill in the north of Spain. Journal of Wildlife Diseases 41: 371–378.

Belanger, M., L. Tan, N. Askin, and C. Wittnich. 2010. Chronological effects of the Deepwater Horizon Gulf of Mexico oil spill on regional seabird casualities. Journal of Marine Animals and Their Ecology 3: 10–14.

Briggs, K. T., D. B. Lewis, W. B. Tyler, and G. L. Hunt, Jr. 1981. Brown Pelicans in southern California: habitat use and environmental fluctuations. Condor 83: 1–15.

Camilli, R., C. M. Reddy, D. R. Yoerger, B. A. S. Van Mooy, M. V. Jakuba, J. C. Kinsey, C. P. McIntyre, S. P. Sylva, and J. V. Maloney. 2010. Tracking hydrocarbon plume transport and biodegradation at Deepwater Horizon. Science 330: 201–204.

Chen, J., and M. S. Denison. 2011. The Deepwater Horizon oil spill: environmental fate of the oil and the toxicological effects on marine organisms. Journal of Young Investigators 21: 84–95.

Crone, T. J., and M. Tolstoy. 2010. Magnitude of the 2010 Gulf of Mexico oil leak. Science 330: 634–643.

Golet, G. H., P. E. Seiser, A. D. McGuire, D. D. Roby, J. B. Fischer, K. J. Kuletz, D. B. Irons, T. A. Dean, S. C. Jewett, and S. H. Newman. 2002. Long-term direct and indirect effects of the 'Exxon Valdez" oil spill on Pigeon Guillemots in Prince William Sound, Alaska. Marine Ecology Progress Series 241: 287–304.

Graham, W. M., R. H. Condon, R. H. Carmichael, I. D'Ambra, H. K. Patterson, L. J. Linn, and F. J. Hernandez, Jr. 2010. Oil carbon entered the coastal plankton food web during the Deepwater Horizon oil spill. Environmental Research Letters 5: 1–6.

Henkel, J. R., B. J. Sigel, and C. M. Taylor. 2012. Large-scale impacts of the Deepwater Horizon oil spill: can local disturbance affect distant ecosystems through migratory shorebirds? BioScience 62: 676–685.

Humple, D. L., H. M. Nevins, E. M. Phillips, C. Gibble, L. A. Henkel, K. Boylan, and D. J. Girman. 2011. Demographics of *Aechmophorus* grebes killed in three mortality events in California. Marine Ornithology 39: 235–242.

Irons, D. B., S. J. Kendall, W. P. Erickson, L. L. McDonald, and B. K. Lance. 2000. Nine years after the Exxon Valdez oil spill: effects on marine bird populations in Prince William Sound, Alaska. Condor 102: 723–737.

Mallory, M. L., S. A. Robinson, C. E. Hebert, and M. R. Forbes. 2010. Seabirds as indicators of aquatic ecosystem conditions: a case for gathering multiple proxies of seabird health. Marine Pollution Bulletin 60: 7–12.

Michel, J., E. H. Owens, S. Zengel, A. Graham, Z. Nixon, T. Allard, W. Holton, P. D. Reimer, A. Lamarche, M. White, N. Rutherford, C. Childs, G. Mauseth, G. Challenger, and E. Taylor. 2013. Extent and degree of shoreline oiling: Deepwater Horizon oil spill, Gulf of Mexico, USA. PLoS ONE 8: e65087.

Moreno, R., L. Jover, C. Diez, F. Sardá, and C. Sanpera. 2013. Ten years after the Prestige oil spill: seabird trophic ecology as indicator of long-term effects on the coastal marine ecosystem. PLoS ONE 8: e77360.

Nevins, H. M., and H. R. Carter. 2003. Age and sex of Common Murres *Uria aalge* recovered during the 1997–98 Point Reyes tarball incidents in central California. Marine Ornithology 31: 51–58.

Parnell, J. F., M. A. Shields, and D. Frierson, Jr. 1984. Hatching success of Brown Pelican eggs after contamination with oil. Colonial Waterbirds 7: 22–24.

Peakall, D. B., J. Tremblay, W. B. Kinter, and D. S. Miller. 1981. Endocrine dysfunction in seabirds caused by ingested oil. Environmental Research 24: 6–14.

Peterson, C. H., S. D. Rice, J. W. Short, D. Esler, J. L. Bodkin, B. E. Ballachey, and D. B. Irons. 2003. Long-term ecosystem response to the Exxon Valdez oil spill. Science 302: 2082–2086.

Piatt, J. F., C. J. Lensink, W. Butler, M. Kendziorek, and D. R. Nysewander. 1990. Immediate impact of the 'Exxon Valdez' oil spill on marine birds. Auk 107: 387–397.

Reddy, C. M., T. I. Eglinton, A. Hounshell, H. K. White, L. Xu, R. B. Gaines, and G. S. Frysinger. 2002. The West Falmouth oil spill after thirty years: the persistence of petroleum hydrocarbons in marsh sediments. Environmental Science and Technology 36: 4754–4760.

SAS Institute. 2008. SAS Enterprise Guide 4.2 for SAS OnDemand. SAS Institute, Inc., Cary, NC.

Selman, W., T. J. Hess, Jr., B. Salyers, and C. Salyers. 2012. Short-term response of Brown Pelicans (*Pelecanus occidentalis*) to oil spill rehabilitation and translocation. Southeastern Naturalist 11: 1–16.

Sharp, B. E. 1996. Post-release survival of oiled, cleaned seabirds in North America. Ibis 138: 222–228.

Shields, M. 2002. Brown Pelican (*Pelecanus occidentalis*). In: The birds of North America online (A. Poole, ed.). Cornell Lab of Ornithology, Ithaca, NY.

Spies, R. B., S. D. Rice, D. A. Wolfe, and B. A. Wright. 1996. The effects of the Exxon Valdez oil spill on the Alaskan coastal environment. American Fisheries Society Symposium 18: 1–16.

United States Fish and Wildlife Service (USFWS) [online]. 2011. U.S. bird impact data and consolidated wildlife reports. <www.fws.gov/home/dhoilspill/collectionreports.html> (Accessed 14 February 2012).

Votier, S. C., B. J. Hatchwell, A. Beckerman, R. H. McCleery, F. M. Hunter, J. Pellatt, M. Trinder, and T. R. Birkhead. 2005. Oil pollution and climate have wide-scale impacts on seabird demographics. Ecology Letters 8: 1157–1164.

———, T. R. Birkhead, D. Oro, M. Trinder, M. J. Grantham, J. A. Clark, R. H. McCleery, and B. J. Hatchwell. 2008. Recruitment and survival of immature seabirds in relation to oil spills and climate variability. Journal of Animal Ecology 77: 974–983.

Walter, S. T., M. R. Carloss, T. J. Hess, G. Athrey, and P. L. Leberg. 2013. Movement patterns and population structure of the Brown Pelican. Condor 115: 788–799.

Whitehead, A., B. Dubanksy, C. Bodinier, T. I. Garcia, S. Miles, C. Pilley, V. Raghunathan, J. L. Roach, N. Walker, R. B. Walter, C. D. Rice, and F. Galvez. 2011. Genomic and physiological footprint of the Deepwater Horizon oil spill on resident marsh fishes. Proceedings of the National Academy of Sciences USA 109: 20298–20302.