

**Caribbean Roseate Tern and North Atlantic Roseate Tern**  
**(*Sterna dougallii dougallii*)**

**5-Year Review:**  
**Summary and Evaluation**



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## **5-YEAR REVIEW**

### **Roseate Tern (*Sterna dougallii dougallii*)**

#### **1.0 GENERAL INFORMATION**

##### **1.1 Reviewers**

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###### **Cooperating Field Offices/Stations:**

###### Region 5

Long Island/New York Field Office  
Maine Field Office  
Monomoy National Wildlife Refuge  
Steward B. McKinney National Wildlife Refuge  
Petit Manan National Wildlife Refuge

###### Region 4

Raleigh, North Carolina Field Office  
Athens, Georgia Field Office  
Vero Beach, South Florida Field Office  
Florida Keys National Wildlife Refuge

###### **Cooperating Regional Office:**

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##### **1.2 Methodology Used to Complete the Review**

The roseate tern (ROST) 5-year review was compiled by Michael Amaral, the lead recovery biologist for the endangered Northeast roseate tern population (ROST-NE) and Jorge E. Saliva, lead recovery biologist for the threatened Caribbean population (ROST-CA). Section 2.3 of the review, Updated Information and Current Species Status, was prepared by M. Amaral and J.E. Saliva in collaboration with Ian C.T. Nisbet, ROST-NE recovery team member, on contract to the U.S. Fish and Wildlife Service (USFWS). USFWS field offices, national wildlife refuges (NWRs), and Federal and State natural resource agency personnel responsible for the recovery of

this species were contacted for the most current information on occurrences, threats, and recovery activities. Academic researchers, non-Governmental organizations (NGO), and other biologists conducting research on the ROST were also contacted. Figure 1 was prepared by Robert Houston of the USFWS Gulf of Maine Coastal Program. Carolyn Mostello, tern project leader for Massachusetts Division of Fisheries and Wildlife (MDFW), prepared Tables 1 and 2 and Figure 2. Ellen Jedrey of the Coastal Waterbird Program of the Massachusetts Audubon Society prepared Figure 3, and Jeffrey Spendelov of the U.S. Geological Survey, Patuxent Wildlife Research Center, provided valuable insights and editorial improvements.

In addition to relevant information from the recovery plans for the Northeast population (USFWS 1998) and Caribbean population (USFWS 1993), the principal basis for this review includes scientific literature published since the respective recovery plans were completed; additional information provided by State, NGO, and USFWS biologists; and other information compiled during annual meetings of the ROST-NE recovery team. The Caribbean population does not have a designated recovery team; instead, information is maintained and recovery efforts are coordinated by the USFWS Caribbean Field Office.

This review was also facilitated by publication of the Recovery Strategy for the Roseate Tern (*Sterna dougallii*) in Canada (Environment Canada 2006), the COSEWIC Assessment and Update Status Report on the Roseate Tern in Canada (COSEWIC 2009), the Tern Management Handbook – Coastal Northeastern United States and Atlantic Canada (Kress and Hall 2004), and the Draft Environmental Impact Report on the New Bedford Harbor Tern Restoration Project – Roseate Tern Nesting Habitat Enhancement at Bird Island in Marion, Massachusetts (MDFW 2002).

### 1.3 Background

**1.3.1 FR Notices announcing initiation of this review:** 73 FR 76373 (December 16, 2008) Notice of Endangered and Threatened Wildlife and Plants; Initiation of 5-Year reviews of 7 listed species. 75 FR 17153 (April 5, 2010), Notice of Status Review of Roseate Tern; Request for information and clarification. The April 5, 2010, FR expanded the scope of the 5-year review to include the Caribbean population.

#### 1.3.2 Listing history:

**FR notice:** Determination of Endangered and Threatened Status for Two Populations of the Roseate Tern, 52 FR 42064

**Date listed:** November 2, 1987

**Entity:** Populations of the subspecies, *Sterna dougallii dougallii*

**Classification:** Endangered – Northeast Population: United States, Atlantic Coast south to North Carolina, Canada, (Newfoundland, Nova Scotia, Québec) Bermuda. Threatened – Caribbean population: Western Hemisphere and adjacent oceans, including United States (Florida, Puerto Rico, Virgin Islands) where not listed as endangered; Canada – designated as endangered by the Committee on the status of endangered wildlife in Canada (1999) and listed as endangered under the Species at Risk Act (SARA, section 37) in 2003.



### **1.3.3 Associated rulemakings:**

47 FR 58454 (December 30, 1982), Review of Vertebrate Wildlife for Listing as Endangered or Threatened Species.

**1.3.4 Review history:** The ROST-NE and ROST-CA were included in a cursory 5-year review conducted for all species listed before 1991 (56 FR 56882). Although no other 5-year reviews have been completed for these populations, an extensive status and literature review was conducted prior to the first update of the recovery plan for the Northeast population, completed in 1998, and the recovery plan for the Caribbean population, completed in 1993. Since the completion of the Northeast population recovery plan update, Environment Canada (2006) issued a Recovery Strategy for the portion of the Northeast population that breeds in Atlantic Canada. Environment Canada (2006) provides a more current assessment of status, threats, and recommended actions to effect recovery of the roseate population breeding there.

### **1.3.5 Species' Recovery Priority Number at start of 5-year review:**

The recovery priority number for both listed roseate tern populations is 3, indicating a high degree of threat and high recovery potential for this subspecies.

### **1.3.6 Recovery plans:**

**Name of plan:** Roseate Tern (*Sterna dougallii*) Recovery Plan, Northeast Population, First Update

**Date issued:** November 5, 1998

**Date of previous plan:** 1989

**Name of plan:** Caribbean Roseate Tern (*Sterna dougallii*) Recovery Plan

**Date issued:** September 24, 1993

**Date of previous plan:** N/A

**Name of plan:** Recovery Strategy for the Roseate Tern (*Sterna dougallii*) in Canada

**Date issued:** October 2006

**Date of previous plan:** 1993

## **2.0 REVIEW ANALYSIS**

### **2.1 Application of the 1996 Distinct Population Segment (DPS) Policy**

#### **2.1.1 Is the species under review a vertebrate?**

Yes. The DPS policy is, therefore, applicable.

### 2.1.2 Is the DPS policy applicable?

Yes. Section 3 of the Endangered Species Act of 1973 (ESA) defines “species” to include subspecies and “any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature (emphasis added).” In 1996, the USFWS and National Marine Fisheries Service (Services) published a joint policy guiding the recognition of DPSs of vertebrate species (61 FR 4722). As the roseate tern is a vertebrate that was listed prior to 1996, we evaluate evidence for DPSs within the subspecies *S. dougallii dougallii* (see 61 FR 4724: “The Services maintain that the authority to address DPSs extends to species in which subspecies are recognized ...”). The DPS policy specifies three elements to assess whether a population segment under consideration for listing may be recognized as a DPS: (1) The population segment’s discreteness from the remainder of the species to which it belongs, (2) the significance of the population segment to the species to which it belongs, and (3) the population segment’s conservation status in relation to the ESA’s standard for listing (61 FR 4722).

The Northeast and Caribbean populations of the roseate tern were treated as distinct entities at the time of listing in 1987, when the Northeast population was listed as endangered and the Caribbean population was listed as threatened. However, the 1987 listing predated the Services’ 1996 DPS policy (61 FR 4722). There are other non-listed populations of the subspecies (*S. d. dougallii*) that breed in the British Isles, northwest France, and the Azores and Canary Islands and along the south and east coasts of Africa (Gochfield *et al.* 1998). The following discussion briefly summarizes the basis for retaining the two populations on the list of threatened and endangered wildlife based on evidence that they are both discrete and significant to the status of the species as a whole.

#### Discreteness:

A vertebrate population segment may be considered discrete if it satisfies either of the following two conditions:

1. It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors. Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation.
2. It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the ESA.

The Northeast and Caribbean populations are separated by a gap of about 1,600 kilometers (nearly 1,000 miles) between the Bahamas and eastern Long Island, New York. A substantial separation has existed since the earliest historical records of the species in the nineteenth century (Nisbet 1980, Gochfeld *et al.* 1998) and has become wider in recent decades with the continuing northward contraction of the range of the Northeast population. Roseate terns were last recorded breeding in Virginia and Maryland in the 1930s and in New Jersey in the 1970s, and have virtually disappeared from the south shore of Long Island during the 2000s (see NE section 2.3.1.5).

No genetic studies of the Caribbean population have been carried out, so no information is available regarding genetic divergence between the two populations. There is some potential for interchange of individual birds between the two populations, because they are known to winter at the same sites in eastern Brazil (Hays *et al.* 1999), and marked birds from the Northeast population have been identified on migration in Puerto Rico in May, August, and September (J. Spendelow, unpubl. data; J.E. Saliva, pers. comm.). However, there is little evidence for actual interchange of breeders. Hays *et al.* (1999) listed four sight-records at northeast colonies of color-banded birds that had apparently been marked in Caribbean colonies, but they regarded these records as inconclusive: “However, because of the possibility of misbanding or misreading it is essential to read the band numbers or trap the birds wearing foreign color combinations to provide firm evidence of Caribbean birds in northern nesting colonies.” No such evidence has been obtained, despite intensive trapping and re-sighting of roseate terns throughout the northeast breeding range from 1988 to the present-day (see NE section 2.3.1.2.1). Hence, interchange of breeding individuals between the two populations, if any, must be very small.

Apart from genetic divergence, birds from the two populations differ in a number of morphological and ecological characteristics. They differ markedly in the coloration of the bill during the breeding season, i.e., northeastern birds have the bill entirely black until about the time of chick hatching, when they start to develop red at the base, whereas Caribbean birds rarely have an entirely black bill, even at the start of the breeding season, and the bill becomes two-thirds red by the time of egg-laying (Gochfeld *et al.* 1998). Although no critical comparisons have been published, Caribbean birds appear to be smaller in linear measurements and body-masses (Gochfeld *et al.* 1998). They also lay smaller eggs (Nisbet 1981; see CA sections 2.3.1.2.5 and 2.3.1.2.6), generally lay smaller clutches (see CA section 2.3.1.2.3), their chicks grow more slowly and reach smaller asymptotic masses (see CA sections 2.3.1.2.5 and 2.3.1.2.6), and average productivity is much lower (see CA section 2.3.1.2.5; Nisbet and Ratcliffe 2008). Caribbean birds usually nest on smaller islands, often with little or no vegetation, and their nests are usually in the open, whereas Northeast birds usually nest under cover (Gochfeld *et al.* 1998). Caribbean birds feed on a completely different range of fish from those eaten by northeastern birds (Gochfeld *et al.* 1998) and appear to be more dependent on feeding on predatory fish (Shealer 1995), whereas northeastern birds are out-competed by common terns (*Sterna hirundo*) over schools of predatory fish (Duffy 1986, Safina 1990).

The Northeast population is also largely discrete from the population in northwest Europe (which is itself small, endangered, and intensively managed; Cadiou 2010). Although genetically similar (Lashko 2004), these two populations are separated by >3,000 km (1,860 mi) at their closest breeding sites (Cape Cod–Azores), and there is very little evidence for exchange of individual birds between the breeding populations (see NE section 2.3.1.3). The Northeast population is very similar to the northwestern European population in all the characteristics mentioned in the previous paragraph, and both are distinct from the Caribbean population in these characteristics (Gochfeld *et al.* 1998, Ratcliffe *et al.* 2005). The Northeast population has low genetic diversity (Szczyś *et al.*

2005a), perhaps reflecting recent population bottlenecks and indicating that it may be at high risk of undergoing catastrophic population declines in the future.

Based on the preceding factors, the Caribbean and Northeast populations are discrete from one another and are discrete from other populations of the subspecies *S. dougallii dougallii*. Condition 1 of the 1996 DPS policy is thus met.

Significance:

Under the 1996 DPS policy, if a population segment is determined to be discrete, we then consider its biological and ecological significance relative to the larger taxon to which it belongs (61 FR 4722). This consideration may include, but is not limited to, the following factors:

1. Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon,
2. Evidence that loss of the discrete population segment would result in a significant gap in the range of the taxon,
3. Evidence that the discrete population segment represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historic range, or
4. Evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics.

Based on the best information available, both the Northeast and Caribbean populations of the roseate tern are “significant.” The Caribbean population is the only tropical population of roseate terns in the North or South Atlantic Oceans and, hence, the only tropical population of the subspecies *S. d. dougallii* (Lashko 2004). If it were lost, recolonization from the tropical populations in the Indian or Pacific Oceans would be extremely unlikely because of the great distances (>10,000 km) and the land barriers posed by Africa and Central America; in any case, the birds would come from populations that are likely to be genetically and morphologically distinct (Lashko 2004). Recolonization from the Northeast population would be hypothetically possible, but very slow, and from a population that is morphologically somewhat distinct and ecologically very distinct (see discreteness discussion above), so that any colonizers would be poorly adapted to Caribbean conditions.

The Northeast population is one of six discrete temperate populations of a species that is otherwise primarily tropical (Nisbet and Ratcliffe 2008) and is one of only two temperate populations of *S. d. dougallii* (the third population, in South Africa, is extremely small and genetically intermediate between the two subspecies; Lashko 2004). If the Northeast population were lost, recolonization from the northwestern European population (which is very similar genetically, morphologically, and ecologically) would be theoretically possible, but very slow based on the limited evidence for interchange summarized in NE section 2.3.1.3. In any case, the northwestern European population is very small and even more endangered than the Northeast population, so cannot be relied on as a

refugium or gene pool for the combined temperate populations of the North Atlantic. Recolonization from the Caribbean population would be unlikely for the reasons set out in the previous paragraph.

Loss of either population would result in a significant range contraction. Loss of both the Northeast and Caribbean populations would mean that there would be no representation of the species in the Americas. The best available scientific information thus supports recognition of two listable roseate tern entities, a Northeast DPS and a Caribbean DPS, consistent with the ESA definition of “species”.

**Note:** In the following sections of the Review Analysis, the endangered Northeast population and threatened Caribbean population are addressed separately.

## **ENDANGERED NORTHEAST POPULATION**

### **2.2 Recovery Criteria**

#### **2.2.1 Does the species have a final, approved recovery plan containing objective, measurable criteria?**

Yes. The 1998 recovery plan for the ROST-NE contains objective and measurable recovery criteria. It should also be noted that Environment Canada (2006) issued a recovery strategy for the roseate tern in Canada. About 120 to 150 pairs of roseate terns breed in Canada, where the species is similarly listed as endangered. The Environment Canada recovery strategy also contains objective and measurable criteria but acknowledges that with only 5 percent of the northeastern breeding population, recovery in Canada relies heavily on the recovery of the population in its northeastern U.S. range.

#### **2.2.2 Adequacy of recovery criteria:**

##### **2.2.2.1 Do the recovery criteria reflect the best available and most up-to date information on the biology of the species and its habitat?**

The recovery criteria may need revision pursuant to information that has become available since preparation of the original and updated recovery plans for the Northeast population. It has been 12 years since the last comprehensive review of this population was completed (USFWS 1998b) and 21 years since the recovery criteria were initially developed (USFWS 1989). The primary objective ( $\geq 5,000$  nesting pairs distributed among six or more sites with 200+ pairs, and productivity of  $\geq 1.0$  chicks per pair) remains a challenging but achievable goal. However, the secondary objective (expand the number of breeding colonies to 30 or more sites and expand breeding range to historically occupied areas south of the current range) and, in particular, the delisting objective (increase the population to its historical high of 8,500 pairs) do not appear realistic or achievable.

Over the past two decades, a comprehensive program comprising colony stewardship, nesting habitat enhancement, predator management and population monitoring, has been implemented at all major and most of the smaller breeding colonies; however, the recovery goal of increasing the population to historical levels has proven elusive, because threats have not been significantly reduced. The Northeast population has only briefly exceeded 4,000 (peak season<sup>1</sup>) nesting pairs (1999 to 2000), and only three colonies have consistently supported 200 or more roseate nesting pairs. Thus, the goal of reaching 5,000 breeding pairs distributed among six large colonies has not been met. However, the productivity goal of 1.0 chicks per pair does appear achievable, as productivity at major colonies often exceeds 1.0 chicks/pair (Nisbet and Spendelov 1999, Roseate Tern Recovery Team [RTRT] 2009). It should be noted that recent data suggest a regionwide reduction in productivity since 2000, for unknown reasons (see NE section 2.3.1.2.3); nonetheless, productivity at the major colonies has still exceeded 1.0 in most years (see NE section 2.3.1.2.4).

USFWS policy (USFWS 1990) describes “recovery” as the process by which the decline of an endangered or threatened species is arrested or reversed and threats to its survival are neutralized, so that its long-term survival in nature can be ensured. The 8,500 breeding pair goal is based on an estimate of abundance during the 1930s (Nisbet 1980), and it is unknown whether this historic high can be attained even if contemporary anthropogenic limiting factors can be addressed. It is also unusual to identify the highest population level estimated for a species (or, in this case, a population) as a recovery criterion. On this basis, the recovery goal to restore the Northeast population to a historical high of 8,500 breeding pairs warrants review.

**2.2.2.2 Are all of the 5 listing factors relevant to the species addressed in the recovery criteria (and is there new information to consider regarding existing or new threats)?**

No. The five listing factors are not explicitly addressed in the recovery criteria. The recovery criteria described below include numerical, distributional, productivity and persistence targets, but these offer only an indirect measure of whether the five listing factors have been addressed. However, it is highly unlikely that the recovery objectives can be met without successfully mitigating the suite of threats facing the species. For example, the population will not increase to 5,000 pairs if predation, disease, or overutilization are a significant problem, and the distribution target of six large colonies will not be met if competition by gulls or other factors limiting nesting habitat availability, such as erosion, remain serious problems. New information that warrants consideration in regard to the listing factors is discussed in NE section 2.3.2.

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<sup>1</sup> Peak period estimates are based on nest counts on or about the date of first hatching (typically June 10-20).

**2.2.3 List the recovery criteria as they appear in the recovery plan, and discuss how each criterion has or has not been met, citing information:**

Primary Objective: To increase the northeast nesting population of roseate terns (United States and Canada) to 5,000 breeding pairs. This total should include at least six large colonies ( $\geq 200$  pairs) with high productivity ( $\geq 1.0$  young fledged per pair for 5 consecutive years) within the current geographic distribution. This will reduce the possibility of extirpation of the Northeast population.

Secondary Objective: (1) To expand the number of roseate tern breeding colonies to 30 or more sites; (2) to expand the breeding range to historically occupied areas south of the current range.

Reclassification Objective: Evaluate with regard to reclassifying from Endangered to Threatened when the primary objective is met.

Delisting Objective: Delisting will be warranted if the nesting population reaches the historic high level of the 1930s, 8,500 pairs.

The primary objective, to increase the Northeast nesting population of roseate to 5,000 breeding pairs, has not been met. Only in a single year, 2000, when about 4,310 peak season pairs and 4,850 total season pairs were recorded, has the population estimate approached the 5,000-pair objective (see NE section 2.3.1.2.2).

The primary objective goes on to state that the total should include at least six large colonies with high productivity within the current geographic distribution. This condition has been partially met. Although only three colonies have consistently supported 200 or more roseate nesting pairs since 1998, productivity at specific large colonies during the past decade has often exceeded 1.0 chicks fledged per pair (Nisbet and Spendelov 1999; see NE section 2.3.1.2.4).

The first condition under the secondary objective, to expand breeding colonies to 30 or more sites, has not been met. Since 1998, between 13 and 21 sites in the United States have supported breeding pairs of roseate terns, with only an additional 3 to 4 sites in Atlantic Canada supporting more than a few pairs. Overall, the number of breeding sites occupied by the population since 2000 has declined by more than 30 percent.

The second condition of the secondary objective, to expand the breeding range to historically occupied areas south of the current range, is also unmet. In fact, since 1998 there has been a further contraction of the breeding range to the north, and fewer formerly occupied sites along southern Long Island are now active. For example, Gardner's Island/Cartwright Point supported more than 150 pairs from 2002-2004 and in 2007, but fewer than 30 pairs in 2009 (RTRT 2009).

## **2.3 Updated Information and Current Species Status**

### **2.3.1 Biology and habitat:**

#### **2.3.1.1 New information on the species' biology and life history:**

The updated recovery plan for the Northeastern population (USFWS 1998b) summarized information available through 1997. In addition, a comprehensive monograph on the roseate tern in North America (covering both the Northeast and Caribbean populations) was published in the *Birds of North America* series (Gochfeld *et al.* 1998). Thus, this section (2.3.1) focuses on new information on the Northeast population that has become available since 1998. This includes the Kress and Hall (2004) *Tern Management Handbook*, which covers all species of terns nesting in the northeastern coastal United States and Atlantic Canada, including roseate terns. The handbook includes detailed summaries of the history of tern populations in these regions, as well as extensive information on limiting factors and management.

#### **2.3.1.2 Abundance, population trends, demographic features, and/or demographic trends:**

##### **2.3.1.2.1 Metapopulation Project**

The Northeast population of roseate terns has been monitored and studied intensively since 1987. The largest component of these efforts is the Cooperative Roseate Tern Metapopulation Project (CRTMP), organized by the U.S. Geological Survey's Patuxent Wildlife Research Center (PWRC) and directed by Dr. J. Spendlow. The project has focused on the six largest colonies of roseate terns south and west of Cape Cod. This initially included Cedar Beach (New York), Falkner Island (Connecticut), Great Gull Island (New York), and Bird Island (Massachusetts). Cedar Beach subsequently was dropped after it was abandoned by roseate terns in 1995, and Ram Island (Massachusetts) and Penikese Island (Massachusetts) were incorporated into the project in 1995 and 2003, respectively. See Figure 1 for locations.

The CRTMP is broadly designed to characterize the structure and dynamics of the roseate tern "metapopulation", with the long-term goal of constructing population models that will guide management decisions. The CRTMP has included a large-scale program of (1) Marking adults and chicks with Bird-Banding Laboratory (BBL) bands and, since 1992, field-readable (FR) bands; (2) marking adults with combinations of three or four color-bands; (3) intensive re-sighting of individually marked adults both at breeding colonies and at staging sites; (4) measurement of productivity and other breeding parameters at each site in each year; and (5) a number of ancillary studies. Although not formally part of the CRTMP, similar but less systematic monitoring, banding, and re-sighting

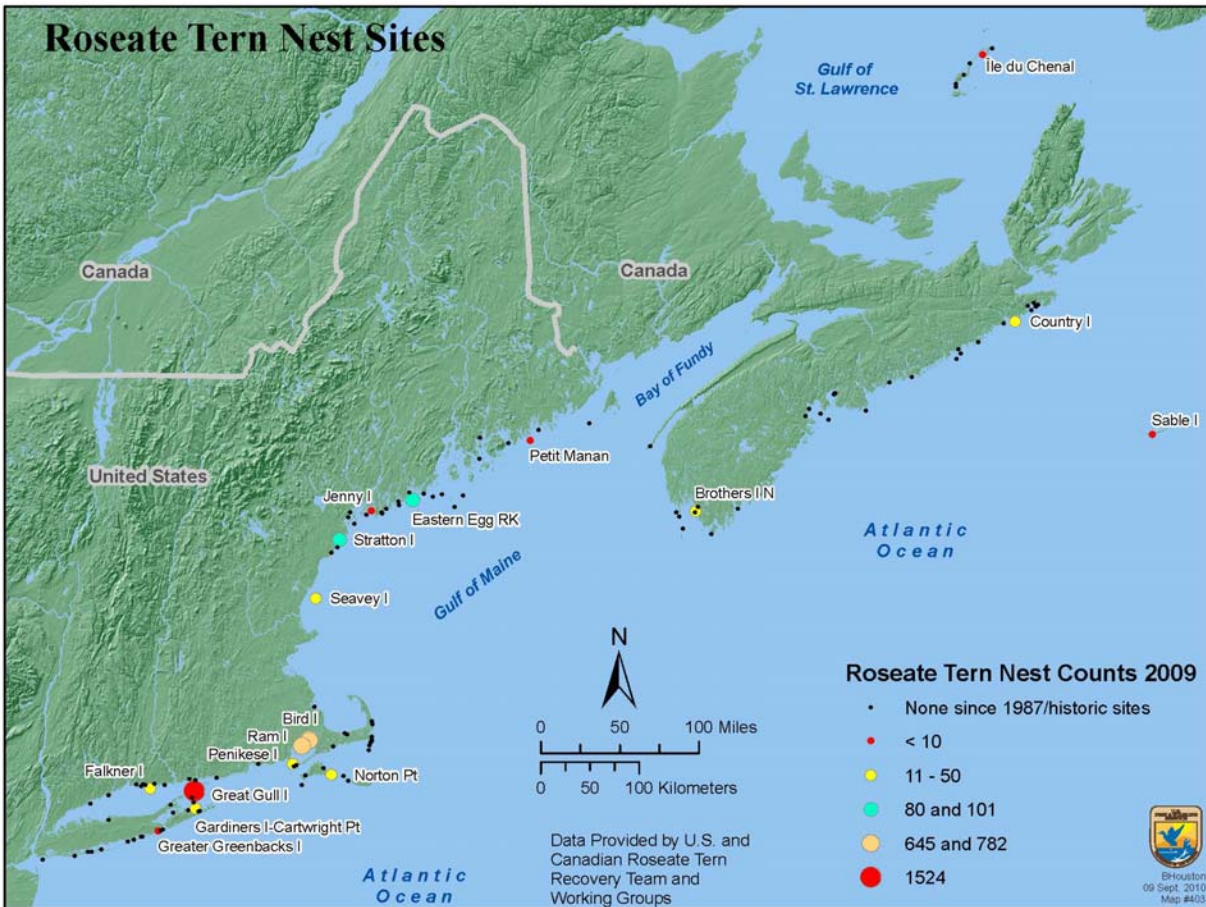


activities have been carried out at most of the other important breeding sites, including most sites in the Gulf of Maine and two sites in Canada. The only colony of more than 50 pairs where no such activities have been carried out is Gardiner's Island/Cartwright Point, New York.

Results from the first 12 years of the CRTMP were summarized by Nisbet and Spendelow (1999), and the mark-recapture component was described by Spendelow *et al.* (1995, 2008). Currently, about 75 percent of the entire regional population is marked with BBL and FR bands, about 40 percent of the adults are marked with individual color-combinations, and about 60 percent of adults are trapped or re-sighted annually (J. Spendelow, unpubl. data).

On the basis of the results available through 1998, Nisbet and Spendelow (1999) divided the regional population into two subregional groups: the "warm water" group south and west of Cape Cod, and the "cold water" group north and east of Cape Cod, including Canada. The warm water subregion includes about 90 percent of the total population and has been intensively studied under the CRTMP. The cold water subregion differs in a number of characteristics, e.g., it is more widely scattered among a number of small colonies, and the birds appear to have somewhat different diets and lower average productivity. Although not formally studied by mark-recapture methods, interchange of individual birds between the two groups is thought to be less than dispersal among colony sites within each group (Nisbet and Spendelow 1999; Spendelow *et al.* 2008, 2010).

**Figure 1.** Distribution of the Northeast population of the Roseate Tern



#### **2.3.1.2.2 Abundance and population trends**

The entire range of the Northeast roseate tern population has been surveyed since 1988, either every year (Massachusetts, New Hampshire, Maine and the largest colony sites in New York, Connecticut, and Nova Scotia [NS]), in alternate years (remainder of New York), or sporadically (remainder of Connecticut and Nova Scotia). Except for some islands in Gardiner's Bay (east Long Island, New York) that have been occupied sporadically and may have been missed in some years, it is believed that almost the entire population is located and counted at least biennially. Methods of estimating numbers of breeding pairs have been improved and standardized, and validated methods are now used at almost all sites. In most cases, a single nest count is made at the end of the "peak period" of nesting (determined for each site as the interval between the date when the first nest is started and a date early in the hatching period 23 to 28 days later). The nest count is adjusted for potential under- or over-counting, in most cases by resurveying part of the nesting area but in other cases by subjective judgment. The adjusted "peak period" counts are believed to be accurate to  $\pm 5$  to 10 percent, except at the site of the largest colony at Great Gull Island, New York, where most nests are concealed under boulders and many are likely to be missed on the surveys. At this site, adjustment factors are typically subjective and are around +20 percent (G. Cormons, unpubl. data).

At most sites, a second estimate is made of the number of nests established after the peak period, but this estimate is usually subjective rather than a precise count because most colony sites have dense vegetation by the end of the season, so that late nests are difficult to find. Consequently, "total season" estimates are less precise and less comparable among sites than "peak period" counts, and primary weight is therefore placed on "peak period" counts in assessing population changes. Total season estimates are typically 5 to 20 percent larger than peak period counts: part of the difference represents young birds 2 to 4 years old nesting for the first time, and part represents pairs re-nesting after having failed earlier, sometimes at another site. From a demographic standpoint, the total number of pairs (including female-female pairs and trios; see NE section 2.3.1.2.6) that establish nests during the season lies between the peak period and total season values, but the effective number of breeding pairs (those that breed successfully and contribute to future generations) is closer to the peak period value. Tables 1 and 2 summarize peak period and total season counts for the cold water and warm water subregions, respectively, for each site in each year that it was known to be occupied. These tables are based on data compiled by C. Mostello of the Massachusetts Division of Fisheries and Wildlife from reports supplied each year by colony managers and biologists and provided to the Northeast Roseate Tern Recovery Team.

**Table 1a.** Peak period estimates of roseate tern pairs in the cold water subregion of the northeastern United States, 1998 to 2009. nd = no data (likely to have been present in at least some years). Source: Carolyn Mostello (MDFW).

Site (listed from north to south)	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Petit Manan I., Maine (ME)	19	28	15	16	27	31	29	9	23	5	4	4
Seal I., ME	1	1	1	1	0	0	0	0	0	0	0	0
Matinicus Rock, ME	0	0	0	0	0	0	0	0	0	0	1	0
Metinic I., ME (North end)	0	0	0	0	0	3	1	0	2	0	0	0
Eastern Egg Rock, ME	144	149	165	145	160	163	110	136	113	118	129	101
Pond I., ME	0	0	0	0	0	2	12	1	1	0	0	0
Jenny I., ME	8	10	0	0	0	0	2	11	14	16	2	3
Outer Green I., ME	0	0	0	0	0	0	8	36	6	7	0	0
Stratton I., ME	86	100	104	127	98	40	11	2	84	79	67	76
Seavey I., New Hampshire (NH)	0	1	1	0	8	42	107	64	33	52	37	34
Plymouth Beach, Massachusetts (MA)	12	0	0	0	0	0	0	0	0	0	1	0
Gray's Beach, Yarmouth, MA	0	1	0	1	0	0	1	0	0	0	0	0
Nauset New I., Eastham, MA	3	3	3	3	0	0	0	0	0	0	0	0
Minimoy I., Chatham, MA	Nd	nd	nd	nd	nd	10	24	29	24	43	33	0
S. Monomoy I., Chatham, MA	22	27	3	6	3	3	1	1	2	2	0	0
TOTAL	295	320	292	299	298	294	306	289	302	322	274	218
Number of sites	8	9	7	7	5	8	11	9	10	8	8	5

**Table 1b.** Total season estimates of roseate tern pairs in the cold water subregion of the northeastern United States, 1998 to 2009. nd = no data. Note: Total season and peak period estimates were not supplied separately for most sites in Maine prior to 2004. Source: Carolyn Mostello (MDFW).

Site (listed from north to south)	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Petit Manan I., ME	19	28	15	16	27	31	29	9	23	5	4	4
Seal I., ME	1	1	1	1	0	0	2	1	1	0	0	0
Matinicus Rock, ME	0	0	0	0	0	0	0	1	0	0	1	0
Metinic I., ME (North end)	0	0	0	0	0	3	1	0	2	0	0	0
Eastern Egg Rock, ME	144	149	165	145	160	163	130	146	113	118	130	101
Pond I., ME	0	0	0	0	0	2	15	1	1	0	0	0
Jenny I., ME	8	10	0	0	2	2	5	11	14	16	2	3
Outer Green I., ME	0	0	0	0	0	0	13	42	6	8	0	0
Stratton I., ME	86	100	109	130	98	40	15	2	90	83	67	80
Seavey I., NH	0	1	1	0	25	63	112	67	38	57	40	40
Plymouth Beach, MA	12	0	0	0	0	0	0	0	0	0	0	0
Gray's Beach, Yarmouth, MA	0	1	0	1	0	0	1	0	0	0	0	0
Nauset New I., Eastham, MA	3	3	4	4	0	0	0	0	0	0	0	0
Minimoy I., Chatham, MA	nd	nd	nd	nd	nd	15	26	30	27	56	37	0
S. Monomoy I., Chatham, MA	40	32	3	6	3	3	1	1	2	2	0	0
TOTAL	313	325	298	303	315	322	350	311	317	345	279	228
Number of sites	8	9	7	7	5	9	12	11	11	8	8	5

**Table 2a.** Peak period estimates of roseate tern pairs in the warm water subregion of the northeastern United States, 1998–2009. nd = no data. Source: Carolyn Mostello (MDFW).

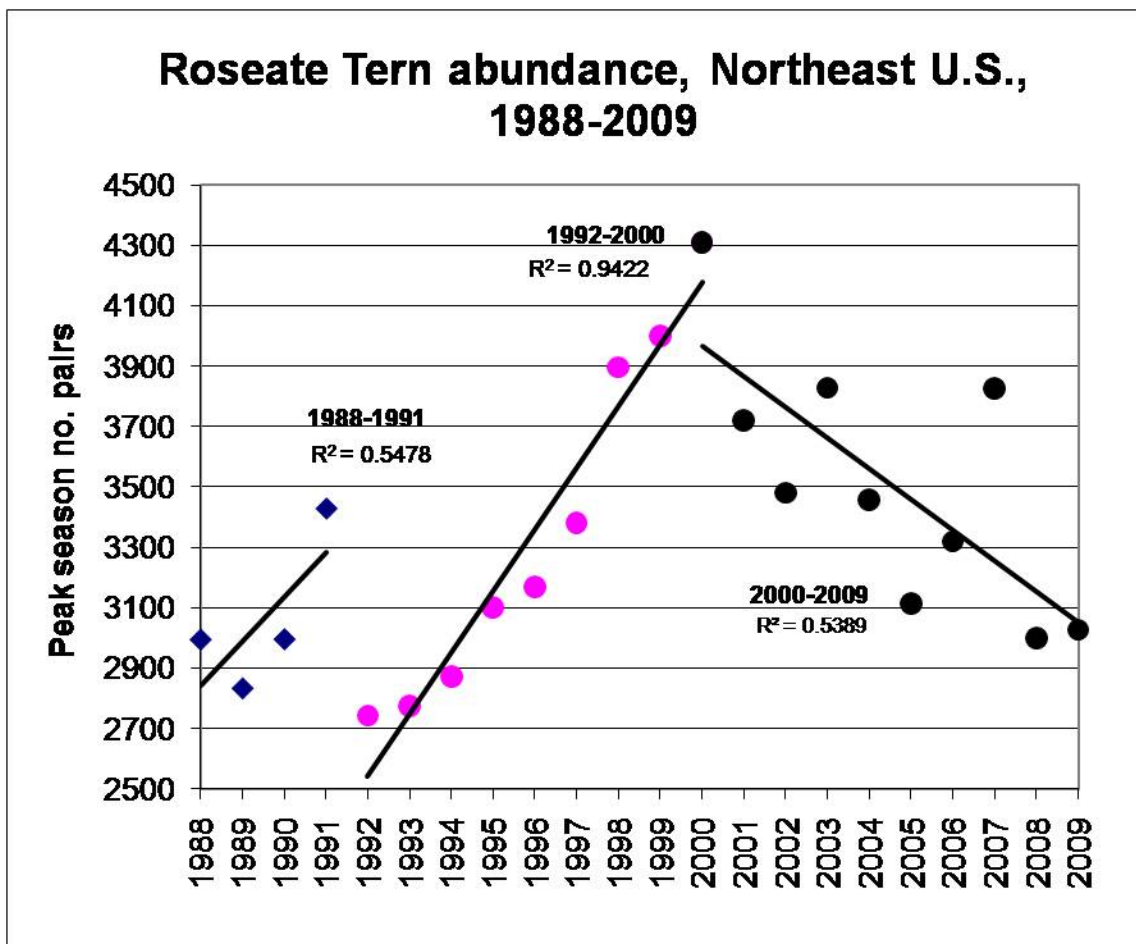
Site (listed from east to west)	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Smith's Point, Nantucket, MA	0	0	0	0	0	0	0	19	0	0	0	0
Muskeget I., Nantucket, MA	0	5	0	0	3	2	2	2	0	0	0	0
Dead Neck–Sampson's I., Cotuit, MA	0	0	0	0	0	0	0	1	0	0	0	0
Bird I., Marion, MA	1113	1148	1130	1062	505	904	554	680	1111	919	747	708
Ram I., Mattapoisett, MA	543	630	988	626	952	557	936	724	463	661	566	588
Penikese I., Gosnold, MA	0	0	0	0	0	251	9	76	48	102	66	43
Great Gull I., New York (NY)	1690	1747	1762	1562	1505	1613	1352	1195	1227	1546	1288	1413
Gardiner's Point I., New York	46	2	0	0	0	2	0	0	0	0	0	0
Gardiner's I./Cartwright I., NY	0	2	0	65	156	155	248	90	80	216	27	27
Young's I., Smithtown, NY	nd	nd	nd	nd	1	1	0	0	0	0	0	0
Falkner I., Connecticut (CT)	115	110	110	95	65	45	37	44	62	54	32	28
Warner I., Southampton, NY	31	28	9	0	0	1	1	0	0	0	0	0
Lane's I., Southampton, NY	32	14	18	5	0	4	4	2	0	2	0	0
Greater Greenbacks I., Southampton, NY	2	0	0	4	1	0	0	0	2	2	2	2
East Inlet I., Brookhaven, NY	8	0	0	0	0	0	0	0	0	0	0	0
Pattersquash I., Brookhaven, NY	nd	nd	nd	nd	nd	nd	nd	2	0	0	0	0
Sexton I., Islip, NY	nd	nd	nd	nd	nd	2	nd	2	0	nd	0	0
Goose Flat, Babylon, NY	16	1	2	4	1	0	11	11	25	2	1	0
Breezy Point, Queens City, NY	5	0	0	0	0	0	0	0	0	0	0	0
TOTAL	3601	3687	4019	3423	3189	3537	3154	2848	3018	3504	2729	2809
Number of sites	11	10	7	8	9	12	10	13	8	9	8	7

**Table 2b.** Total season estimates of roseate tern pairs in the warm water subregion of the northeastern United States 1998 to 2009. nd = no data. Note: Total season and peak period estimates were not made separately for the smaller sites in New York. Source: Carolyn Mostello (MDFW).

Site (listed from east to west)	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Smith's Point, Nantucket, MA	0	0	0	0	0	0	0	25	0	0	0	0
Muskeget I., Nantucket, MA	0	5	0	0	3	2	2	2	0	0	0	0
Dead Neck–Sampson's I., Cotuit, MA	0	0	0	0	0	0	0	1	0	0	0	0
Norton's Point, MV, MA	0	0	0	0	0	0	0	0	0	0	0	50
Bird I., Marion, MA	1195	1288	1280	1092	545	969	623	862	1223	975	880	782
Ram I., Mattapoisett, MA	605	696	1129	701	998	590	991	744	482	690	609	645
Penikese I., Gosnold, MA	0	0	0	0	0	266	9	84	50	111	73	50
Great Gull I., NY	1855	1814	2047	1715	1695	1723	1466	1273	1324	1636	1359	1524
Gardiner's Point I., NY	69	2	0	0	0	2	0	0	0	0	0	0
Gardiner's I./Cartwright I., NY	12	2	28	87	156	204	322	90	80	247	27	27
Young's I., Smithtown, NY	nd	nd	nd	nd	1	1	0	0	0	0	0	0
Falkner I., CT	120	110	115	100	70	46	43	53	62	54	32	41
Warner I., Southampton, NY	39	35	9	0	0	1	1	0	0	0	0	0
Lane's I., Southampton, NY	32	14	18	5	0	4	4	2	0	2	0	0
Greater Greenbacks I., Southampton, NY	2	0	0	4	1	0	0	0	2	2	2	3
East Inlet I., Brookhaven, NY	8	0	0	0	0	0	0	0	0	0	0	0
Pattersquash I., Brookhaven, NY	nd	nd	nd	nd	nd	nd	nd	2	0	0	0	0
Sexton I., Islip, NY	nd	nd	nd	nd	nd	2	nd	2	0	nd	0	0
Goose Flat, Babylon, NY	16	1	2	4	1	0	11	11	25	2	1	0
Breezy Point, Queens City, NY	5	0	0	0	0	0	0	0	0	0	0	0
TOTAL	3958	3967	4628	3708	3470	3770	3472	3151	3246	3717	2983	3122
Number of sites	12	10	8	8	9	12	10	13	8	9	8	8

Figure 2 plots the total number of peak period nests in the U.S. Northeast population from 1988 to 2009. Trends from 1988 to 1998 were described in the 1998 recovery plan update (USFWS 1998b) and by Nisbet and Spindel (1999). Numbers increased at average rates of 4 to 5 percent/year, except from 1991 to 1992 when they declined by about 20 percent. This decline has been attributed to Hurricane Bob, which passed through the main staging area of the population on 21 August 1991 and apparently eliminated many adults and most juveniles (see NE section 2.3.1.2.2). The increasing trend continued from 1998 to 2000 but abruptly reversed to a decline at about 4 percent/year starting in 2001. The highest total of 4,308 pairs in 2000 was 5 to 10 percent above both trend lines and may have resulted from overestimates at one of more colony sites: a total number of 4000 to 4100 pairs would better fit the trend line (see Figure 2 and Tables 1 and 2) and would be within the range of uncertainty in the individual counts. The increasing and decreasing trends were manifested at all the major sites (see Table 2) and evidently resulted from factors that affected the entire warm water subregion. However, the recent decrease was not clearly manifested in the cold water subregion until after 2007 (see Table 2).

**Figure 2.** Number of Roseate Tern Peak Season Nests in the Northeast Population, 1988 to 2009.





### 2.3.1.2.3 Adult survival

Survival of adults within the warm water subregion has been estimated since 1988, using mark-recapture modeling (Spendelow *et al.* 1995, 2009; Lebreton *et al.* 2003). Multi-site modeling has proved challenging, and the CRTMP data set has supported a number of methodological advances (Gould and Nichols 1998; Spendelow *et al.* 2002; Lebreton *et al.* 2003, 2009; Nichols *et al.* 2004).

The most recent set of estimates for adult survival was based on mark-recapture-re-sighting data from 11,029 individual birds at five breeding sites over an 18-year period, 1998 to 2006 (Spendelow *et al.* 2008). This incorporated and superseded survival estimates for shorter periods in the past (Spendelow *et al.* 1995, Lebreton *et al.* 2003) and those for single sites (Spendelow *et al.* 2002). The average annual survival rate over all sites was about 0.82, but the best-fitting models indicated substantial differences among sites and years (Table 3). Both models included in Table 3 indicated that the survival rate for 1991 to 1992 was lower than those for all other years. This has been attributed to Hurricane Bob, which passed through the main staging area for birds from these sites on 21 August 1991 and appears to have eliminated most juveniles and many adults (Nisbet and Spendelow 1999; Spendelow *et al.* 2002, 2008; Lebreton *et al.* 2003). The earlier analysis by Lebreton *et al.* (2003) suggested that the average survival rate over all three colonies was reduced to about 0.67 in 1991 to 1992, but the larger data set suggests that the effect was smaller than this and differed among colonies, with little or no change for birds marked or re-sighted at Great Gull Island in 1991 (see Table 3).

The other noteworthy feature of the survival estimates in Table 3 is that there was no measurable difference between survival rates in the period before 2000 (other than in 1991 to 1992) and those in the period from 2000 to 2004. In fact, averaged over all sites, the survival rates in these periods were  $0.835 \pm 0.006$  (standard error) and  $0.835 \pm 0.006$ , respectively (Spendelow *et al.* 2008). Thus, the marked change in the population trend from increasing in the period before 2000 (other than in 1991 to 1992) and decreasing in the period from 2000 to 2004 (Figure 2) was evidently not due to a change in adult survival.

The average adult survival rate of about 0.82 (all years) or 0.835 (non-hurricane years) for roseate terns is unusually low compared to survival rates of other terns and small gulls that have been estimated using the same mark-recapture methods. Most of these estimates have been in the range 0.85–0.92 (Stenhouse and Robertson 2005).

**Table 3.** Estimates of adult survival probabilities for roseate terns in the warm water subregion of the Northeast population (from Spendelov *et al.* 2008).

The table shows results from two models that fit the data about equally. Both models gave separate estimates of survival for three periods: 1988 to 1990, 1991, and 2000 to 2004. Model A grouped the period 1992 to 1999 with 1988 to 1990, while model B grouped it with 2000 to 2004. Each number in the table is an estimate of the average annual survival rate (survival from one breeding season to the next) over the range of years indicated: “1991” in the table indicates an estimate of survival over the year from 1991 to 1992, etc. Both models indicated small but distinct differences in survival rates among the five breeding sites. Standard errors are in parentheses.

	Site				
Period	Falkner Island	Great Gull Island	Penikese Island	Ram Island	Bird Island
<b>Model A</b>					
1998 to 1990, 1992 to 1999	0.861 (0.013)	0.784 (0.017)	—	0.814 (0.024)	0.848 (0.010)
1991	0.667 (0.049)	0.821 (0.058)	—	—	0.755 (0.037)
2000 to 2004	0.793 (0.028)	0.832 (0.028)	0.811 (0.058)	0.869 (0.017)	0.822(0.014)
<b>Model B</b>					
1998 to 1990	0.834 (0.040)	0.753 (0.070)	—	—	0.843 (0.031)
1991	0.667 (0.048)	0.777 (0.105)	—	—	0.739 (0.039)
1992 to 2004	0.851 (0.013)	0.802 (0.012)	0.812 (0.058)	0.856 (0.013)	0.837 (0.008)

#### 2.3.1.2.4 Productivity

Productivity is the average number of chicks raised to fledging per pair in 1 year. Chick survival and productivity are difficult to measure reliably in roseate terns, because chicks are concealed in dense cover and are often moved by the parents. Repeated searching for them may enhance these movements or have other adverse effects (Nisbet *et al.* 1990, Burger *et al.* 1996). Hence, attempts to follow chicks after the first few days of life usually yield underestimates of survival and productivity (Nisbet *et al.* 1990). As an example, 14 roseate tern chicks were color-banded at Country Island, Nova Scotia, in 2009. Biologists attempted to follow them to fledging but could confirm survival of only four, so they concluded that productivity was low. However, 13 of the 14 fledglings were identified at staging sites in Massachusetts during August to September (Jedrey *et al.* 2010).

Nisbet *et al.* (1999) devised a method for estimating colony productivity that avoided many of these difficulties. It was based on the finding that, in the absence of predation, survival of the first hatched, or A-chicks, is usually very high, so that variations in productivity result mainly from variations in the number and survival of the B-chicks (second hatched). Based on 10 years of data from Falkner Island, Nisbet *et al.* (1999) found that survival of the B-chicks could be predicted with 83 percent reliability from their body-mass on day 2 of life.

Consequently, colony productivity could be estimated from a single measurement on each brood in a representative sample (although implementation of the method required checking each nest daily around the time of hatching). The method was partly validated using data from Bird Island and has been adopted since 1999 for estimating productivity at Bird, Ram, and Penikese Islands. However, it has not yet been validated or used elsewhere, and its applicability is questionable in the cold water subregion, where survival of the A-chicks is thought to be lower even in the absence of predation (S. Hall, unpubl. data). Consequently, at most sites outside Buzzards Bay, other methods of estimating productivity are used, and these are subject to unquantified biases and errors (Nisbet *et al.* 1990, Burger *et al.* 1996).

Chick survival and productivity are especially difficult to measure at Great Gull Island, New York, which supports the largest colony in the Northeast region, because most nests are concealed under boulders and chicks remain hidden through most of the rearing period. Estimates of productivity at Great Gull Island were mainly obtained using a modified version of Method 1 of Nisbet *et al.* (1990), but this required monitoring of chicks for varying periods and is liable to yield underestimates of productivity, because chicks at Great Gull Island are very hard to find as they grow older. The B-chick method of Nisbet *et al.* (1999) was tested at Great Gull Island by visiting biologists in 2003 and 2004, as part of the Natural Resources Damage Assessment for the Bouchard No. 120 oil spill,

but the data obtained are confidential until the NRDA process is completed.

Estimates of productivity in the period 1998 to 2009 are included in Table 4 (coldwater subregion) and Table 5 (warm water subregion). Most of the estimates listed in the tables were of productivity during the peak period of nesting (see section NE 2.3.1.2.1), but this was not always specified and some of the estimates probably included varying numbers of post-peak nesters. Productivity of post-peak pairs is usually low, so the inclusion of these pairs would have led to downward bias. Methods of estimation varied among sites and were not always clearly stated, so the tabulated values are subject to the biases and potential errors discussed above. These caveats must be borne in mind when using the data in Tables 4 and 5 and the means derived from them in Tables 6 and 7.

Tables 4 and 5 include productivity estimates for 24 different sites, including at least one estimate for all but two of the sites that supported more than eight pairs in the peak period in any year from 1998 to 2009. The exceptions were Plymouth Beach and Smith's Point, Massachusetts, which were occupied only in single years and by small numbers. Estimates are available for colonies that included between 92 percent and 99.9 percent of the total peak period pairs in the region in each year. Table 6 summarizes the mean productivity in each of six subregions in each of the 12 years. Productivity was markedly lower at Falkner Island than at any other large colony throughout the period. This was partly due to predation that occurred from 2000 onwards, but productivity at Falkner Island has also been relatively low in earlier years (J. Spendelow, unpubl. data). Productivity was also slightly, but consistently, lower at colony sites in the Gulf of Maine than in the warm water subregion (other than Falkner Island). Productivity was consistently higher in larger colonies (Table 7). This was partly due to predation at the smaller colonies, but the same trend was evident at other sites and in years without predation. It probably resulted from a general tendency of roseate terns to move away from unfavorable sites and accumulate at the most favorable sites.

**Table 4.** Productivity estimates (fledglings per pair) for roseate terns in the cold water subregion of the northeastern United States, 1998 to 2009. Source: Carolyn Mostello (MDFW). Note: productivity estimates, supplied by island monitors, were obtained by various methods. Blank entries indicate either that no birds nested at the site or that productivity was not estimated; 0 means that one or more pairs nested but no chicks were raised.

Site (listed from north to south)	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
Petit Manan I., ME	1.05	1.30	1.00	1.10	0.72	0.55	0.62	0.44	0.74	0.16	0.75	0.33
Seal I., ME	0	1.00	1.00	0			1.00	1.00	0			0.78
Matinicus Rock, ME								0				
Metinic I., ME (North end)						1.70	1.00		0.50			
Eastern Egg Rock, ME	0.84	1.24	1.28	0.78	1.53	1.24	0.83	0.90	0.95	1.06	1.03	1.50
Pond I., ME						1.00	0.43	0	0			
Jenny I., ME		0.70			0	0.50		1.18	1.00	1.72	0	
Outer Green I., ME							1.15	0.64	1.47	1.36		
Stratton I., ME	1.12	1.68	1.40	1.17	0.69	0.05	0.69	1.50	0.95	1.41	1.25	1.31
Seavey I., NH				1.00	0.88	0.87	0.95	0.65	0.97	1.21	1.18	1.09
Nauset New I., Eastham, MA				0	0							
Minimoy I., Chatham, MA						1.70	1.13	0.73	1.00	1.03	1.00	
S.Monomoy I., Chatham, MA	0.97	0.78	1.00	0.83	1.00	1.33	1.00	0	0.33	1.00		

**Table 5.** Productivity estimates for roseate terns in the warm water subregion of the northeastern United States, 1998 to 2009. Source: Carolyn Mostello (MDFW).

Site (listed from east to west)	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
<i>Bird I., Marion, MA</i>	1.40	1.00	1.10	1.20	1.02	1.25	1.25	0.95	1.29	1.26	1.12	1.44
Ram I., Mattapoisett, MA	1.45	1.04	1.11	1.05	0.96	1.14	0.92	0.93	1.00	1.16	1.25	1.04
Penikese I., Gosnold, MA						0.87	0.97	0.79	0.44	0.54	1.42	0.73
Great Gull I., NY	1.50	1.40	1.27	1.13	0.90	1.00	1.17	1.60	1.30	0.90	0.98	1.10
Gardiner's Point Island, NY		1.0										
Gardiner's Island/Cartwright Point, NY				0.75								
Falkner I., CT	0.74	0.70	0.75	0.72	0.18	0.26	0.53	0.67	0.34	0.54	1.06	1.15
Young's Island, NY					0	0						
Warner I., Southampton, NY	1.05	1.07	0			0						
Lane's I., Southampton, NY		0	0	0								
Goose Flat, Babylon, NY								1.82				

Overall, mean productivity was high in 1998 to 1999, declined to low values in 2001 to 2004, and showed a partial recovery thereafter. This pattern was manifested in all subregions (Table 6), so it presumably resulted from factors operating on a region-wide scale. At least at Bird Island, productivity had been consistently high in 1970 to 1972 (when monitoring stopped), in 1980, and between 1987 (when monitoring resumed) and 1997: annual means ranged from 1.09 to 1.60 fledglings/pair with an overall mean of about 1.29 (Nisbet and Drury 1972, Nisbet *et al.* 1990, Burger *et al.* 1996, Nisbet and Spendelow 1999, I. Nisbet, unpubl. data). The relatively low values at the Buzzards Bay sites in 2001 to 2005 were exceptional, and even the higher values in 2006 to 2009 were still below the range for 1970 to 2000 (it should be noted that methods of estimating productivity at the Buzzards Bay sites changed in 1999, but it is unlikely that this change could explain more than a small part of the apparent shift to lower values). It appears, therefore, that there has been a regionwide decrease in mean productivity, starting in 2000. This may have contributed to the observed decline in the breeding population between 2000 and 2009, but it cannot have been the only cause, because birds raised in 2001 (the first year of reduced productivity) would not have recruited to the breeding population until 2004 or 2005 and would not have contributed to the peak period numbers until 2005 or 2006. The decline in the breeding population was already marked in 2001 (Figure 2), and recruits in those years would have come from the cohorts raised in 1996 to 1999 when productivity was very high—even higher than the 1970 to 1995 average (see Table 5; I. Nisbet, unpubl. data).

**Table 6.** Mean productivity of roseate terns in six subregions of the northeastern United States, 1998 to 2009 (from data in Tables 4 and 5). Means were calculated by weighting productivity estimates for each site/year in proportion to the number of peak nests. “Nests” are the total numbers of peak-period nests at sites where productivity was measured: these comprised 92–99.9 percent of the regional population in each year (see Tables 1a and 2a).

Subregion	Sites	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	Mean
Gulf of Maine	13	0.95	1.24	1.29	0.87	1.18	0.87	0.86	0.79	0.95	1.19	1.09	1.01	1.033
nests		272	313	291	299	296	294	303	289	302	322	269	218	3468
Buzzards Bay, MA	3	1.42	1.27	1.10	1.06	0.98	1.16	1.04	0.94	1.18	1.24	1.19	1.24	1.151
nests		1656	1778	2118	1688	1457	1712	1498	1480	1822	1682	1379	1339	19,410
Eastern Long Island, NY	4	1.50	1.40	1.27	1.11	1.00	1.00	1.17	1.60	1.30	0.90	0.98	1.10	1.185
nests		1690	1751	1762	1627	1505	1613	1352	1195	1227	1546	1288	1298	17,970
Falkner Island, CT	1	0.74	0.70	0.76	0.73	0.18	0.26	0.53	0.67	0.34	0.54	1.06	1.16	0.615
nests		115	110	110	95	65	45	37	44	62	54	32	28	797
S. shore of Long Island, NY	3	1.05	1.17		0.00		0.00		1.82					1.108
nests		31	28		5		1		11					66
Northeast Total	24	1.40	1.30	1.17	1.04	0.94	1.00	1.07	1.19	1.19	1.08	1.09	1.16	1.139
nests		3764	3994	4308	3714	3323	3666	3191	3019	3213	3604	2968	2998	41,711

**Table 7.** Mean productivity of roseate terns of the Northeast population, grouped by colony size.

Number of peak-period nests in colony	Mean productivity $\pm$ standard error	Number of cases
1 to 5	0.69 $\pm$ 0.06	40
6 to 10	0.85 $\pm$ 0.07	38
11 to 48	0.98 $\pm$ 0.07	36
52 to 251	1.13 $\pm$ 0.11	14
463 to 1762	1.18 $\pm$ 0.09	22

#### 2.3.1.2.5 Juvenile survival, age at first breeding and recruitment

Rates of juvenile survival (from fledging to first breeding) are very difficult to estimate because ages at first breeding and rates of dispersal among sites have to be considered simultaneously. Estimates of these parameters for roseate terns are available only for the period 1988 to 1997: Spendelov *et al.* (2002) for Falkner Island only and Lebreton *et al.* (2003) for the three major sites. Table 8 gives Lebreton *et al.*'s (2003) estimates of survival to age 2 for three sites over 10 years. These estimates of juvenile survival varied very widely among sites and years, from about 0.04 to about 0.75. Estimates of survival from fledging to age 2 were low at all three sites for the period 1991 to 1993 (averaging about 0.10), which was again attributed to Hurricane Bob. Over all other years and sites, the average survival to age 2 was about 0.43. However, these estimates must be qualified, because the birds included in the study (both "first-hatched A" and "second-hatched B") had been banded at various ages between hatching and fledging.

Although chick survival was high at all three sites in these years, some of the chicks included in the analysis would presumably have died prior to fledging, so the estimates in Table 8 would be somewhat lower than the true survival from fledging to age 2.

**Table 8.** Estimates of survival probability of juvenile roseate terns raised at the three major sites in the warm water subregion of the Northeast population from fledging to age 2 years over 10 years, 1988 to 1997 (from Lebreton *et al.* 2003).  $\pm$  denotes standard error.

Time period	Site		
	Falkner Island	Bird Island	Great Gull Island
1998 to 1990	0.310 $\pm$ 0.111	0.325 $\pm$ 0.087	0.352 $\pm$ 0.147
1989 to 1991	0.320 $\pm$ 0.148	0.340 $\pm$ 0.090	0.505 $\pm$ 0.165
1990 to 1992	0.336 $\pm$ 0.127	0.378 $\pm$ 0.088	0.490 $\pm$ 0.144
1991 to 1993 (hurricane year)	0.063 $\pm$ 0.057	0.185 $\pm$ 0.095	0.065 $\pm$ 0.044
1992 to 1994	0.593 $\pm$ 0.227	0.748 $\pm$ 0.204	0.496 $\pm$ 0.167
1993 to 1995	0.547 $\pm$ 0.172	0.275 $\pm$ 0.109	0.498 $\pm$ 0.159
1994 to 1996	0.354 $\pm$ 0.154	0.415 $\pm$ 0.150	0.354 $\pm$ 0.154
1995 to 1997	0.407 $\pm$ 0.249	0.046 $\pm$ 0.066	0.434 $\pm$ 0.259



These analyses have not been updated for years later than 1997, so it remains unknown whether the marked change in the population trend from increasing through 2000 (other than in 1991 to 1992) to decreasing in the period from 2000 to 2004 (see Figure 2) could have been due to a change in juvenile survival and a consequent failure of recruitment. However, trapping of breeding adults continues to yield substantial numbers of birds breeding at ages 3 to 9 (J. Spendelow, unpubl. data).

Lebreton *et al.* (2003) also estimated ages at first breeding for the same birds (Table 9). Averaged over all sites and years, the probability of breeding for birds that survived to ages 2, 3, 4 and 5 was about 0.02, 0.51, 0.85 and 0.97, respectively (the analysis required the assumption that all would have bred by age 6). Spendelow *et al.* (2002) had earlier derived estimates for age of first breeding at Falkner Island only: 0 at age 2, 0.75 at age 3 and 0.92 at age 4.

Recruitment of new birds into the breeding population has not been estimated, except through modeling (see NE section 2.3.1.2.9).

**Table 9.** Ages at first breeding of roseate terns at the three major sites in the warm water subregion of the Northeast population, 1988 to 1997 (from Lebreton *et al.* 2003). Each entry in the table is the estimate ( $\pm$  standard error) of the probability that a bird that survived to ages 2, 3, 4, 5 would have entered the breeding population. Estimates are averaged over all years.

Breeding site	Age			
	2	3	4	5
Falkner Island	0.010 $\pm$ 0.012	0.515 $\pm$ 0.085	0.620 $\pm$ 0.145	0.912 $\pm$ 0.244
Bird Island	0.046 $\pm$ 0.036	0.674 $\pm$ 0.180	1.000	1.000
Great Gull Island	0.019 $\pm$ 0.023	0.448 $\pm$ 0.159	0.668 $\pm$ 0.411	1.000

#### 2.3.1.2.6 Dispersal

Dispersal among colony sites was also estimated simultaneously in the modeling studies reported by Spendelow *et al.* (1995) and Lebreton *et al.* (2003). Tables 10 and 11 give estimates of natal dispersal (movements of birds raised as chicks at one colony site to breed for the first time at another site) and adult dispersal (movements of birds that bred at one colony site in 1 year to breed at another site in the next year), respectively. The diagonals in both tables give fidelity rates (probabilities that a bird will remain at the same site). The natal fidelity rate at the small colony at Falkner Island (about 0.58) was much smaller than those at the large colonies at Bird and Great Gull Islands (about 0.90). The natal dispersal rate from the small colony at Falkner Island to the large and nearby colony

at Great Gull Island was much larger (0.37) than those between any other pair of colonies (all 0.08 or less). Adult fidelity rates were much higher than natal fidelity rates, but the spatial patterns were similar (Tables 10 and 11). The adult fidelity rate at the small colony at Falkner Island (about 0.885) was much smaller than those at the large colonies at Bird and Great Gull Islands (about 0.98). Again, the adult dispersal rate from the small colony at Falkner Island to the large and nearby colony at Great Gull Island was much larger (0.092) than those between any other pair of colonies (all 0.023 or less).

**Table 10.** Natal dispersal rates of roseate terns among the three major sites in the warm water subregion of the Northeast population, 1988 to 1997 (from Lebreton *et al.* 2003). Each entry in the table is the estimate ( $\pm$  standard error) of the probability that a bird raised as a chick at one site would breed for the first time at another site. Values on the diagonal (\*) are fidelity rates (1 – sum of emigration rates); standard errors were not estimated for fidelity rates. Estimates are averaged over all years.

	Site		
First breeding site	Falkner Island	Bird Island	Great Gull Island
Falkner Island	0.582*	0.004 $\pm$ 0.004	0.020 $\pm$ 0.009
Bird Island	0.047 $\pm$ 0.060	0.913*	0.082 $\pm$ 0.038
Great Gull Island	0.378 $\pm$ 0.130	0.082 $\pm$ 0.042	0.898*

**Table 11.** Adult dispersal rates of roseate terns among the three major sites in the warm water group of the Northeast population, 1988 to 1997 (from Lebreton *et al.* 2003). Each entry in the table is the estimate ( $\pm$  standard error) of the probability that a bird that bred at one site would breed at another site in the next year. Values on the diagonal (\*) are fidelity rates (1 – sum of emigration rates); standard errors were not estimated for fidelity rates. Estimates are averaged over all years.

	Breeding site in first year		
Breeding site in next year	Falkner Island	Bird Island	Great Gull Island
Falkner Island	0.885*	0.001 $\pm$ 0.001	0.007 $\pm$ 0.002
Bird Island	0.023 $\pm$ 0.014	0.985*	0.013 $\pm$ 0.007
Great Gull Island	0.092 $\pm$ 0.024	0.014 $\pm$ 0.009	0.981 *

Note that the fidelity and dispersal rates reported in Tables 10 and 11 are probabilities of movement by an individual bird. Although in both cases the *probabilities* that an individual bird would move from Falkner Island to Great Gull Island were much higher than the *probabilities* that an individual bird would move in the opposite direction, this is offset by the fact that there were about 10 times more birds that could have moved at Great Gull Island (see Table 2) and that productivity was higher at Great Gull Island (see Table 5). Multiplying the movement probabilities in Tables 10 and 11 by the mean numbers of pairs and estimates of mean productivity at each site, the net movements from Falkner Island to Great Gull Island can be calculated to have been about 22 juveniles and 12 adults each year. In spite of this net loss, numbers of breeding pairs at Falkner Island remained constant at about 130 pairs during this period. The net outflow of adults from Falkner Island probably increased after 1997, as numbers declined by 80 percent during the ensuing 12 years (see Table 2), and the net outflow of juveniles was probably reduced because productivity declined at the same time (see Table 5).

Estimates of dispersal rates have not been updated since 1997, but detailed studies are ongoing and have been extended to Ram and Penikese Islands. Preliminary results indicate that both natal and adult dispersal rates have been much higher among the three colony sites in Buzzards Bay (Bird, Ram and Penikese Islands) than between those sites and Great Gull Island. In particular, mark-recapture data suggest that the colony at Penikese Island has been unstable, with many birds breeding there for only 1 year before moving back to Bird or Ram Islands (J. Spendelow, unpubl. data). Also, large numbers of breeding birds moved to and from Bird and Ram Islands, in response to alternating episodes of owl predation and the hazing conducted at Ram Island in 2003 in response to the Bouchard No. 120 oil spill (see NE section 2.3.1.7.8.1).

All the above data on dispersal rates refer to the warm water subregion of the regional population breeding south and west of Cape Cod. There has been no trapping of adult roseate terns at cold water sites north and east of Cape Cod and no systematic program of re-sighting, so there are no formal estimates of dispersal rates within the cold water subregion or between the cold water and warm water subregions. However, chicks have been banded with FR bands at many sites in the cold water subregion, and FR bands have been read in recent years at many sites in the Gulf of Maine and at The Brothers in Nova Scotia. These re-sightings have documented many movements of birds among the cold water sites, and a number of sightings of birds raised as chicks at the warm water sites breeding at cold water sites. However, to date there have been only two or three records of natal dispersal in the reverse direction (S. Hall, L. Welch, and J. Spendelow, unpubl. data). Although no quantitative estimates are available, it is thought that movements between the two subregions are

much smaller than dispersal movements within either subregion, that most of the dispersal is from the warm water sites towards cold water sites, and that most of the movement between the subregions is of birds emigrating to breed for the first time (natal dispersal; Nisbet and Spendelow 1999, Spendelow *et al.* 2010). Very recently, however, six adult roseate terns that bred at sites in the warm water subregion from 2004 to 2006 were found at breeding sites in the Gulf of Maine in 2005 to 2006 (Spendelow *et al.* 2010).

#### **2.3.1.2.7 Sex-ratio**

It had long been conjectured that the sex-ratio of roseate terns in the northeast breeding population is skewed towards females, because of the prevalence of “supernormal clutches” (SNCs): clutches of 3 or 4 eggs versus the normal clutch-size of 1 to 2 eggs) and the low hatching success of eggs in SNCs (Nisbet 1981, USFWS 1998b, Gochfeld *et al.* 1998). Nisbet and Hatch (1999) investigated this phenomenon at Bird Island in 1992 to 1994, using genetic methods to sex birds attending both SNCs and normal clutches. They found that most (>80 percent) SNCs were attended by two or more females, usually without a male partner, but they also found that about 7 percent of normal clutches were similarly attended by two females. Overall, they estimated that about 12 percent of all clutches at Bird Island were attended by two or more females; about 1.5 percent of clutches were attended by trios, but most of these consisted of three females rather than two females and one male. They estimated the overall sex-ratio of the breeders as 127 females:100 males (*i.e.*, 56 percent female), and gave evidence that other females were present but did not breed. They compiled data on the frequencies of SNCs in prior years and concluded that this frequency (and, by inference, the amount by which the sex-ratio was skewed towards females) had increased 2- to 4-fold since 1970. About 46 percent of the eggs attended by female-female pairs were fertile and hatched (vs. 98 percent hatchability in eggs laid by females in female-male pairs). Overall, Nisbet and Hatch (1999) calculated that the average productivity of females without male mates was 0.34 fledglings/female, vs. about 1.35 for females mated to males. Consequently, the overall average productivity in the colony was only about 80 percent of that which would have been achieved if all females had male mates. This problem has worsened in recent decades.

Genetic methods have also been used to sex birds at Falkner Island (Szczyz *et al.* 2005b; D. Shealer and J. Spendelow, unpubl. data). The results have not been reported but similarly show that SNCs and some normal clutches were attended by female-female pairs, which probably amounted to about 5 percent of all clutches there (J. Spendelow, unpubl. data). Genetic sexing has not been used at other sites in the region, but SNCs have been recorded at all sites for which extensive data are

available, making it likely that the sex-ratio is skewed towards females throughout the regional population (Nisbet and Spendelov 1999). The frequencies of SNCs are sometimes similar to those recorded by Nisbet and Hatch (1999) at Bird Island but tend to be somewhat smaller, especially at sites in the cold water subregion (USFWS unpubl. data). However, this may be because average clutch-sizes are smaller in the cold water region, *i.e.*, more females lay only one egg. Hence it is likely that a higher proportion of female-female pairs would lay 2-egg clutches rather than SNCs at these sites; thus, a lower frequency of SNCs does not necessarily imply fewer female-female pairs or a less skewed sex-ratio.

Investigations into the cause of the skewed sex ratio have so far been inconclusive. Using genetic sexing, Szczys *et al.* (2001) found a female-biased sex-ratio at hatching (55 percent females among 342 hatchlings) at Bird Island in 1997; the female bias was significant among early A-eggs (first in clutch), but not among late A-eggs or B-eggs (second in clutch). However, Szczys *et al.* (2005a) found no sex-ratio skew at hatching (50 percent female and 50 percent male among 586 hatchlings) at Falkner Island over a 5-year period (1998 to 2002). In both cases, there was no difference between male and female chicks in early growth parameters or in survival (Nisbet and Szczys 2001, Szczys *et al.* 2005a).

Nichols *et al.* (2004), using a novel method of analysis developed for the purpose, derived estimates of sex-specific survival of adult roseate terns at Falkner Island over 8 years (1993 to 2000). Apparent survival rates of females were higher than those of males in each of the 8 years (range 0.70 to 0.97 for females, 0.66 to 0.90 for males; mean difference 0.05). Although the difference was large, the study was based on re-sighting data from Falkner Island only. Given that emigration from Falkner Island is quite high (see Table 11), the difference in apparent survival could have been confounded by differential emigration of males. Multi-site estimates of sex-specific survival of roseate terns, taking account of dispersal, have not yet been attempted.

Teets (1998) found that male roseate terns take a larger role than their mates in feeding chicks at the time of fledging and usually leave the breeding colony with the A-chick soon after it fledges, leaving the female behind to tend the B-chick (if any). Since many roseate tern pairs do not raise two chicks to fledging, this means that males expend more energy than females in raising chicks. This might lead to lower survival of males.

#### **2.3.1.2.8 Non-breeding**

The proportion of roseate terns that “skip” breeding for 1 year, or otherwise fail to breed, is very difficult to estimate, because non-breeding cannot be distinguished from non-detection in most mark-recapture

studies. Based on intensive observations at Falkner Island, where most birds were individually marked in the period 1992 to 2003 and would have been detected whether breeding or not, J. Spendelow (unpubl. data) identified a small proportion (roughly 2 to 5 percent) of birds that frequented the nesting area but did not breed. Based on behavior (such as begging and soliciting copulation) in the years when they did not breed, and/or sexing when they did breed in other years, these were probably all females. Spendelow did not detect any birds known to be males that frequented the nesting area but did not breed. This suggests that most or all males breed every year, while many females either do not breed or pair with other females and breed with low success (see previous section).

Nisbet and Ratcliffe (2008) showed that roseate terns in this and other temperate populations have much smaller year-to-year fluctuations in breeding numbers (relative to trend lines) than those in tropical populations, and inferred that the frequency of non-breeding was low at the temperate sites but high at the tropical sites.

#### **2.3.1.2.9 Population modeling**

Arnold (2007) prepared a population viability analysis (PVA) for the Northeast population of the roseate tern, which was incorporated into the Cape Wind Energy Project/Environmental Impact Report/Development of Regional Impact (EOEA 2007). A PVA is a stochastic model that is used to calculate population trajectories and consequently extinction probabilities for animal populations for varying periods into the future. PVA models are formulated to incorporate a set of estimates of demographic parameters and their likely variability; they are run many times (typically, 10,000 iterations), each with a different set of parameters selected at random from within the pre-specified ranges. The ensemble of model outputs yields estimates of the probabilities that the population will decline to extinction within a specified period (typically 50 years). The calculations are then re-run with a different set of initial demographic parameters, to test the likely effect of management actions that might result in changes in these parameters.

The value of PVAs in prediction has been debated extensively in the conservation biology literature (*e.g.*, Boyce 1992, Fieberg and Elner 2000, Reed *et al.* 2002). PVAs are commonly used in conservation and management decisions and have a wide range of uses. They are useful as tools to elucidate the most important life history stages to focus conservation efforts and also have value as predictive tools. They allow all available data on the life history of an organism to be considered and to integrate them into a single analysis. Although PVAs generate predictions of extinction probabilities at various times in the future, these should not be taken as literal estimates of the probability that the population will

actually go extinct, because environmental factors and demographic parameters will undoubtedly change. They are most often used to predict the likely effects of conjectured changes in the demographic parameters; these can be useful in informing management decisions (Meffe and Carroll 1994).

Table 12 lists the parameters adopted by Arnold (2007) for her roseate tern PVA. Most parameters were derived from those reported by Lebreton *et al.* (2003), which were based on mark-recapture data from 1988 to 1997 (see NE sections 2.3.1.2.3 and 2.3.1.2.5 and Tables 3, 8 and 9); productivity estimates were derived from some of the same sources as those cited in section 2.3.1.2.3 and in Tables 5 and 6. The “best-case” scenario was intended to mimic the observed growth trajectory (*i.e.*, increasing numbers) and estimated survival rates in 1988 to 1997 (excluding the hurricane year 1991). The “worst-case” scenario used the lowest survival rates in the period 1988 to 1997 (again excluding the hurricane year). The “recent trend” scenario adjusted the survival rates to be consistent with the decreasing trend of the population since 2000. The relative probabilities of these three scenarios (1, 9, and 90 percent, respectively) represented the author’s judgment that “the short period of growth observed from 1988 to 2000 was an anomaly and the current negative growth rate is more realistic” (Arnold 2007). The model was designed to apply to males, because males are the limiting sex (see NE section 2.3.1.2.7) and are thought to breed every year (see NE section 2.3.1.2.8).

The model was then run 10,000 times with parameter values selected at random within the ranges defined by the variances in Table 12. The probability of “quasi-extinction” (defined as reduction in number of males to 500) was 9 percent at 15 years, 42 percent at 25 years, and 95 percent at 50 years. A hypothetical “take” of 20 males per year increased these probabilities to 16, 57, and 97 percent, respectively. A hypothetical take of 100 males per year increased these probabilities to 64, 94, and 99 percent, respectively.

**Table 12.** Estimates of demographic parameters used in the population viability analysis (variances in parentheses) (from Arnold 2007).

	Scenario		
	Recent trend	Worst-case	Best-case
Probability of occurrence	90 percent	9 percent	1 percent
Demographic parameter			
Productivity (male-female pairs)	1.0560 (0.0890)	same as recent trend	same as recent trend
Probability of breeding age 2	0.0323 (0.0004)	same as recent trend	same as recent trend
Probability of breeding age 3	0.5669 (0.0133)	same as recent trend	same as recent trend
Probability of breeding age 4	0.8351 (0.0785)	same as recent trend	same as recent trend
Annual adult survival rate	0.8719 (0.0061)	0.7648 (0.0061)	0.8700 (0.0061)
Annual juvenile survival rate	0.5629 (0.0675)	0.6151 (0.0675)	0.7233 (0.0675)
Sex-ratio of breeders (M/F)	0.45	same as recent trend	same as recent trend

Numerous objections could be made to the selection of parameter values, variances, and scenario probabilities used by Arnold (2007) (*e.g.*, the worst-case scenario was not as bad as the event actually observed in 1991, and the population trend modeled in the best-case scenario was observed in 10 of 20 of the other years in which the population was monitored rather than 1 percent). However, if the PVA is used for the more limited purpose of assessing the likely consequences of incidental take (see above), it is probably reasonable in its conclusions that the effect of take of 20 males/year (<1 percent reduction in adult survival rate) would be small, and that the effect of take of 100 males/year (>3 percent reduction in adult survival rate) would be much larger.

Nisbet and Ratcliffe (2008) constructed a much simpler life-table model of the Northeast population, for the specific purpose of testing whether the available estimates of demographic parameters were consistent with the observed population trends. The model was designed to estimate the rate of population increase  $r$ , given the fraction of adults that breed  $f$ , productivity  $p$ , juvenile survival (from fledging to age 2)  $j$ , and annual adult survival  $s$ . It assumed that all adults breed for the first time at age 4 years. Otherwise, parameters were taken from the sources listed in footnotes to Table 13. Note that the parameter  $j$  is survival to age 2 and this is equivalent to the square of the annual juvenile survival rates listed in Table 12. The calculated values of  $r$  agreed with the observed values



during the two periods when the population was decreasing: 1991 to 1992 and 2000 to 2006. However, the model also predicted population declines at about 3 percent per year during the periods 1988 to 1990 and 1992 to 2000, when the observed trend was an increase at about 5 percent per year. Given the other demographic parameters listed in Table 13, annual adult survival would have to be increased to about 0.91, or juvenile survival would have to be increased to about 0.75, to be consistent with the observed rate of increase in numbers during these two periods. Nisbet and Ratcliffe (2008) pointed out similar but larger discrepancies for tropical populations of roseate terns, and concluded: “Despite the intensive study of roseate terns in several parts of their world range during the last 25 years ... important features of the demography of [the] populations remain poorly understood.”

#### **2.3.1.2.10 Metapopulation structure and dynamics**

Based on the demographic information summarized in previous sections, the Northeast population of the roseate tern appears to conform to a classic “source-sink” metapopulation (Pulliam 1988, Hanski 1999). It includes three large central colonies at Bird, Ram, and Great Gull Islands and about 75 peripheral sites, of which only 10 to 25 have been occupied at any one time (Figure 1). At present, a few of the peripheral sites that are occupied are close to the central sites, but most are scattered in a band extending from 160 to 1,130 km (100 to 700 mi) to the northeast through the cold water subregion (see Figure 1). In recent years, the peripheral sites have supported varying numbers from 1 to 250 breeding pairs each, versus 500 to 1700 pairs at each of the three central sites. The central sites and a few of the larger peripheral sites have been occupied for many years, but the smaller colonies are less stable: local colonization, extirpation, and recolonization have been frequent, especially at sites with very small numbers (see Tables 1 and 2). Productivity is higher at the central sites than at the peripheral sites; thus, the former are sources and some (but not all) of the latter are sinks. Dispersal of adults and juveniles among the three central sites is fairly frequent, especially between the sites that are closest to each other (Bird and Ram islands) (see Tables 10–11; J. Spendelov, unpubl. data). Juvenile dispersal is sufficiently high to lead to complete population mixing within one generation (see Table 10). Two-way dispersal among the peripheral sites and between the central and peripheral sites has been documented but has been quantified in only one case where the net movement was from one of the larger peripheral sites towards the nearest central site (see NE section 2.3.1.2.6). Under metapopulation theory, these two-way movements are thought to enhance the overall stability and genetic diversity of the population and to reduce the probability of extinction (Pulliam 1988, Hanski 1999).

**Table 13.** Partial life-tables for the Northeast population of roseate terns during three periods of contrasting population trends (from Nisbet and Ratcliffe 2008).

Parameter	Years		
	1988 to 1990, 1992 to 2000	1991 to 1992	2000 to 2006
Fraction breeding $f$	0.90	0.90	0.90
Productivity $p$	1.10 <sup>a</sup>	1.10 <sup>a</sup>	1.10 <sup>a</sup>
Juvenile survival $j$	0.38 <sup>b</sup>	0.315 <sup>c</sup>	0.38 <sup>d</sup>
Adult survival $s$	0.835 <sup>e</sup>	0.716 <sup>e</sup>	0.835 <sup>e</sup>
Rate of increase $r$			
calculated <sup>f</sup>	−0.03	−0.17	−0.03
Observed	+0.05	−0.20	−0.04

<sup>a</sup> Average for region, including sites listed in Tables 4 and 5 (Roseate Tern Recovery Team, unpubl. data).

<sup>b</sup> Average over all cohorts 1988–1995, excluding values listed in footnote c; estimates from Lebreton *et al.* (2003).

<sup>c</sup> Mean of values for 1988 and 1989 cohorts, assumed recruits in 1992 were drawn equally from these cohorts.

<sup>d</sup> No data for relevant cohorts; assumed to be the same as in the first period.

<sup>e</sup> Derived from a model which constrained survival rates to be equal across all sites (Spendelov *et al.* 2008).

<sup>f</sup> Linear regression of population estimates over period(s) for which  $r$  is calculated; data from Spendelov *et al.* (2008).

### **2.3.1.3 Genetics, genetic variation, or trends in genetic variation:**

Lashko (2004) used two genetic markers – ND2 and ND6 mitochondrial DNA sequences and microsatellite genotyping – to investigate current and historical relationships among roseate terns throughout their global range. She found that birds breeding in the North Atlantic Ocean were genetically distinct from those in the Indo-Pacific Oceans, which had been separated historically by the African continent. Each group was genetically homogeneous, apparently having expanded from glacial refugia in tropical latitudes since the last ice age (20,000 years ago). The temperate breeding populations in northeastern North America and western Europe were apparently established since that time. Birds sampled in Ireland and the United States were both genetically distinct from those in the Azores, but were not significantly different from each other (Lashko 2004). This similarity may result either from the fact that the Irish and U.S. populations have low genetic diversity resulting from historical bottlenecks or from gene flow due to exchange of individuals between the populations. Recent observations have provided limited evidence for interchange of individuals among the three groups: four roseate terns banded in Ireland have been sighted at colonies in the northeast United States (Nisbet and Cabot 1995, Hays *et al.* 2002a, I. Nisbet unpubl. data), two from the northeast United States and one from the Azores have been reported in Ireland (Newton and Crowe 2000), and one probably from the United States has been reported in the Azores (Hays *et al.* 2002b). However, all these sightings were made by reading bands by telescope and they have not been confirmed by trapping, despite intensive trapping of the northeast breeding population (Spendelov *et al.* 2008). Although several of these birds were reported within nesting areas, none was actually seen at a nest. Hence, the extent of interchange of breeding individuals among the populations, if any, remains conjectural.

Szczys *et al.* (2005a) used some of the same microsatellite markers to compare roseate terns from Bird and Falkner Islands in the United States to those in Western Australia. They also found that the Atlantic and Australian birds were genetically distinct from each other, but found no significant differentiation between birds from Bird and Falkner Islands. They reported that the U.S. birds had much lower genetic diversity than the Australian birds, perhaps reflecting recent historical population fluctuations and bottlenecks; however, they found little evidence of inbreeding.

No genetic studies of the Caribbean population have been documented.

### **2.3.1.4 Taxonomic classification or changes in nomenclature:**

There have been no changes in the taxonomic classification or nomenclature of roseate terns. Both the Northeast and Caribbean populations remain classified in the subspecies *Sterna d. dougallii*. Lashko (2004) confirmed that this subspecies is distinct from *S. d. gracilis* of the Indo-Pacific region, but she did not evaluate the Caribbean birds.

At a higher taxonomic level, Bridge *et al.* (2005) used mitochondrial DNA sequences from 33 species to construct a phylogeny of the terns (Sternini). They recommended a revision to recognize 12 genera (versus 3 to 10 in earlier classifications); this recommendation has been accepted by the American Ornithologists' Union Checklist Committee (AOU 2008). Under this classification, the genus *Sterna* is restricted to 13 species of "typical black-capped terns." Within the genus, the roseate tern appears most closely related to the white-fronted tern (*S. striata*) of Australia and the black-naped Tern (*S. sumatrana*) of the Indo-Pacific region, then to the Arctic (*S. paradisaea*), common (*S. hirundo*), South American (*S. hirundinacea*), and Antarctic (*S. vittata*) terns. A previously conjectured relationship with the crested terns (*Thalasseus* spp.), based on similarities in behavior and plumage structure, was not confirmed.

#### **2.3.1.5 Spatial distribution, trends in spatial distribution, or historic range:**

The breeding range of the Northeast population of roseate terns had been shrinking before 1998, and this trend continued in 1998 to 2009. Numbers along the south shore of Long Island, New York, although incompletely documented, continued to decline and that area was largely abandoned by 2009. Numbers at Falkner Island (the only breeding site west of Great Gull Island in Long Island Sound) declined to their historically lowest level by 2009; and numbers at Petit Manan Island, Maine (the easternmost breeding site in the United States) declined from a high of 31 pairs in 2003 to only 4 pairs in 2009 (Tables 1 to 2). Numbers have also declined at the two major sites in Atlantic Canada (A.Boyne and T. D'Eon, unpubl. data).

The percentage of birds in the three largest colonies (Bird, Ram, and Great Gull Islands) has remained in the range 80 to 90 percent throughout the period 1988 to 2009, with no clear tendency to increase or decrease during that time (see Tables 1 and 2).

For distribution in winter and on migration, see NE section 2.3.1.7.2

#### **2.3.1.6 Habitat or ecosystem conditions:**

The coastal islands used for nesting by roseate terns in the Northeast are subject to dynamic changes both in conformation and vegetative cover. The most pervasive and important changes that have been reported since 1998 are erosion and the spread of invasive plants.

Most islands in the Gulf of Maine are rocky and consequently stable, but colony sites from Cape Cod westward are either on islands composed of glacial till or on barrier islands, most of which are subject to erosion, reshaping and overwash. Among islands composed of glacial till, Bird and Ram Islands have eroded

significantly in the past 100 years, and unless action is taken, Ram Island will likely disappear within the next 40 years (ACRE 2009). Erosion and salt water intrusion through the 160+ year-old revetment on Bird Island have reduced the extent of upland habitat suitable for nesting by terns there by approximately 50 percent (MDFW 2002, U.S. Army Corps of Engineers [USACE] 2005).

Falkner Island, Guilford, Connecticut, within the Stewart B. McKinney NWR, is an important roseate tern colony and in the late 1980's and early 1990's, from 140 to 180 pairs of roseate terns nested at Falkner (USFWS 1998), placing it among the five largest colonies in the Northeast population. Erosion of the eastern slope, caused by rain and surface water runoff, was occurring at a rate of 6 inches per year, and was threatening to destabilize the historic Falkner Island Lighthouse (Demos and Paiva 1998). Phase 1 of a shoreline protection and erosion control project, designed by the New England District, USACE, was constructed following the 2000 roseate tern breeding season. A rock revetment was constructed altering roseate tern nesting habitat on the beach at the northwestern, and 60 percent of nesting habitat on the eastern shore of the island (Spendelow and Kuter 2001, Rogers and Spendelow 2007).

Although the USFWS assessed the effects of the proposed project on the endangered roseate tern and worked with the USACOE to modify the project to minimize adverse effects (USFWS 1998a), filling of crevices with gravel was not completed as designed and there was evidence that as many as 20 percent of the chicks raised on the island in 2001 fell into these fissures from which they could not escape (Spendelow and Kuter 2001, Spendelow *et al.* 2002, Rogers and Spendelow 2007). It is not known whether such losses continued in subsequent years, because chicks were not followed as closely as in 2001, but some roseate terns continued to raise chicks successfully in or near to the revetment (Stewart B. McKinney NWR unpubl. data). However, the project has had untoward effects on some of the roseate terns breeding there. The nesting habitats of three of six sub-colonies on the island were directly modified by the revetment, chicks have become entrapped in the labyrinth spaces within the rock, and a small number of adults and fledglings are thought to have died within these spaces (Spendelow 2003; J. Spendelow, pers comm. 2004; R. Potvin, pers. comm. 2009). More importantly, in preventing erosion of the island's east shore, the revetment has removed the source of sand that nourished and helped maintain the spit at the north end of the island. As this spit is starved of material and further erodes, the important nesting habitat that it provides to roseate terns has become degraded in size and quality.

Erosion and its prevention at Falkner Island is an interesting case study that demonstrates that even if significant monetary resources are available to address coastal erosion, adverse affects to roseate tern nesting habitat may continue. Measures to mitigate the adverse effects resulting from this project are found in the Recommendations section.

For comparison, Great Gull Island is protected from erosion with rip-rap boulders that were emplaced long before terns settled on the island. Roseate tern chicks are raised deep under these boulders and productivity has been high for many years (see Table 4). At Bird Island, many chicks move into crevices within the retaining wall and appear to survive well there (J. Spendelov and I. Nisbet, unpubl. data).

Formerly, many roseate terns in the Northeast population nested on barrier islands such as Cedar Beach, New York, and North Monomoy Island, Massachusetts (Nisbet 1980, 1981; USFWS 1998b), and Warner Island, New York (see Table 2). Currently, few do so, except sporadically at Gardiner's Island/Cartwright Point, New York (see Table 2). This site appeared to have been recently overwashed during a USFWS site visit in August 2009 (M. Amaral, unpubl. data). Up to 43 pairs have nested recently at "Minimoy" Island, Massachusetts (see Table 2), but that site also was subject to erosion and was largely abandoned in 2009 to 2010 (Monomoy NWR, unpubl. data). Several roseate terns originally banded as chicks at Monomoy three or more years earlier (and suspected of having nested there in 2007 and/or 2008) showed up at colony sites in Buzzards Bay in 2009 and 2010 and were trapped and color-banded (J. Spendolow, pers. comm.).

Many coastal islands are vegetated primarily with invasive alien species from Europe, or other sources that outcompeted native species long ago. Vegetation cover on these islands is unstable, proliferating in mild years with fertilization from bird guano, but set back sporadically by severe winters or overwash. New species can invade or existing species can outcompete others as conditions change. Examples of invasive species that are affecting habitat quality at some roseate tern sites (*e.g.*, Great Gull Island, New York; Bird, Ram and Penikese Islands, Massachusetts; Seavey Island, New Hampshire; Outer Green and Stratton Islands, Maine) include common reed (*Phragmites*), blackberry (*Rubus*), Japanese bittersweet (*Celastrus*), European grasses, woodbine/Virginia creeper (*Parthenocissus*), wild radish (*Raphanus*), and black mustard (*Brassica*). In Massachusetts, even native plant species such as beach pea (*Lathyrus japonica*), bindweed (*Convolvulus sepium*), ragweed (*Ambrosia artemisiifolia*), dock (*Rumex* spp.), and American beach grass (*Ammophila breviligulata*) reach heights and/or densities that become adverse to nesting terns, even to roseate terns which habitually nest in moderately heavy vegetation (C. Mostello and I. Nisbet, unpubl. data).

### **Foraging Habitat**

During the breeding season, roseate terns forage over shallow coastal waters, sometimes near the colony and at other times at distances of over 20 miles (32 km) (Heinemann 1992). Roseate terns tend to concentrate in places where prey fish are brought close to the surface by the vertical movement of water. Hence, they usually forage over shallow bays, tidal inlets and channels, tide-rips and sandbars over which tidal currents run rapidly (Nisbet 1981; Duffy 1986; Safina

1990; Heinemann 1992; Casey, Kilpatrick, and Lima, unpubl. data 1996). Roseate tern studies strongly suggest that the species is a visual forager (Safina 1990; Heinemann 1992; Casey, Kilpatrick and Lima, unpubl. data 1996; Hatch and Brault 2007; Rock *et al.* 2007). Roseate terns forage mainly by plunge-diving and by contact-dipping or surface dipping over shallow sandbars, reefs, or schools of predatory fish (Gochfeld *et al.* 1998). Gochfeld *et al.* (1998) also report that they tend to fly into the wind, hover, and dive from a height of 3.3 to 20 feet (1 to 6 meters), and up to 40 ft. (12 m) at times.

In the only foraging study of roseate terns within the Northeast population that utilized telemetry, Rock *et al.* (2007) found that while roseate terns nesting at County Island, Nova Scotia, sometimes foraged as far as 7.2 miles (24 km) from the colony, on average they foraged much closer, 2.1 mi (7 km), and especially in locations within 6 miles (10 km) of the colony, at water depths less than 16.5 ft. (5 m). The authors recommended that critical foraging habitat for the roseate terns at County Island, *i.e.*, shallow areas (< 5 m depth) within 10 km of the colony, should be protected (Rock *et al.* 2007).

### **2.3.1.7 Other:**

#### **2.3.1.7.1 Predation**

Predation by one or more predatory species has been reported at all roseate tern breeding sites that have been monitored in the period 1998 to 2009. Regionwide, the incidence of predation appears to have been increasing, and some sites such as Great Gull Island, New York, and Bird Island, Massachusetts, that had little or no predation in the 1970s and 1980s were affected in some or most years in the 1990s and 2000s. The predators are listed below in approximate descending order of importance, based on the number and size of colony sites affected and the severity of their effects.

Great horned owl (*Bubo virginianus*). Predation by great horned owls has been documented at five roseate tern colony sites in the period 1998 to 2009: Pond Island, Maine (annually; up to 4 owls trapped and removed per year); Jenny Island, Maine (intermittently); South Monomoy Island, Massachusetts (most years), Bird Island, Massachusetts (2001 to 2003), and Ram Island, Massachusetts (2005 to 2008), as well as at several sites in Canada, including The Brothers Islands (see below). Owl predation on common and arctic terns was reported at several other sites in Maine during the same period (Gulf of Maine Seabird Working Group [GOMSWG] annual reports). Prior to 1998, owl predation had been recorded at several other sites in Massachusetts: Tern Island, Gray's Beach, Nauset New Island, North Monomoy Island and Dead Neck/Sampson's Island (Nisbet 1981; I. Nisbet, unpubl. data). However, it appears to have increased in frequency and has occurred since the mid-1990s at Bird and Ram Islands, where it was formerly unrecorded.

Characteristic features of predation on both roseate and common terns by great horned owls were described by Nisbet (1975, 1981) and Nisbet and Welton (1984). Owls usually start raiding tern colonies early in the season and kill a few adult terns, but the terns then start deserting the colony at night and few terns are then taken until the chicks start to hatch. The owls then start taking small chicks, sometimes in large numbers, while other chicks die of exposure during cold nights, or are attacked by ants or black-crowned night-herons (see below) while their parents are absent. In proportion to relative numbers, predation by owls typically falls more heavily on adult roseates than on adult commons, but much more heavily on common tern chicks (Nisbet 1975, 1981; Nisbet and Welton 1984; I. Nisbet, unpubl. data.). We know of no documented case where substantial numbers of roseate tern chicks were taken; however, as many as 30 adult roseate terns have been killed in a single year. Numbers of adult roseates known to have been killed by great horned owls in this period 1998 to 2009 are as follows: Bird Island, 23; Ram Island, 42; The Brothers, 11. This tally is likely to be incomplete (USFWS, unpubl. data).

A frequent response of common and roseate terns to predation by owls, or other nocturnal predators such as mink (*Mustela vison*) and black-crowned night-herons (*Nycticorax nycticorax*), is to desert the colony and move to another site, either in the same or in the next year. Roseate terns are more prone to move than are common terns, which often stay for several years after the roseates leave. Table 14 shows peak period counts of roseate tern pairs at the three Buzzards Bay sites in 1996 to 2006. Numbers fluctuated much more at individual sites than in Buzzards Bay as a whole. The fluctuations were reciprocal, with decreases at Ram Island matched by increases at Bird and Penikese Islands, and *vice versa*. The decreases were associated with either substantial predation by owls or the oil-spill related hazing that occurred at Ram Island in 2003. In years when the predation or hazing at Ram Island occurred at the time of egg-laying, roseate terns appear to have moved in the same season, before laying. In years when predation started after egg-laying, roseate terns appeared to have moved in the next year. The numbers of pairs that moved on each occasion were on the order of 400 to 600 pairs. In 2003, the hazing at Ram Island resulted in a shift to Bird and Penikese Islands, in spite of the owl predation at Bird Island in 2002 that would have been predicted to cause a shift away from that site in 2003. Several other examples were summarized by Nisbet (1981).

Mink. We know of only one report of predation by mink on roseate terns in the Northeast prior to 1998 (Anon. 1988). Since 1998, predation by mink has been reported at Jenny Island, Maine (1998 to 1999, 2007 to 2008), The Brothers Islands, Nova Scotia (2003 to 2004), Stratton Island, Maine (2003), Bird Island, Massachusetts (2004), and Outer Green Island, Maine (2005). Predation by mink on common and arctic terns has also



been reported at Petit Manan, Ship, and Pond Islands, Maine, during the 2000s. Mink predation can be particularly problematic due to the difficulty of trapping them, their efficient predation on adult terns, and their habit of surplus killing. As with other predators that prey on adult terns, predation by mink usually falls disproportionately on roseate terns, whereas predation on eggs and chicks falls disproportionately on common terns. The disproportionate effect on adult roseate terns probably indicates that they have a greater tendency to remain at their nests in the face of danger; in the case of mink, it may also result from the roseates nesting under cover, so that they do not see the mink coming. The disproportionate effect on common tern eggs and chicks probably results from the fact that common terns nest in the open, and mink hunt visually by day (I. Nisbet, unpubl. data).

**Table 14.** Numbers of roseate tern pairs at the three colony sites in Buzzards Bay sites, showing reciprocal fluctuations at Ram, Bird and Penikese Islands, associated with predation and hazing. P indicates moderate to heavy predation by Great Horned Owls; H indicates hazing; E indicates that the event occurred early enough in the breeding season for birds to change sites in the same season, before laying eggs; L indicates that the event occurred after most birds had laid eggs, so that change in sites would have occurred in the following year.

Year	Bird Island	Ram Island	Penikese Island	Total
1996	996	656		1652
1997	1179	253 PE		1432
1998	1113	543		1656
1999	1143	630		1778
2000	1130	988		2128
2001	1062 PL	626		1688
2002	503 PL	952		1457
2003	904 PL	557 HE	251	1712
2004	554	936	9	1499
2005	680	724 PL	76	1430
2006	1111	463 PE	48	1622

At Bird Island in 2004, a mink killed 23 adult roseates in four nights, but then left the island (MDFW, annual report). At The Brothers Islands, Nova Scotia, mink killed at least 5 adult roseate terns and many more chicks in 2003 and 10 to 12 adult roseate terns in 2004. No roseate tern chicks fledged from The Brothers in those years (T. D'Eon *in* COSEWIC 2009). Between 2003 to 2008, almost 10 percent of the adult population of roseate terns nesting at the two largest colonies in Canada (Country Island and The Brothers) was killed by mink, and the population there declined from 130 to 100 pairs during that period (COSEWIC 2009).

Black-crowned Night-Heron. Black-crowned night-herons were regarded as major predators on terns (primarily common terns) in the Northeast at the time the first update of the recovery plan for the Northeast population was issued (USFWS 1998b). Since then, substantial predation by this species on roseate terns has been reported only at Falkner Island, Connecticut, and Stratton Island, Maine. At Falkner Island, predation by night-herons was first recorded in 1996 and increased in the following years in spite of repeated control efforts. Initially, night-herons mainly took common tern eggs and chicks around the time of hatching, and predation on roseate terns was relatively light. In 2002, a single night-heron started taking much larger numbers of eggs early in incubation, including most clutches of roseate terns on the island, even though most of these were concealed deep within tires. Although this night-heron was later removed, more than half of the roseate terns left the site in 2002 and 2003, and the colony has continued to dwindle (see Table 2). At Stratton Island, night-herons took large numbers of eggs of both common and roseate terns in 2002 and 2004; black-crowned night-herons nest on the island and most of the predation on terns was carried out by one or two specialist individuals (S. Hall, unpubl. data). Black-crowned night-herons have been reported from several other sites since 1998 and have taken large numbers of common tern eggs and chicks at Monomoy Island, Massachusetts (USFWS, unpubl. data). Besides direct predation, black-crowned night-herons cause nocturnal abandonment and this may have contributed to low productivity of roseates at Stratton Island and other sites (S. Hall, pers. comm.).

Herring and Great Black-backed Gulls (*Larus argentatus* and *L. marinus*). In Canada, predation and displacement by herring gulls and great black-backed gulls are thought to be the main factors limiting distributions of terns, including roseate terns (Lock *et al.* 1993, Whitman 1999), and these gulls are considered the major avian predators of roseate terns there (COSEWIC 2009). At Country Island, Nova Scotia, gull predation is reported to be the main factor limiting tern productivity (Environment Canada 2006). In the United States, there is often heavy predation by herring and great black-backed gulls on eggs and chicks of common and arctic terns at breeding sites in Maine (GOMSWG, minutes of annual meetings), but there are only a few documented cases of predation on eggs or chicks of roseate terns in the period 1998 to 2009 (*e.g.*, at Stratton Island in 2002 and 2004, and at Penikese Island, Massachusetts, in 2004 to 2006; see below).

South of Maine, the main effect of herring and great black-backed gulls on terns is indirect, by occupying islands or habitats within islands that were formerly used by terns. This was identified as the main factor involved in the reduction in numbers and site occupation by terns, including roseate terns, in the Northeast between 1930 and 1980 (Nisbet 1980, Gochfeld *et al.* 1998, USFWS 1998b). In Canada, displacement by gulls is still

considered an important threat, *e.g.*, a colony of at least 200 pairs of terns (mixed species) at Pearl Island, Nova Scotia, was apparently displaced by gulls in 2007 (COSEWIC 2009). In the United States, however, displacement by herring and great black-backed gulls had become a relatively minor factor by 1998, with the cessation of expansion by these two gull species in the Northeast, the decline in numbers of herring gulls starting about 1980 and of great black-backed gulls about 1995 (Nisbet *et al.* in press), and the restoration of several important former tern colony sites by controlling gulls. Programs to remove herring and great black-backed gulls from potential tern nesting islands or from areas within islands, by lethal or non-lethal means, have been conducted at 18 sites since the 1960s (Kress and Hall 2004; Nisbet 1980; I. Nisbet, unpubl. data; S. Hall pers. comm.). Start dates for these programs were as follows: Bird Island, Massachusetts (1967), Tern Island, Massachusetts (1968), Eastern Egg Rock, Maine (1974), Monomoy Islands NWR, Massachusetts (1980, 1994), Seal Island MWR, Maine (1984), Stratton Island, Maine (1986), Jenny Island, Maine (1991), Ram Island, Massachusetts (1992), Gardiner's Point Island, New York (1993), Ship and Trumpet Islands, Maine (1994), Pond Island NWR, Maine (1996), Metinic Island, Maine (1996), Seavey Island, New Hampshire (1997), Penikese Island, Massachusetts (1998), Muskeget Island, Massachusetts (2000), Outer Green Island, Maine (2002), Eastern Brothers Island, Maine (2007); and Sheep Island, New Brunswick (2002). All of these programs resulted in increases in numbers of common terns for 1 or more years, 12 of 18 resulted in more or less permanent establishment of common terns, and 7 of 18 resulted in more or less permanent establishment of roseate terns (see Tables 1 and 2), including what are now two of the largest colonies of roseate terns in the region. Intensive management of gulls continued for several years at some of these sites, and low-level management continues in each year at all sites, to prevent gulls from resettling or encroaching on tern nesting areas, and to remove specialist individuals that are preying on terns (Kress and Hall 2004; USFWS, unpubl. data.).

Even after establishment of terns in gull-free areas, herring and great black-backed gulls (often non-breeders or visitors from other sites) frequently continue to prey on terns. Both species sometimes take eggs or chicks; most such predation is by one or a few individual gulls that specialize in taking terns (Kress and Hall 2004; USFWS, unpubl. data.). This predation is locally significant for common and arctic terns, especially in Canada and Maine (see above), but there are few documented records of herring or great black-backed gulls preying on roseate tern eggs or chicks. Heavy predation on eggs of both common and roseate terns was recorded at Penikese Island, Massachusetts, in 2004, 2005, and 2006 and was traced to great black-backed gulls that took eggs at night (MDFW, annual reports). This nocturnal predation and accompanying disturbance may have contributed to the high frequency of nest desertion by roseate

terns and the fluctuating numbers nesting at Penikese Island in those years (Table 15). Similar nocturnal predation on common tern eggs at Stratton Island, Maine, in 1994 and 1995 was described by Nocera and Kress (1996). After chicks fledge, both gull species (especially great black-backed gulls) take fledglings in the air at many sites. Again, however, this predation infrequently affects roseate terns, because they fly better than common terns at fledging and are more closely attended by their parents (I. Nisbet, unpubl. data).

Raccoon (*Procyon lotor*). A raccoon killed 14 adult roseate terns at Bird Island in May 2007 before it was trapped and removed (MDFW, annual report 2007). A mammalian predator, possibly a raccoon, killed some adults and chicks at Great Gull Island in 2005 and 2006 (G. Cormons, Great Gull Island annual reports), but the identity of the predator was not confirmed and numbers of roseate terns that were killed were not reported.

Peregrine Falcon (*Falco peregrinus*). Peregrine falcon numbers in the northeastern United States (based on counts of territorial pairs) more than doubled between 1998 and 2009 (M. Amaral, unpubl. data). Also, peregrine falcon breeding pairs now occur in proximity to the coast in most northeastern States and the province of Nova Scotia. In Maine, peregrine falcons believed to be nesting at Acadia National Park make frequent forays, often several times each day, to the tern colony at Petit Manan Island, where they take adult terns and fledglings and cause severe disturbance and temporary colony abandonment (L. Welch, unpubl. data). Peregrine falcons are seen sporadically at many other colony sites during the breeding season. They occasionally take terns (mainly common terns) and cause considerable disruption, even when they do not kill any. There has been no recurrence of the heavy predation on adult roseate terns that occurred at Bird Island in 1991 (Nisbet 1992).

Ruddy Turnstone (*Arenaria interpres*). Ruddy turnstones are present at many tern nesting islands during May and early June. They often enter tern nesting areas and take eggs, sometimes in large numbers, as at Bird Island in 2005 (Kress and Hall 2004; MDFW, unpubl. data). Although this predation usually falls more heavily on common terns, ruddy turnstones regularly take roseate tern eggs, even when these are concealed under vegetation or in nest boxes (MDFW, unpubl. data).

Canada Goose (*Branta canadensis*). Canada Geese nesting at Bird Island, Massachusetts, began taking tern eggs in 1998 and did so in increasing numbers until 2003, when control measures were started. As with most egg predators, this predation fell most heavily on common terns, but many roseate tern eggs were taken also, even from within nest boxes (MDFW, annual reports).

Laughing Gull (*Larus atricilla*). Laughing gulls breed in several tern colony sites from the Monomoy Islands north through the Gulf of Maine to Petit Manan Island. They have been increasing rapidly and are expanding into areas used for nesting by terns, especially roseate terns (National Audubon Society and Monomoy and Maine Coastal Islands NWRs, unpubl. data). At the edges of laughing gull nesting areas, they regularly destroy tern eggs or kill chicks in neighboring territories. Because they occupy similar microhabitats to roseate terns, their territories often abut, and it is likely that laughing gulls take roseate eggs and chicks. However, such predation would be difficult to observe and there are no documented records of their doing so (L. Welch, unpubl. data). Laughing gulls have also displaced roseate terns from preferred habitats on several islands (L. Welch, pers. comm.). In recent years, control measures to limit these adverse effects have been started at most of the sites in the region where they nest (GOMSWG, minutes of annual meetings).

Unidentified mammal. Rats were formerly important predators of common and roseate terns at several sites in Massachusetts (Nisbet 1981), but no case of substantial predation by rats has been reported in the region in the period 1998 to 2009. An unidentified mammal took large numbers of eggs of both species at Ram Island, Massachusetts, in 2004, but the signs of predation suggested a smaller mammal than a rat. The only mammal trapped was a short-tailed shrew (*Blarina brevicauda*) (MDFW, annual report).

Other predators. Other predators on roseate terns that have been reported in the region in the period 1998 to 2009 include red fox (*Vulpes vulpes*), northern raven (*Corvus corax*), and American crow (*C. brachyrhychos*), mainly in Canada. Other predators recorded at colonies in the United States include snowy owl (*Nyctea scandiaca*) (Stratton Island, Maine, 2002; Seavey Island, New Hampshire, 2009); ants (Penikese Island, sporadically); northern harrier (*Circus cyaneus*) (South Monomoy and Penikese Islands, annually); and coyote (*Canis latrans*) (Monomoy Islands, controlled annually). However, there were no documented cases of predation by these predators on roseate terns during this period.

*Summary.* Overall, although many different predators have been recorded preying on roseate terns in the region, the most important population effects have been the killing of adult terns by great horned owls, mink, and raccoons. Together, these predators are known to have killed 102 adult roseates in the warm water subregion and 29 in the cold water subregion in the period 1998 to 2009. It is not clear that all kills were reported, and many more are likely to have been missed, especially at Great Gull Island. At the Canadian sites, these kills approached 2 percent of the adult population annually during 2003 to 2008 (see above). In the United States, they amounted to less than 1 percent of the adult population

annually in both the cold water and warm water subregions. By itself, this would not be a major drain on the population, but predation by these three species also causes major disruption of nesting and has led to repeated shifts away from affected sites, sometimes to sites that are less suitable. Predation by gulls and black-crowned night-herons has also been important at several of the smaller colony sites. The overall impact of predation on roseate terns would undoubtedly have been much greater in the absence of the intensive monitoring and predator management efforts that are conducted annually at all the major nesting sites.

#### **2.3.1.7.2 Disease and parasites**

Occasionally, individual roseate terns are found dead and predation is not the apparent cause. For example, the necropsy of a roseate tern found dead in Barnstable, Massachusetts, in 2009 revealed the presence of aspergillosis, causing fungal pneumonia and air sacculitis (U.S. Geological Survey/National Wildlife Health Center 2009). This bird tested negative for avian influenza. While disease is sometimes found to have contributed to the death of individual birds, it is infrequent and not known to cause population level effects in this species.

#### **2.3.1.7.3 Winter quarters, migration and staging**

Except for the predation events previously reported in section 2.3.1.7.1, most mortality of roseate terns takes place away from the breeding sites, during staging or migration or while in winter quarters (Nisbet 1981, Gochfeld *et al.* 1998). However, only little information is available about distribution, movements or ecology during these periods, and nothing is known about causes of death. Although fairly intensive studies have been conducted since 1998, the results of these studies remain largely unpublished.

Based on band recoveries (Nisbet 1984, Hays *et al.* 1997), northeastern roseate terns are thought to migrate through the eastern Caribbean and along the north coast of South America, and to winter mainly on the east coast of Brazil between 10° and 18° S (Gochfeld *et al.* 1998). The Northeast population recovery plan update (USFWS 1998b) summarized observations by H. Hays, P. Lima and others at Mangue Seco, Bahía (11° S) in 1995 to 1997. These observations have subsequently been published (Hays *et al.* 1999). At Mangue Seco, roseate and common terns apparently fed at sea during the daytime, arrived at the roosting site (where terns rest over night) after dark, and left before first light. Ninety roseates were caught in mist nets in December 1996 and February 1997, but only seven of these had been banded in the Northeast (versus seven from the Caribbean, where far fewer birds had been banded in prior years)

(Hays *et al.* 1999). Hence, it appeared that most of the birds roosting at the site were from the Caribbean population.

In collaboration with Pedro Lima (of Brazil), H. Hays has continued to search for and trap roseate terns in Brazil and has found other roosting sites (RTRT minutes), but no details are available. The only published information on the daytime activity of roseate terns in the winter quarters is a single observation of about 17 in a mixed flock with common terns resting on the water 11 km offshore at 11° 30' S in March 1995 (Hays *et al.* 1997).

C. Mostello and I. Nisbet (unpubl. data) attached geolocators (light-level data loggers) to roseate terns at Bird Island, Massachusetts, in 2007 and 2009 and retrieved six (two in 2008 and four in 2010) when the birds returned to Bird Island in the following year. To date, data have been downloaded and analyzed for only the first two birds. Both spent the winter along the east coast of Brazil (11° to 18° S). They staged around and south of Cape Cod in July and August, migrated directly from Cape Cod to the eastern Caribbean in late August, and staged in Puerto Rico and northern Brazil on their migration to eastern Brazil. The one bird tracked in spring staged in northern Brazil, Puerto Rico and the Dominican Republic, before returning to Bird Island. Both birds were partly pelagic while staging and on migration, but not in winter. Downloading and analysis of data from the other four birds is ongoing (C. Mostello and I. Nisbet, unpubl. data).

Migrants from the northwestern Atlantic breeding population may pass through the nesting range used by the locally breeding Caribbean population from May to early June during spring migration, and again from late August to September during fall migration (J. Spendelow, unpubl. data).

Trull *et al.* (1999) reported on staging of roseate terns around Cape Cod and islands to the south (Nantucket and Martha's Vineyard) in the post-breeding period based on counts of resting flocks, aerial surveys and sightings of FR and color-bands over several years. They identified 20 sites around Cape Cod where roseate terns (and common terns) staged during daylight hours between 24 July and 22 September. All sites were on open beaches or sand flats, usually near the end of barrier beaches. Roseate terns had been recorded in thousands at four of the sites. Individually-marked birds from eight breeding sites throughout the range were identified among staging flocks. Only two roosting sites were identified, one of which was at South Beach, Chatham, where large numbers had roosted for many years. Roseate terns appeared to disperse throughout the breeding range in July and August, re-aggregating on outer Cape Cod in late August and early September prior to southward

migration in mid-September. Trull *et al.* (1999) suggested that at least half of the entire Northeast population was concentrated around Cape Cod at that time. Terns at the staging sites were seen to be disturbed by human pedestrians (11 of 20 sites), dogs (6 of 20), beach vehicles (6 of 20), or aircrafts (2 of 20). Gulls were seen taking terns (of one or both species) at two sites, disturbing terns at four others, and displacing terns at five.

Since 2007, J. Spendelow (unpubl. data), Blake (2010), and personnel of the Massachusetts Audubon Society (MAS) (Jedrey *et al.* 2010; MAS, unpubl. data) have extended Trull *et al.*'s (1999) work with intensive observations throughout the same area. Among 54 sites visited on a regular basis in 1 or more years, 15 sites supported >1000 terns, with a high percentage of roseate terns on a regular basis (Figure 3). These sites had a total area of only 12 km<sup>2</sup> at low tide and the terns that used these sites were regularly disturbed by human pedestrians, dogs, motor vehicles and/or aircraft.

Blake (2010) reported on overlapping surveys of roseate and common terns staging at 35 sites around Cape Cod and the islands in 2007 to 2008. Terns were present in thousands at 15 sites, and in tens of thousands at four sites; one site (Coastguard Beach/Nauset Marsh, Orleans) at times supported a substantial fraction of the entire Northeast population of roseate terns. Terns at this site sometimes rested on salt marsh, as well as on the beach or tidal flats. Terns were thought to be roosting at four sites. As in the study by Trull *et al.* (1999), terns appeared to congregate in largest numbers at sites along outer Cape Cod in the first three weeks of September, just prior to southward migration.

H. Goyert (College of Staten Island, City University of New York) is currently studying foraging of roseate and other terns and their distribution offshore in May to September. Preliminary analysis of data from 2006 to 2009 shows common and roseate terns to be widely distributed in small numbers at sea southeast of Cape Cod and throughout the Gulf of Maine, east to the southeast edge of Georges Bank. There was a statistical association of terns with schools of tuna (*Thynnus* spp.) (H. Goyert, unpubl. data). A flock of terns, including roseate terns, was photographed while resting on the sea on September 27, 2006 (M. Martin, unpubl. data). These studies are ongoing, and it remains to be determined whether numbers of roseate terns dispersed offshore are comparable with those concentrated inshore.

#### **2.3.1.7.4 Food and foraging**

Since 1999, intensive observations of chick provisioning by roseate terns at several breeding sites in Maine have been conducted by S. Hall and seasonal personnel of National Audubon Society's Project Puffin (S. Hall,



unpubl. data). Table 15 summarizes data collected at Stratton Island and Eastern Egg Rock, the two sites with the most comprehensive data. The mean provisioning rate was higher at Stratton Island than at Eastern Egg Rock (1.39 vs. 0.95 food items/brood/hour, respectively). The predominant prey species in the diet were American Sand Lance (*Ammodytes americanus*: 78.9 percent by frequency) at Stratton Island, and hake spp. (61.0 percent) and Atlantic Herring (*Clupea harengus*: 26.4 percent) at Eastern Egg Rock. In this respect, the diet at Stratton Island was similar to those recorded at breeding sites to the south and southwest (Gochfeld *et al.* 1998), whereas that at Eastern Egg Rock was markedly different. Diet compositions varied markedly from year to year, for example, the frequency of sand lance at Stratton Island varied among years from 30 percent to 96 percent, with herring and hake as the main alternative; the frequency of hake at Eastern Egg Rock varied from 24 percent to 79 percent, with herring as the main alternative. (Gochfeld *et al.* 1998).

Kimmons (unpubl. dissertation) used stable isotopes of nitrogen ( $N^{15}$ ) and carbon ( $C^{13}$ ) in feathers to characterize the total diet of arctic, common, and roseate tern chicks. His results indicate that diets of roseate tern chicks were similar to those of common tern chicks at the same sites, Stratton Island, Outer Green Island, and Eastern Egg Rock. The stable isotope signatures were consistent with the diets recorded by observing fish fed to chicks at Stratton Island (Table 16) but did not support the visual observations from Eastern Egg Rock, suggesting either that some fish were misidentified there or that the “unidentified” fish included fish with markedly different isotope signatures from those of fish that were identified.

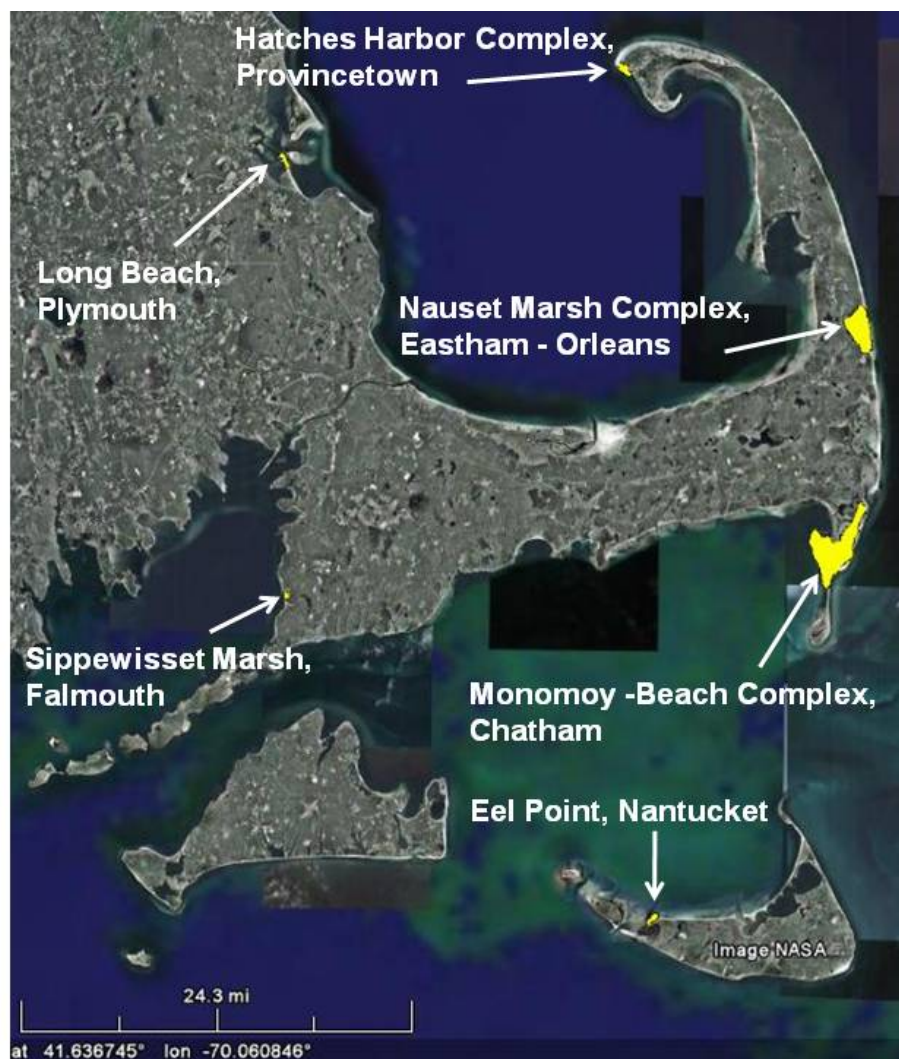
H. Goyert (College of Staten Island, City University of New York) is currently studying foraging of roseate and other terns and their provisioning of chicks at Bird Island, Massachusetts. Preliminary analysis of data from 2009 indicate that the predominant direction of departures of commuting roseate terns was to the southeast (towards Woods Hole and Vineyard Sound), with proportionately fewer departures to the NNE (towards Mashnee Flats) than were reported by Heinemann (1992) at the same site. Directions of arrival of roseate terns carrying fish were similar to directions of departure. Among prey ( $N = 245$ ) delivered to roseate tern chicks in June to July, 86.9 percent were American sand lance, 7.8 percent Atlantic herring, and 2.4 percent anchovies (*Anchoa* spp.). The mean provisioning rate was about 0.8 fish/brood/hour. Mean fish lengths were 1.9 bill-lengths (bl) for sand lance, 1.5 bl for herring and 1.6 bl for anchovy. This information is consistent with that previously reported for this site (Gochfeld *et al.* 1998).

**Figure 3.** Sites in southeastern coastal Massachusetts with greater than 1000 roseate terns on a regular basis during the post-breeding period.

# **Sites w/> 1000 terns and high % of ROST on a Regular Basis**

## **Total**

	Area (km <sup>2</sup> )
Hatches Harbor	0.25
Wood End	1.23
Nauset Marsh Complex	0.65
North Beach mainland	0.54
South Beach	1.29
South Monomoy Island	0.50
Eel Point	0.02
Smith's Point/Madaket	0.57
Madaket Flats	0.07
Esther Island	0.09
Muskeget Shoals	0.21
Great Point -The Galls	0.02
Katama	0.06



**Table 15.** Composition of food items fed to Roseate Tern chicks at two colony sites in the Gulf of Maine, 1999 to 2009 (Source, National Audubon Society, unpubl. data).

	Stratton Island	Eastern Egg Rock
Total food items	4,548	3,257
Mean feeding rate (items/brood/hour)	1.39	0.95
Diet by frequency <sup>1</sup>		
American sand lance	78.9 percent	5.7 percent
Hake spp.	11.5 percent	61.0 percent
Atlantic herring	8.7 percent	26.4 percent
All others	0.9 percent	7.9 percent

<sup>1</sup> Percentages of food items that were identified: unidentified items comprised 13.2 percent of the total at Stratton Island and 27.9 percent at Eastern Egg Rock.

**Table 16.** Number of sites occupied by breeding pairs of roseate terns in the northeastern United States.

Year	1998	99	2000	01	02	03	04	05	06	07	08	09
# Sites	20	19	15	15	14	21	23	24	19	17	16	13

Small-scale studies of chick-provisioning were conducted at Penikese Island, Massachusetts, in 2005 to 2009. Pooling data from all watches, the mean provisioning rate was 0.97 fish/brood/hour or 0.61 fish/chick/hour; 72 percent of fish identified (N = 121) were American sand lance and 18 percent were Atlantic herring (MDFW, unpubl. data).

Shealer and Spendelov (2002) and Shealer *et al.* (2005b) studied kleptoparasitism (food-stealing) among roseate terns breeding at Falkner Island, Connecticut. Ten birds (3 to 4 percent of the total number present at the site) specialized in stealing food from common terns, and this behavior was consistent over 10 years. Most (8 of 10) of the specialized kleptoparasites were females. These birds never went to sea to fish, but made sallies from their nest sites to seize fish from common terns, usually as the common terns were passing food to their own chicks (Shealer and Spendelov 2002). These tactics were efficient and profitable, and Shealer *et al.* (2005b) reported that pairs in which one partner was a kleptoparasite consistently raised more chicks than pairs that foraged at sea.

#### **2.3.1.7.5 Human disturbance**

Nisbet (2000) published a Commentary on the relationship between human disturbance and breeding colonial waterbirds; with a detailed review of previously published reports of effects of disturbance on terns, including roseate terns. Nisbet (2000) concluded that there was little scientifically acceptable evidence that human disturbance has substantial adverse effects on terns, and that terns are capable of habituating to high levels of investigator activity and visits by non-investigators to their colony sites. In the Northeast, almost all roseate tern colony sites have high levels of human activity from investigators and/or managers. Both roseate and common terns have high tolerance for human activity around and within the nesting areas. A high tolerance for hands-on research, such as weighing chicks and trapping adults, has been acquired through many years of monitoring and research and may increase the birds' tolerance to casual human intrusions by non-investigators (Zingo *et al.* 1997, Zingo 1998). Roseate and common terns have less tolerance for human activity while staging in the post-breeding period (I. Nisbet and J. Spendelow, pers. comm.)

Although they generally congregate at the ends of barrier beaches or at other sites that are relatively remote from human activity, they are regularly disturbed there by pedestrians, dogs and vehicles (Trull *et al.* 1999; MAS and J. Spendelow, unpubl. data). They do not allow such close approach at the staging sites as they do at the breeding sites, and consequently spend much time flying, especially at high tides when space is limited for both birds and humans.

There is little information on the extent of human disturbance and the terns' response to it in the winter quarters in South America. However, common and other tern species are known to rest and roost on beaches and sand flats similar to those used when staging around Cape Cod; and roseate terns are thought to frequent some of the same locations, at least for roosting at night. Since they are present there in the local summer, it is possible that they are similarly exposed to human recreational activity.

### **2.3.2 Five-factor analysis:**

#### **2.3.2.1 Factor A. Present or threatened destruction, modification or curtailment of its habitat or range:**

The 1987 final listing rule (52 FR 42065) and the 1998 recovery plan (USFWS 1998b) address the primary cause for the loss of nesting habitat—the expansion of breeding herring gulls and great black-backed gulls onto islands used for nesting by terns. As populations of these large and aggressive gulls expanded in the Northeast during the mid-20th century, terns were displaced from their favored

nesting sites. While recent evidence suggests that populations of these large gulls have now stabilized or are declining (*e.g.*, Welch *et al.* 2010, Nisbet *et al.* in press) populations of laughing gulls have expanded on Petit Manan Island, Maine, to the detriment of the roseate tern population there (RTRT 2008). See NE section 2.3.1.7.1 for more detail.

The coastal islands used for nesting by roseate terns in the Northeast are subject to dynamic changes both in conformation and vegetative cover (see NE section 2.3.1.6). The most pervasive and important changes that have been reported since 1998 are erosion and the spread of invasive plants. The erosion of critical nesting areas such as Bird and Ram Islands, Buzzards Bay, Massachusetts, the north spit on Falkner Island, Connecticut, The Brothers, Nova Scotia, and others are discussed in NE section 2.3.1.6. The loss and degradation of nesting habitat due to erosion is among the most significant threats facing the species.

In regard to new information pertaining to roseate tern habitat, the 1998 recovery plan did not reference climate change or sea level rise as potential threats to the species. Although erosion and overwashing of nests during high tides and storm surges (symptomatic of rising sea levels) were discussed as a general threat affecting some colonies in the 1998 plan, climate change may have additional broad-reaching effects on marine food webs as well (North American Bird Conservation Initiative [NABCI] 2010). This recent report identifies oceanic birds as “vulnerable” because of their low reproductive potential, use of islands for nesting, and reliance on rapidly changing marine ecosystems.

Over the past 100 years, the globally-averaged sea level has risen approximately 10 to 25 centimeters (Rahmstorf *et al.* 2007), a rate that is an order of magnitude greater than that observed in the past several 1000 years (Douglas 2001 as cited in Hopkinson *et al.* 2008). The IPCC suggests that by 2080, sea level rise could convert as much as 33 percent of the world’s coastal wetlands to open water (IPCC 2007). Although rapid changes in sea level are predicted, estimated time frames and resulting water levels vary due to the uncertainty about global temperature projections and the rate of ice sheets melting and slipping into the ocean (IPCC 2007, CCSP 2008).

Potential effects of sea level rise on coastal beaches may vary regionally due to subsidence or uplift, as well as the geological character of the coast and nearshore (CCSP 2009, Galbraith *et al.* 2002, Gutierrez *et al.* 2007). In the last century, for example, sea level rise along the U.S. Gulf Coast exceeded the global average by 13 to 15 cm, because coastal lands there are subsiding (EPA 2009). Sediment compaction and oil and gas extraction compound tectonic subsidence (Penland and Ramsey 1990, Morton *et al.* 2003, Hopkinson *et al.* 2008). Low elevations and proximity to the coast make all non-breeding foraging and roosting habitats used by roseate terns vulnerable to the effects.

Rising sea levels are expected to inundate many habitats used by roseate terns during their annual cycle, such as sandy beaches, barrier islands, and sand flats (NABCI 2010). Several important roseate tern breeding sites are low profile islands with significant area less than 10 feet above sea level (*e.g.*, Ram and Bird Islands, MA, North Brothers Island, Nova Scotia, and Gardner's Island/Cartwright Point, NY). All these sites are seriously threatened by erosion; see NE section 2.3.1.6 for more detail on those sites and for comments on an erosion control program at Falkner Island, CT. Effects on the integrity of these and other nesting sites throughout the species range as a result of sea level rise have not been adequately evaluated.

Roseate terns forage over sand bars and shoals, where tidal forces and shallow water depths make small prey fish more available to plunge-diving birds like terns. If exposed during lower tides, roseate terns also favor these habitats for resting and roosting, as they often have less human recreational disturbance than beaches nearby. Sand mining, the practice of extracting (dredging) sand from sand bars, shoals, and inlets in the near shore zone, is a potential threat to these habitats because removing these sand sources can alter depth contours (Hayes and Michel 2008), and may make them unsuitable for use by foraging and resting roseate terns. We do not have a good estimate of the amount of sand mining that occurs across the roseate tern breeding range, but due to the increasing demand for sand to replenish eroding beaches, it is anticipated the practice will increase.

A recently identified threat to roseate tern habitat, not described in the listing rule or recovery plans, is the spread of invasive plants into roseate breeding habitat. Like invasive species elsewhere, exotic and invasive plants in the coastal environment may reproduce and spread quickly, often out-competing native plant species. Invasive plants have the potential to alter the structure, microhabitat and accessibility of island habitats used by nesting roseate terns. If left uncontrolled, invasive plants may create dense, impenetrable clumps or otherwise convert sparsely vegetated sand to dense vegetation, resulting in the loss or degradation of roseate nesting habitat. Invasive plants are seriously affecting habitat quality at several important roseate tern colony sites (*e.g.*, Great Gull Island, New York, Seavey Island, New Hampshire, Bird and Penikese Islands, Massachusetts, and Outer Green Island and Eastern Egg Rock, Maine). Although attempts are made at many breeding sites to control invasive plants and to maintain suitable habitat for both roseate and common terns, this has proved both challenging and labor-intensive. None of the current programs can yet be regarded as successful. See NE Section 2.3.1.6.

Curtailement of the species' range was identified as a factor during the listing of the species and is addressed through the inclusion of the secondary recovery objective, expanding: (1) The number of breeding colonies to 30 or more, and (2) the range to historically occupied areas south of the current range (USFWS 1998b). Since 1998, the number of sites in the northeastern United States with documented breeding pairs has ranged from 13 to 21 (see Table 16). In general,

there has been a continued northward contraction of the breeding range, as former breeding sites along southern Long Island, Shinnecock Bay to Rockaway Inlet, have become unoccupied (see Figure 1 and NE section 2.3.1.5).

In Canada, roseate terns have bred at 28 different sites since 1982 (Environment Canada 2006). Most of these sites are coastal islands in Nova Scotia and have never contained more than a few pairs. Only 3 sites have supported 20 or more pairs in the last 10 years: The Brothers, Grassy Island, and Country Island.

After roseate tern chicks fledge, the birds depart their nesting colonies and congregate at post-breeding staging areas where they rest, forage and put on energy reserves needed for their migration to South America for the winter (see NE section 2.3.1.7.3). It is also at this time that young of the year practice their flying and foraging skills. Habitats important to the species during fall staging are sand flats or beaches, usually at or near the end of barrier beaches far from human access (Trull *et al.* 1999). As the summer season progresses, roseate terns from throughout the North Atlantic breeding range mix with large staging flocks of common terns and collect on the beaches and sand spits of southeastern Massachusetts, particularly outer Cape Cod and the islands (Trull *et al.* 1999). More recently (2007 to present), the Massachusetts Audubon Society, U.S. Geological Survey, and other cooperators have studied habitat use and post-breeding movements of roseate terns on the outer Cape. Through the reading of color bands, they confirmed that roseate terns from throughout the North Atlantic breeding range converge to stage at coastal southeastern Massachusetts habitats prior to fall migration (see NE section 2.3.1.7.3).

Trull *et al.* (1999) reported that of the 20 sites most used by staging roseate terns, human disturbance that caused flocks to disperse or flush and then resettle was documented at many locations. Disturbance took many forms, for example, by pedestrians, beach vehicles, aircraft, boats and dogs. Recently, the Coastal Waterbird Program of Massachusetts Audubon Society, the U.S. Geological Survey and other partners confirmed (through observations of color-banded birds), the importance of southeastern Massachusetts beaches to roseate terns from throughout the breeding range (Jedrey *et al.* 2010). Their studies also concurred with the observations of Trull *et al.* (1999) that human disturbance may be influencing what sites are used and how long terns use them.

Roseate terns are thought to roost on remote sand flats at the end of barrier beaches while in their winter quarters (Hays *et al.* 1999). These sites are similar to those used for staging around Cape Cod, and may be similarly subject to human disturbance (see NE section 2.3.1.7.5).

In addition to the effects of human disturbance on habitat availability for the roseate tern at this stage of their annual life cycle, the effect of climate change and sea level rise may bring profound changes to the coastline of outer Cape Cod. Rising sea levels that submerge tidal flats and bars, make beaches more narrow,

cut new inlets into barrier beach systems, and result in other changes to the geomorphology of the outer Cape and islands that will affect habitat availability and the degree to which humans and birds are in competition for space.

Climate change may affect roseate terns in another manner if coastal storms increase in either frequency or intensity. The survival rate for roseate terns in 1991 to 1992 was lower than that measured for all other years (Spendelow *et al.* 2008). This has been attributed to Hurricane Bob, which passed through the main staging area for the Northeast population (coastal southeastern Massachusetts) on August 21, 1991. Hurricane Bob appears to have significantly lowered the survival rate of most juveniles and many adults (Nisbet and Spendelow 1999, Spendelow *et al.* 2002, 2008, Lebreton *et al.* 2003).

#### **2.3.2.2 Factor B. Overutilization for commercial, recreational, scientific, or educational purposes:**

Overutilization for the millinery trade contributed to the historical (pre-1900) decline in roseate tern numbers (52 FR 42066). Under protective laws such as the Migratory Bird Treaty Act of 1918 (MBTA) and changing fashions, hunting of terns for their feathers was eliminated as a threat to the species within the breeding range. The 1987 final listing rule stated that a major cause for the declining number of roseate terns since the 1950's may be the trapping and netting of wintering terns for human consumption along the northeastern coast of South America (Nisbet 1984). Although some terns (predominantly common terns) are known to have been taken (or found dead) by Brazilian fishermen, it is not known if the practice is widespread enough to have population level effects.

Overutilization is not known to be a threat to the species on either the breeding or wintering grounds, but the ecology and threats to the species in the wintering grounds remain poorly studied (Hays *et al.* 1999). Based on current information, overutilization does not appear to be pertinent to the recovery objectives.

#### **2.3.2.3 Factor C. Disease or predation:**

Although disease is sometimes found to have contributed to the death of individual birds, it is infrequent and not known to cause population level effects in this species (see NE section 2.3.1.7.2). In contrast, predation at breeding colonies by a number of different avian and mammalian species is considered a constant threat requiring vigilance by island stewards so that mortality of eggs, chicks and adults can be minimized and colonies are not abandoned (RTRT, unpubl. data). Predation by one or more predatory species has been recorded at all roseate tern breeding sites that have been monitored in the period 1998 to 2009 (see NE section 2.3.1.7.1).

Overall, the most important effects on roseate tern populations have been the killing of adult terns by great horned owls and mink. Documented losses amount



to less than 1 percent of the warm water and cold water subregional populations per year, but predation on adult terns also causes major disruption of nesting and has led to repeated shifts away from affected sites, sometimes to sites that are less suitable. Predation by gulls (great black-backed, herring, and laughing gulls) and black-crowned night-herons has also been important at single sites, and predation by several other species has been recorded (for details, see NE section 2.3.1.7.1). The overall impact of predation on roseate terns in the region would undoubtedly have been much greater in the absence of intensive monitoring and predator control efforts that are conducted at all the major sites.

Regionwide, the incidence of predation appears to have been increasing, and some sites such as Great Gull Island, New York, and Bird Island, Massachusetts, which had little or no avian or mammalian predation in the 1970s and 1980s, were affected in some or most years in the 1990s and 2000s (see NE section 2.3.1.7.1). Predation on roseate and common terns by mink was formerly rare, but has occurred regularly in recent years at breeding sites in Maine and Nova Scotia, probably reflecting a general increase in mink numbers in that region (J. Drury, pers. comm.). Predation by great horned owls also appears to have increased in frequency and has occurred since the mid-1990s at sites such as Bird and Ram Islands, Massachusetts, where it was formerly unrecorded (see NE section 2.3.1.7.1). Predation by peregrine falcons at Petit Manan Island, Maine, has become regular since peregrines were re-established as nesting birds at nearby Mount Desert Island in the 1980s, and the number of peregrines visiting the island and the frequency of their visits has increased steadily (L. Welch, unpubl. data). J. Anderson and A. Reese (manuscript in review) suggest that the increasing frequency of predation by these species on islands in the Gulf of Maine represents re-occupation of their ancestral ranges, from which they were apparently excluded in the 19<sup>th</sup> and 20<sup>th</sup> centuries by human activities (trapping of mink, deforestation for great horned owls, persecution and toxic chemicals for peregrine falcons). If this interpretation is correct, the large numbers of seabirds that nested in the Gulf of Maine and their occupation of inshore islands in the 19<sup>th</sup> and 20<sup>th</sup> centuries may have resulted from abnormally predator-free conditions. It may prove increasingly difficult to maintain these inshore colony sites, where most of the cold water population of roseate terns now nest.

#### **2.3.2.4 Factor D. Inadequacy of existing regulatory mechanisms:**

This factor focuses on adequacy of non-ESA regulatory mechanisms if the protections of the ESA were to be removed. It should be noted that if threats identified under the other listing factors can be abated through means other than non-ESA regulatory mechanisms, the adequacy of these mechanisms would become moot.

Overall, the threats to the ROST-NE associated with the inadequacy of existing regulatory mechanisms remain unchanged since listing in 1988 and development of the 1998 recovery plan. Apart from the ESA, the roseate tern is protected at

the Federal level by both the MBTA. The MBTA bans the trade of roseate terns and their parts and protects the species from take. In general, however, the MBTA provides far less protection than the ESA. Loss of ESA protection through removal of regulatory requirements such as consultation under section 7 and protection from take under section 9 could result in increased disturbance, habitat degradation, and other threats to the species.

Compliance with the consultation requirements of section 7 by Federal agencies such as the U.S. Army Corps of Engineers and Bureau of Ocean Energy Management is considered good. As long as this ESA mechanism remains in place and is effectively implemented, it appears to provide adequate protections for the roseate tern. Once removed, however, there is no corollary protection under other Federal or State statutes or regulations.

Section 9 of the ESA prohibits the unlawful take of an endangered species, although this can be difficult to enforce without the regular presence of monitors and/or law enforcement officers. The remote locations (off-shore islands) where roseate terns nest make enforcement difficult. Although lethal take is considered to be very rare, other forms of take, such as harassment of nesting pairs by people and unleashed dogs, occur periodically. If these prohibitions were to be lifted, the primary regulatory mechanisms to control take would be the MBTA and State endangered species laws. As mentioned before, MBTA protections are not as stringent as the ESA's, so adverse effects on the Northeast roseate tern population could be anticipated.

The roseate tern is listed as endangered in the following northeastern States: Maine, New Hampshire, Massachusetts, Connecticut, and New York. The roseate tern is listed as State-historic in Rhode Island and does not currently nest there. The adequacy of State laws with regard to protection of the roseate tern and its habitat absent the ESA is of concern in some states. For example, in New Hampshire, RSA 212-A extends protection to individuals of a State-listed species, but these protections are less clear with regard to habitat, particularly if a potentially destructive activity does not require a State permit. Similarly, in Connecticut, General Statute 26-310 provides protection to species and habitat where there is an action of a State agency or there is State funding or a permit required. Like section 7 of the ESA, Connecticut GS 26-310 has an affirmative consultation requirement for State agencies whose actions may affect listed species, but there is no such requirement for the actions of private citizens. Although existing regulatory protections are sufficient to prevent intentional take of individuals in most states, disturbance of breeding colonies could occur more frequently in the absence of the ESA, and colony abandonment is possible at some sites. Overall, therefore, the MBTA and some State endangered species statutes are not sufficient to protect nesting, feeding, and staging habitats used by the species.

In Canada, several Federal and provincial laws provide protection to the roseate tern. The roseate tern is federally designated as Endangered (Whittam 1999, COSEWIC 2009) and is further protected by the Federal Migratory Birds Convention Act (1994) as a migratory non-game bird. Canada's Species at Risk Act (SARA) contains prohibitions (sec. 32) that make it an offence to kill, harm, harass, capture or take; possess, collect, buy, sell or trade an individual endangered roseate tern. It is also an offence to damage or destroy its residence, i.e., nest (sec. 33). Critical habitat for the roseate tern, which was identified in the Environment Canada's (2006) recovery strategy, encompasses The Brothers, Nova Scotia; Country Island, Nova Scotia; Sable Island, Nova Scotia; and Deuxième Îlet, Chenal Island, and Paquet Island (in the Magdalen Islands), Québec. Under Canadian migratory bird regulations, it is prohibited to capture, kill, injure, or harass adults, young, and eggs.

Roseate terns are also protected by the Nova Scotia Endangered Species Act, under which it is listed as Endangered. In Québec Province, the roseate tern is considered "likely to be designated as threatened or vulnerable" under the Loi sur les espèces menacées ou vulnérables du Québec (Québec's Act Respecting Threatened or Vulnerable Species). Overall, it appears that adequate regulatory mechanisms are in place to protect the roseate tern in this portion of its range, given that the regulations are effectively enforced.

The degree to which roosting, foraging, and resting habitat of northeastern roseate terns is afforded protection by law in Brazil, French Guyana, Guyana or other South American or Caribbean countries during the non-breeding period is not known.

#### **2.3.2.5 Factor E. Other natural or manmade factors affecting its continued existence:**

##### **Contaminants**

Two significant contaminant related events have occurred in Buzzards Bay, Massachusetts, the release of polychlorinated biphenyls (PCBs) into New Bedford Harbor and the Bouchard-120 oil spill. A brief discussion of these events and their effect on the roseate tern follow:

New Bedford Harbor is located in southeastern Massachusetts at the mouth of the Achusnet River on Buzzards Bay. The harbor and river are contaminated with high levels of hazardous substances and materials, including PCBs and metals, which were emitted by industry from the late 1940's to 1977 and have been transported by currents and biota into the nearby bay environment (New Bedford Harbor Trustee Council 2001). New Bedford Harbor was designated a Superfund site in 1983, making it eligible for Federal clean up action, which has been on-going to the present.

Roseate (and common) terns nesting in Buzzards Bay, Massachusetts, are known to have been exposed to dangerously high levels of PCBs that had accumulated in marine fish preyed on by the birds (Aquatec 1990; Nisbet *et al.* 1996; I. Nisbet, unpubl. data). Two common terns found dead on Bird Island in 1990 were reported to have lethal concentrations of PCBs in their brains and livers (Aquatec 1990, I. Nisbet, unpubl. data). To what extent PCB contamination has affected the roseate terns in Buzzards Bay is unclear, but since nearly 45 percent of all roseate terns in the Northeast population breed on islands in the Bay, pollution is a significant threat to the species.

Analyses of tern eggs collected over recent decades suggests that the remedial activities of the U.S. Environmental Protection Agency (EPA) and USACOE in cleaning up the site have been effective in reducing the levels of PCBs in the Bay ecosystem. In an ongoing study, tern eggs from Buzzards Bay colony sites were collected and pooled over four time periods (1972, 1994–1996, 1998–1999 and 2005); common tern eggs were available for all four periods and roseate tern eggs from the last three. Extracts from the eggs were analyzed for a variety of persistent organic pollutants at the EPA laboratory at Narragansett, Rhode Island. Results indicate that the accumulation of PCBs has declined progressively over time toward non-toxic levels. However, concentrations of an “emerging contaminant”, the flame retardant polybrominated diphenyl ethers (PBDEs) have increased progressively over time (USACOE, C. Mostello, and I. Nisbet, unpubl. data).

Many ecological restoration projects have been funded through the Natural Resource Damage Assessment Restoration (NRDAR) settlement for this site pursuant to the Comprehensive Environmental Response, Compensation and Liability Act of 1980, as amended (42 U.S.C. 9601 et seq.). Since 1999 the New Bedford Harbor Trustee Council has funded the Buzzards Bay Tern Restoration Project, coordinated through MDFW, to “restore” and otherwise promote the conservation of terns in the Bay. This support has allowed MDFW to support a full time biologist position as well as several seasonal biologists to monitor, manage and conduct habitat enhancement work at the three Buzzard’s Bay colonies (Bird, Ram, and Penikese Islands). In addition, New Bedford Harbor NRDAR settlement funds were used to restore about 0.4 acres of degraded habitat on Ram Island in 2010. Both roseate and common terns nested in the restored habitat in 2010 (MDFW, unpubl. data).

#### Bouchard 120 Oil Spill

In April 2003, a barge being transported by the Bouchard Oil transportation company struck a reef and spilled an estimated 98,000 gallons of heating oil into Buzzards Bay, Massachusetts. Ram Island, Mattapoissett, one of the three largest nesting colonies of roseate terns, was moderately oiled and small quantities of oil also appeared at Bird Island and Penikese Islands. Only two terns (of unrecorded species) are known to have been killed by oiling, but many roseate terns examined

at Bird Island during May had small spots and streaks of oil on their plumage (MDFW, unpubl. data). These would have been exposed to oil by ingestion following preening, and potentially also by ingesting contaminated prey.

The oiling of Ram Island occurred just prior to roseate terns returning to nest at the site, and to prevent the birds from coming into direct contact with oil at Ram, the birds were actively hazed for a period of several weeks in April to May 2003, until the island shoreline was cleaned. About 500 pairs of roseate terns were displaced to Bird Island and 250 pairs to Penikese Island in 2003, where they nested for the first time in recent decades (MDFW and USFWS, unpubl. data). Because early breeding by roseate terns is consistently more productive than late breeding (Burger *et al.* 1996), the delay in nesting by pairs that eventually settled at Ram Island and nested, as well as the displacement of other pairs to Bird and Penikese Islands, came at a cost to productivity (MDFW and USFWS, unpubl. data). Estimation of the net effect of oiling and hazing is difficult, however, because productivity at Bird Island in 2003 was higher and that at Penikese Island was lower than that at Ram Island, and productivity at all three sites in 2003 was high relative to the long-term averages (Table 5). The net effects of oiling and hazing on chick production are currently being assessed under the NRDAR process, but details are confidential until the NRDAR is completed.

The barge involved in the 2003 spill was part of a large-scale transport of refined oil products from refineries in New Jersey and Delaware to distributors and consumers in New England (Table 17). There is regular traffic of oil-laden barges through Long Island Sound and Buzzards Bay, which takes them past the largest roseate tern colonies in the region: Falkner, Great Gull, Penikese, Ram, and Bird Islands (see Figure 1). All these sites have been under threat of pollution from barge accidents for many decades. Following the Bouchard accident in 2003, the Commonwealth of Massachusetts imposed stricter regulation of barge traffic, including a requirement for tug escorts through Buzzards Bay, but the threat will not be fully alleviated until the final replacement of single-hulled with double-hulled barges at some time in the future.

**Table 17.** Petroleum Product shipped through Selected Ports within the Breeding Range of the Roseate Tern in the United States North Atlantic in 2008 (USACOE 2010).

<u>Port</u>	<u>Petroleum Product (gallons)</u>
New York, New York	138.77 billion
New London, Connecticut	71.3 million
New Bedford, Massachusetts	19.6 million
Boston, Massachusetts	39.8 billion
Portsmouth, New Hampshire	4.36 billion
Searsport, Maine	3.43 billion
Bucksport, Maine	26.3 million

The restriction of the breeding population within a narrow latitudinal range increases the species vulnerability to the adverse effects of an oil spill. As Table 17 indicates, even though there are no near shore or off shore oil wells on the Atlantic outer continental shelf within the breeding range of the roseate tern, an enormous quantity of petroleum product (nearly 200 billion gallons in 2008) is shipped through northeast ports on an annual basis (USACOE 2010).

### **Wind turbines**

Wind turbine generators have emerged as a potential threat to roseate terns since the 1998 revised recovery plan. The primary threat caused by turbines is the potential for mortality due to striking turbines rotors, towers or guy wires (Everaert and Stienen 2006).

Several wind turbines within the breeding range of the roseate tern in the western North Atlantic have been constructed, and several more are either proposed or planned (USFWS, unpubl. data). Five small-scale wind turbine generators have been constructed on Sable Island, Nova Scotia, where in recent years 4 to 5 roseate tern pairs have nested (A. Boyne, pers. comm.). No roseate tern mortalities have been documented, as the birds nest on the far side of the island from the turbines, but several common terns that nest in the immediate vicinity have been killed (A. Boyne, pers. comm.). Seventeen commercial-scale turbines were constructed on Pubnico Peninsula in 2004 and 2005 and are about 4 km (2.5 mi) from the roseate tern colony at The Brothers Island, Nova Scotia. Avian and bat mortality studies were conducted during 2006 and 2007, and no terns were found (Matkovich 2007). Lastly, a single commercial-scale turbine was constructed at the Massachusetts Maritime Academy in Bourne, Massachusetts, adjacent to Buzzard's Bay and the Cape Cod Canal. Large numbers of roseate terns and common tern feed in waters adjacent to the site and routinely cross the campus to move from the bay to the canal. Vlietstra (2008) conducted seasonal, post-construction monitoring at the site during 2006 and 2007, and reported three bird mortalities but none were terns.

In late 2008, the USFWS completed consultation with the Minerals Management Service (MMS) on a proposal by Cape Wind Associates to construct 130 wind turbine generators approximately 5 miles off the coast of Cape Cod, Massachusetts, in Nantucket Sound (USFWS 2008), but the project has yet to enter the construction phase. The avian risk assessment prepared for this large project estimates that small numbers of roseate terns will be killed annually from collisions with the turbines, but the level of take is not expected to have population level effects on the species (USFWS 2008).

There are many other proposed or conceptual off-shore wind turbine projects planned within the western North Atlantic breeding range for the species. Deepwater, Inc., has proposed an 8-turbine project in waters three miles (5 km) southeast of Block Island, Rhode Island, and has expressed interest in developing

a much larger turbine array further off the Rhode Island coast in the future. Offshore Megawatt has similarly expressed interest in developing a commercial scale, wind turbine project in the waters south of Noman's Island and Martha's Vineyard, Massachusetts.

The USFWS has also provided technical assistance and preliminary comments regarding proposals (in various stages of development) for one to two wind turbine generators to the U.S. Department of Agriculture in Maine, the U.S. Coast Guard and National Guard Training Center in New Jersey, and the National Aeronautics and Space Administration in Virginia. The USFWS has also reviewed lease proposals for single meteorological towers in seven potential wind turbine generator lease blocks located 8 to 17 mi (13 to 27 km) off the coast of Delaware and New Jersey, as well as several potential off-shore wind power demonstration sites in Maine. Although analysis of the best available information indicates that risk from the Cape Wind project to roseate terns is low (USFWS 2008), the prospect of multiple large wind turbine generator projects along potential migration routes poses greater concerns. Studies to determine the most effective methods to assess wind turbine generator risks to roseate terns (and other listed and candidate bird species) on the Outer Continental Shelf are currently in planning stages under the auspices of the Bureau of Ocean Energy Management (formerly MMS).

Risk from wind turbine generators sited between nesting islands and feeding shoals or in the vicinity of beaches used during the fall staging period pose the greatest risk and require further assessment. Impacts may vary with the specific size, number, and configuration of proposed wind turbine arrays and site-specific factors such as juxtaposition of nesting and foraging habitats and weather patterns. Wind turbine generators pose a threat to roseate terns in the foreseeable future, but the magnitude of this threat cannot be assessed without better information about annual and within the breeding season movements. Information needs include migration routes and altitude, flight patterns associated with breeding adults and post-fledged young of the year foraging at nearby sites, and avoidance rates under varying weather conditions.

### **Population trend uncertainties**

A general problem in formulating recovery plans for the Northeast population of the roseate tern is that the factors responsible for its precarious conservation status remain elusive. Twenty-five years of intensive study under the CRTMP have generated detailed numerical information about many aspects of its demography and about the structure and dynamics of the regional "metapopulation" (see NE section 2.3.2.1). Nevertheless, the most recent attempt to construct a life-table model for the population found that the measured demographic parameters were inconsistent with the observed trends in the overall population, including the steady increase from 1988 to 2000 and the abrupt change to a decrease from 2000 to 2009 (Nisbet and Ratcliffe 2008). Despite an apparent regionwide decline in

productivity since 2000 (see NE section 2.3.1.2.3), the overall mean productivity of the population has remained high relative to that of other terns with thriving populations. The factors that control these population trends and ultimately are responsible for the small regional population and restricted range are presumably to be found in patterns of mortality (Nisbet and Ratcliffe 2008). However, most mortality occurs away from the breeding area, on migration and/or in winter. Very little is known about the distribution and ecology of roseate terns from this population at these seasons, and there is no information of any kind about causes of death. Much more information is needed about factors limiting survival in the winter quarters to allow formulation of effective conservation measures.

## **2.4 Synthesis**

The following summarizes the most important new biological information and changes to the threats for the Northeast Population of the roseate tern since the recovery plan was completed in 1998 (USFWS 1998b).

The most urgent contemporary threat facing northeastern roseate tern habitat is the erosion and degradation of nesting habitat on Bird and Ram Islands in Buzzard's Bay, Massachusetts, and at several other colonies throughout the species' breeding range. Together, Bird and Ram Islands support more than 40 per cent of the breeding pairs in the Northeast population (about 1300 to 1400 pairs). Gardners Island-Cartwright Point on Long Island, New York, supported about 150 to 250 pairs in the early 2000s, but currently few pairs nest there and the island showed evidence of being over washed when visited in 2009. The Brothers Island in Nova Scotia is similarly threatened with erosion and while comparatively few pairs nested there in the past decade (30 to 90), it has been the largest roseate tern nesting colony in Canada; its loss would mean a further curtailment of the species' range. Sea level rise will exacerbate erosion of nesting islands and climate change that results in more severe or more frequent coastal storms will affect these habitats and could result in reduced survival of adults, eggs, chicks and fledglings.

Ironically, a project to address severe erosion by stabilizing the shoreline of Falkner Island, Connecticut had untoward effects on the roseate terns breeding there. The nesting habitats of three of six sub-colonies on the island were directly modified by large stone revetment, chicks became entrapped in the labyrinth spaces of the placed rock, and a small number of adults and fledglings are thought to have died within these spaces. More importantly, in preventing erosion of the island's east shore, the revetment has removed the source of sand that maintained the spit at the north end of the island. As this spit further erodes, the nesting habitat that it provides to roseate terns has become degraded in size and quality.

Changes in coastal geomorphology, whether "natural" or the result of man-induced climate change, can have a profound effect on the creation and suitability of roseate tern nesting islands. For example, the emergence of "Minimoy Island" a small island in Massachusetts to the west of North Monomoy provided new nesting habitat for roseate and other species of terns between 2003 and 2008, but has since become eroded and except for a single pair in 2010, has been abandoned by roseate terns. In another dramatic change, sand movement has resulted in the extension of South Beach Chatman, Massachusetts toward South Monomoy Island. This has allowed more



frequent and numerous terrestrial predators, such as coyotes and opossum, to access roseate tern nesting areas that were previously, only infrequently visited by mammalian predators.

An additional serious contemporary threat to the survival and recovery of the Northeast Population is the downward trend in the number of adult breeding pairs. Nearly 1000 fewer adult breeding pairs were recorded nesting among colonies in the Northeast in 2009, a 25 percent decline since the year 2000. The cause or causes for this population reduction are not known, and it has direct bearing on the ability to achieve the numerical and distributional recovery objectives for the species.

While competition for nesting space and predation by herring and black-backed gulls remains a concern at certain colonies, regional populations of these large gulls are either stable or decreasing. However, populations of laughing gulls are increasing and this species is now emerging as an important new threat to roseate terns at some colonies, particularly in Maine.

Human disturbance of roseate terns during the post-breeding staging period is a new threat that warrants further study. Roseate and other terns exhibit large within-season and between-year differences in the coastal habitats they utilize during the period from late July until they depart in mid-September for wintering areas in South America. To what extent human disturbance may be influencing the utilization of fall staging habitat is not known. Similarly, it is not known whether flocks of terns that are temporarily displaced from one site to another due to disturbance, incur an energetic cost that will reduce their fitness for migration and successful over wintering. However, given that virtually the entire Northeast roseate tern population may be present in southeastern coastal Massachusetts during the staging period, these potential threats merit evaluation.

Predation of adults, eggs, and chicks during the breeding season when the birds are concentrated within dense colonies remains a threat, even at those islands that are more remote from the mainland. The once-endangered peregrine falcon has recovered and this avian predator now regularly visits offshore colonies in Maine, and occasionally elsewhere, causing panic flights, temporary colony abandonment, and mortality. A suite of other avian and mammalian predators (particularly mink, raccoons, and great horned owls) appear unpredictably at colonies, and tern island biologists must be vigilant and responsive to minimize egg, chick, and adult mortality as well as other adverse effects that may result in colony abandonment.

The 2003 Bouchard 120 oil spill in Buzzard's Bay, Massachusetts, demonstrated the vulnerability that a large proportion of the roseate tern breeding population in the Northeast could be affected by a single accidental spill. This relatively small spill (less than 100,000 gallons of heating oil) occurred during the onset of the 2003 breeding season and could have been devastating. Statistics indicate that nearly 200 billion gallons of petroleum product (primarily home heating oil) is shipped annually through ports in the northeastern United States. The vulnerability to oil spills and other releases of contaminants remains a significant threat to the roseate tern simply because such a large proportion of the nesting population is concentrated within a narrow latitudinal range. The advance of off-shore wind power development has introduced a new threat not foreseen in the 1998 recovery plan. European studies have demonstrated that common and sandwich (*Thalasseus sandvicensis*) terns are vulnerable to

collision mortality from striking turbine towers and blades when turbines were constructed near a breeding colony. However, it is not known how vulnerable roseate terns are to collision mortality from proposed off-shore wind power developments in the United States. Proposed project locations to date (*e.g.*, Cape Wind in Nantucket Sound, Off-shore Mega Watt south of Martha's Vineyard, and others) may periodically intercept roseate terns during twice-annual migrations or during commuting flights between resting and foraging areas during pre- and post-breeding periods. Avoidance rates (the probability of a roseate tern taking avoiding action when encountering a turbine; see, for example, Chamberlain *et al.* 2006) are not known.

## **THREATENED CARIBBEAN POPULATION**

### **Section 2.0 REVIEW ANALYSIS**

#### **2.1 Application of the 1996 Distinct Population Segment (DPS) Policy**

**2.1.1 Is the species under review a vertebrate?** Yes, therefore the DPS policy is applicable.

**2.1.2** See section 2.1.1 under Northeast Population for the DPS discussion.

#### **2.2 Recovery Criteria**

**2.2.1 Does the species have a final, approved recovery plan containing objective, measurable criteria?** Although the Caribbean roseate tern has a recovery plan, delisting recovery criteria could not be set when the recovery plan was finalized and approved, because critical demographic information for a reliable population assessment was lacking.

##### **2.2.2 Adequacy of recovery criteria:**

**2.2.2.1 Do the recovery criteria reflect the best available and most up-to date information on the biology of the species and its habitat?** No. Knowledge on the biology, feeding ecology, and habitat use of the Caribbean roseate tern has expanded, particularly for populations in Florida, Puerto Rico, and the U.S. Virgin Islands.

**2.2.2.2 Are all of the 5 listing factors relevant to the species addressed in the recovery criteria (and is there new information to consider regarding existing or new threats)?** No. The plan did not address the five-listing factor analysis in the recovery criteria.

**2.2.2.3 List the recovery criteria as they appear in the recovery plan, and discuss how each criterion has or has not been met, citing information:**

The Caribbean roseate tern may be considered for delisting when:

1. Conservation programs to maintain, protect, and enhance populations of this species have been implemented in coordination with countries in the Caribbean where roseate tern populations occur.
2. Populations of roseate terns in the Caribbean remain stable (*i.e.*, without significant decrease in number of breeding birds) or increasing for at least 5 consecutive years.

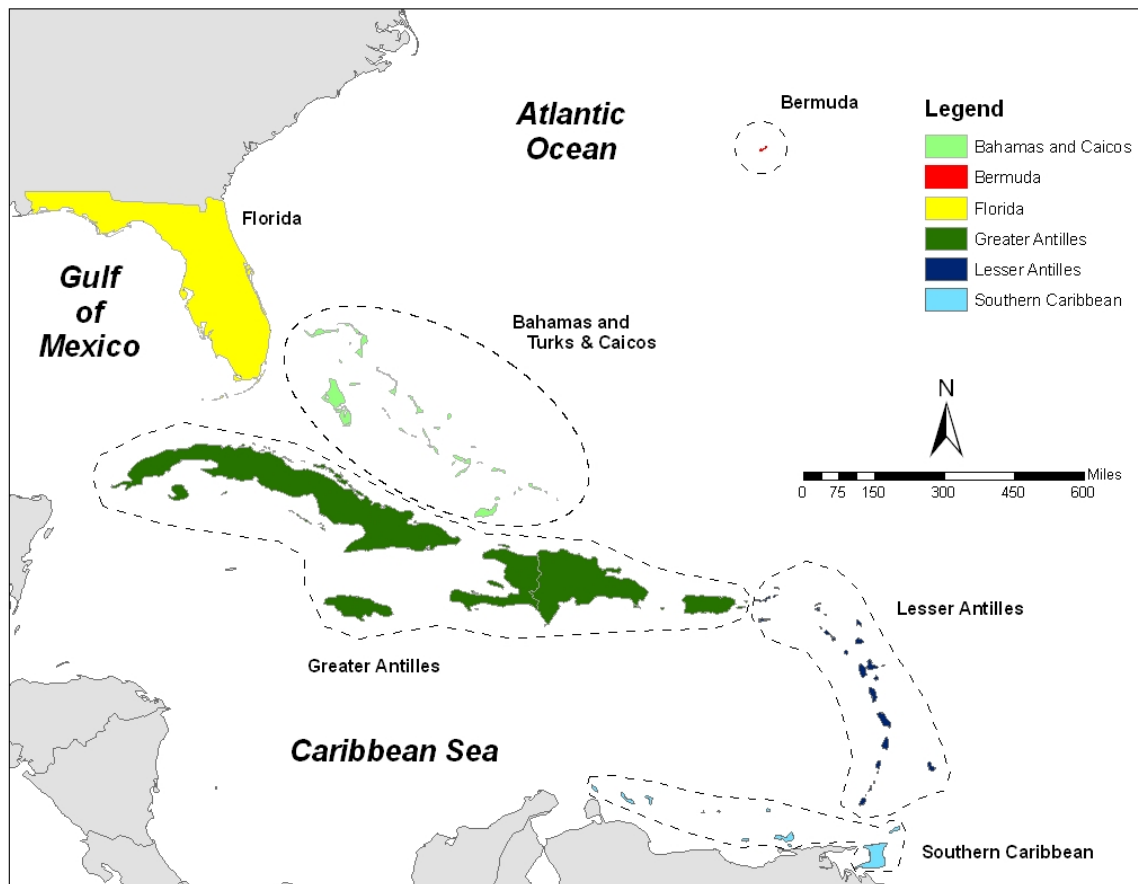
Criterion 1 has not been met. Caribbean-wide conservation programs for this species have not been coordinated, established, or implemented.

Criterion 2 has been partially met. Since 2005, the population of roseate terns in southwestern Puerto Rico has remained stable at close to 1,000 pairs per year. However, other colonies in the Caribbean have not been closely monitored to determine number of breeding pairs per year, to determine population stability, or to ascertain if individuals from other populations have shifted to nesting in southwestern Puerto Rico.

## **2.3 Updated Information and Current Species Status**

The Caribbean roseate tern population appears to constitute cells of a metapopulation (Bradley and Norton 2009) that include Bermuda (formerly); Bahamas Islands; the Florida Keys; Turks and Caicos Islands; Greater Antilles (Cuba, Hispaniola [Dominican Republic and Haiti], Jamaica, and Puerto Rico); United States and British Virgin Islands; Lesser Antilles (Anguilla, Antigua, Barbuda, Guadeloupe Archipelago, Martinique, St. Martin, St. Bartholomew, St. Kitts and Nevis, St. Lucia, St. Vincent, Grenadines and Grenada); Trinidad and Tobago; and islands in the southern Caribbean (Aruba, Bonaire, Curaçao, and formerly islands off Venezuela) (Bradley and Norton 2009). The continental United States (mainland Florida, Georgia, South Carolina, and North Carolina) are not considered within the Caribbean region, but the occurrence of this species in the southeastern United States is included in this review (Figure C1).

**Figure C1.** Range of the Caribbean Roseate Tern Population.



### 2.3.1 Biology and habitat:

#### 2.3.1.1 New information on the species' biology and life history:

The Caribbean roseate tern is a primarily white, slender-winged, long-tailed, typical capped tern (family Laridae, subfamily Sterninae). Its overall length is about 40 centimeters [cm (16 inches (in))], including tail streamers 15 to 25 cm (6 to 10 in) in length (USFWS 1993). It has a black crown, pale-grey upper surface, and immaculate white underparts. Both the upper and under surfaces are paler than in the very similar common tern. In non-breeding plumage, both common and roseate terns have a dark carpal bar over the bend of the wing, although it is slightly lighter in roseate terns (USFWS 1993). The three or four outer primaries (wing feathers) of roseate terns are frosted with silver-grey and edged with black (USFWS 1993). The long tail streamers are pure white, whereas those of common terns (*Sterna hirundo*) are grayish and have a black outer margin (Harrison 1983). Early in the breeding season, there is an evanescent pink or peach bloom on the underparts, visible in some lights (Harrison 1983). The basal three-quarters of the bill in Caribbean roseate terns gradually becomes reddish

orange during the breeding season (Shealer and Saliva 1992), starting as early as when the first eggs are laid. By the time that the terns lay their second egg, all breeding birds have the basal three-quarters of their bills orange. Roseate terns in eastern North America, however, have more black on their bill than Caribbean birds (Saliva 2000), and breeding birds sometimes show little or no orange on their bills during the entire breeding season. In the Caribbean, however, roseate terns with entirely black bills have not been recorded tending clutches of more than one egg, or feeding chicks (J.E. Saliva, USFWS, pers. obs.). Male and female roseate terns are essentially identical in size and color. Roseate terns usually lay one or two eggs, and chicks fledge after 22 to 29 days of age (USFWS 1993).

Since the publication of the Caribbean Recovery Plan in 1993 (USFWS 1993), the status, numbers and population trends of roseate terns in the Caribbean have been reviewed in two major publications, edited by Schreiber and Lee (2000) and Bradley and Norton (2009). See section 2.3.1.2, below, for a summary of data for Florida, Puerto Rico and the U.S. Virgin Islands, and Appendix A, section A.1 for a summary of data from other parts of the Caribbean. During this period, roseate terns have been monitored in Florida by R. Zambrano, in Puerto Rico by J.E. Saliva, and in the U.S. Virgin Islands by J. Pierce (see section 2.3.1.2, below). The most extensive data on the biology of the species in the Caribbean are derived from studies by D. Shealer in southwest Puerto Rico between 1991 and 1994, which were summarized in three reports to the USFWS (Shealer 1991, 1993, 1994), in a Ph.D. thesis (Shealer 1995) and in a series of publications (Shealer 1996, 1998, Shealer and Burger 1992, 1993, 1995, Shealer and Saliva 1992, Shealer *et al.* 2005a). Additional biological studies were reported in two unpublished reports to the USFWS by H. Douglas (2000, 2001).

### **2.3.1.2 Abundance, population trends, demographic features, and/or demographic trends:**

The available data on roseate tern populations throughout the Caribbean (excluding Florida and the Puerto Rico Bank) indicate that most colonies are relatively small, decreasing in size, or abandoning historic sites (Table C1 and C2). The stronghold of the Caribbean population is in the Puerto Rico Bank; which includes Puerto Rico and the Virgin Islands (Saliva 2000; Schreiber and Pierce 2009) (Figures C2 and C3). Apart from the Puerto Rico Bank, colonies larger than 500 pairs appear to only occur in the Bahamas Islands. For most colonies, only an estimated number of pairs have been reported, due to the lack of human resources to conduct nest counts and/or the inaccessibility of some colony locations. Therefore, the actual number of breeding pairs is not known for most islands. Combining several sets of estimates, Bradley and Norton (2009) estimated that the total number of nesting roseate terns for the Caribbean region in 2007 was in the range 3,571 to 7,095 pairs, with a central estimate of 5,412 pairs. This compares with earlier estimates of 4,000 to 6,000 pairs (Saliva 2000 *in* Schreiber and Lee 2000) and 2,500 to 4,000 pairs (van Halewyn and Norton

1984). Given the incomplete and non-quantitative data for many sites, and the tendency for the species to move from site to site on a short time-scale (see below), there is no clear evidence that the regional population has either decreased or increased over the 30-year time-scale of these surveys. Because of the marked year-to-year fluctuations in numbers at most sites, Nisbet and Ratcliffe (2008) suggested that many birds may not breed every year; if so, the total size of the population would be substantially larger than 10,000 birds.

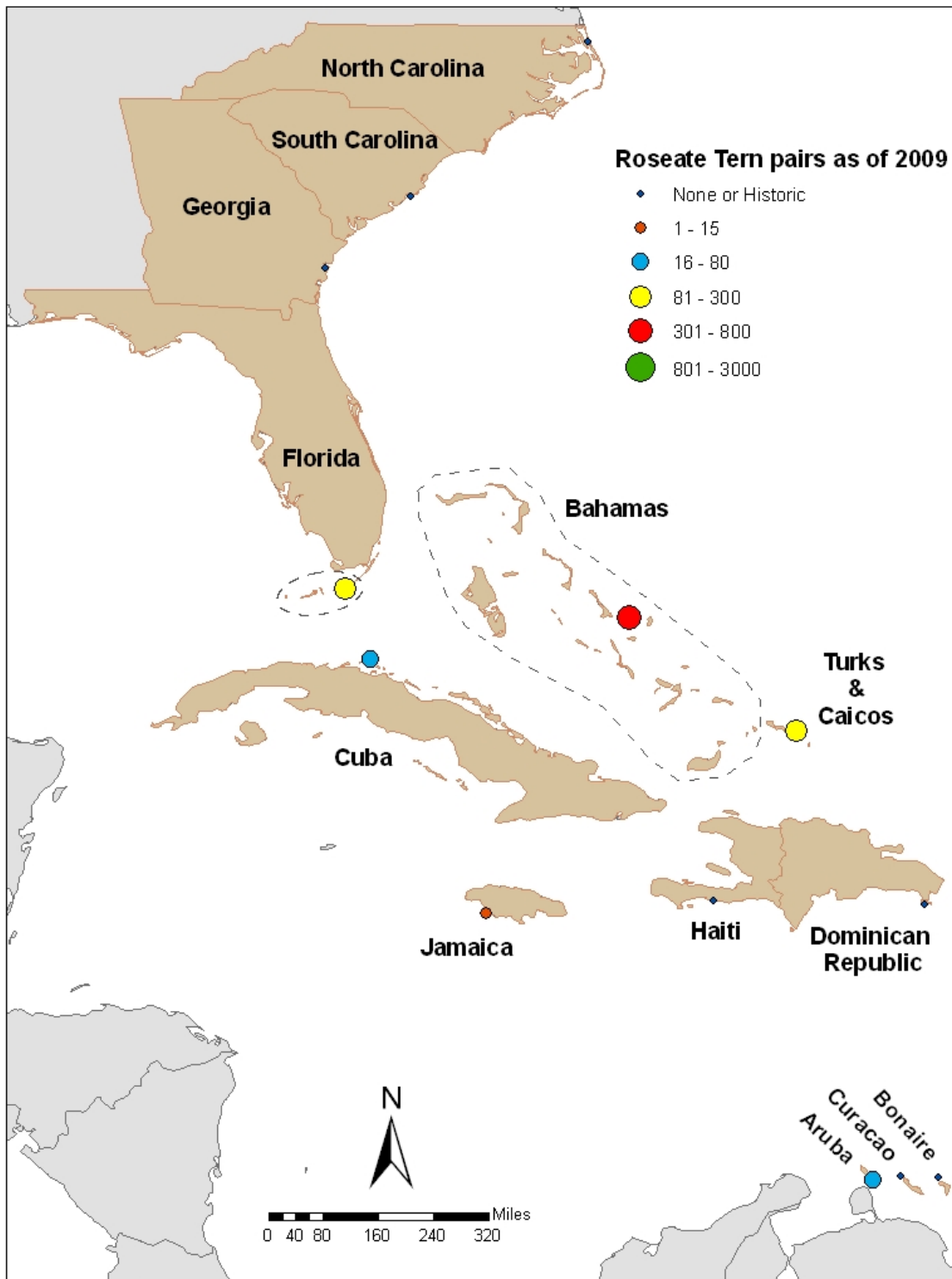
**Table C1.** Estimated maximum number of roseate tern breeding pairs in the Caribbean and current threats to the populations (Bradley and Norton 2009) and Florida (Zambrano 2010).

Location	# of pairs	Major Threats
Virgin Islands	2,500	Egg collection, human disturbance
Puerto Rico	1,400	Human disturbance, native predators
Florida	261	Human disturbance, development
Bermuda Islands	Extirpated	
Bahamas Islands	800	Human disturbance, egg collection
Turks & Caicos	200	Human disturbance, egg collection, native predators
Cuba	50	Development, egg collection, non-native predators
Jamaica	5	Erosion of nesting islands, egg collection
Dominican Republic	Extirpated	
Haiti	Extirpated	
Anguilla	210	Human disturbance, development, non-native predators
French Antilles	50	Development, human disturbance, egg collection
Netherlands Antilles	Unknown	
St. Kitts & Nevis	6	Non-native predators
Antigua, Barbuda, & Redonda	Unknown	
Martinique	450	Human disturbance, egg collection, native predators
St. Lucia	75	Human disturbance, egg collection
St. Vincent, Grenadines, & Grenada	15	Egg collection
Trinidad	25	Egg collection
Tobago	111	Egg collection
Aruba, Curacao, & Bonaire	52	Oil pollution, human disturbance, egg collection

**Table C2.** Roseate tern nest censuses conducted in the U.S. Virgin Islands (USVI), 1987 - 2001.

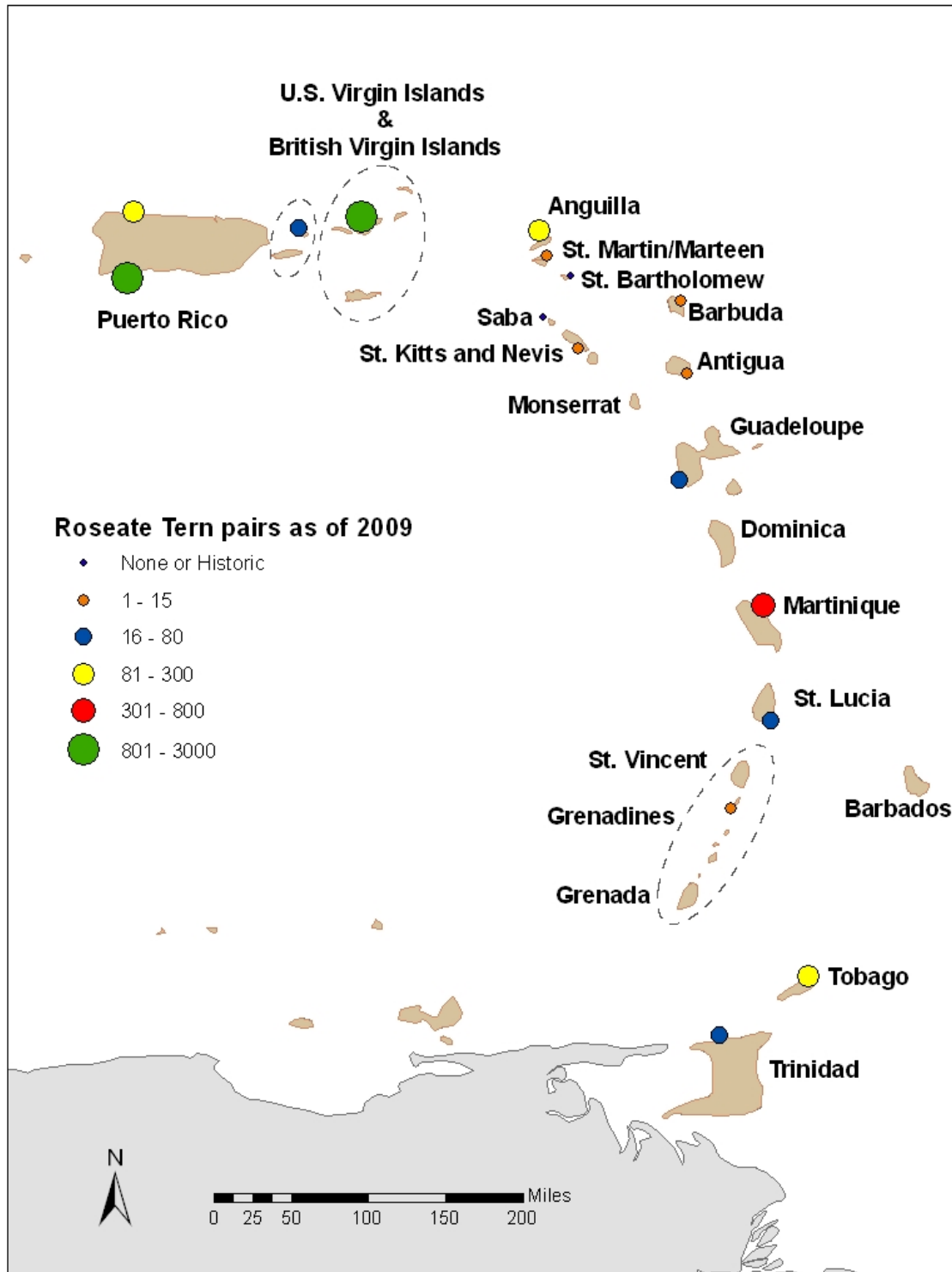
Location	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001*
Kalkun	140	0	6	289	250	472	50	333	74	0	334	257	120	946+	0
Shark	193	0	458	0	877	9	269	0	248	194	0	65	75	314+	0
Pelican	135	339	0	0	0	0	278	0	384	0	0	0	402	0	0
Flat	185	0	0	0	0	0	0	23	0	0	75	0	41	184	0
Booby Rock	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Flanagan Rocks	68	46	0	0	0	74	0	0	0	0	0	0	0	0	0
Saba	0	312	349	134	579	0	0	144	80	350	20	100	0	0	~150
Flanagan	0	672	0	0	0	1019	0	0	0	229	0	0	0	218+	0
Cricket	0	145	0	0	0	0	0	0	21	0	0	0	0	0	0
Dog	0	14	273	0	0	0	0	0	0	0	0	0	0	0	0
Rata	0	0	0	50	0	256	0	0	7	0	198	0	0	0	0
Two Brothers	0	0	0	8	0	0	0	0	0	0	0	50*	0	0	0
LeDuck	0	0	0	0	734	0	0	529	0	0	783	822	~25	0	1200+
Carval Rock	0	0	0	0	0	77	0	0	0	0	0	0	0	0	0
Congo	0	0	0	0	0	0	0	0	0	0	0	273	0	0	0
Little Flat	0	0	0	0	0	30	0	0	0	0	0	30	0	0	0
Ramgoat	0	0	0	0	0	0	0	0	0	0	848	0	0	0	0
Total USVI	728	1528	1081	481	2190	1937	572	1029	814	773	2258	1547	763	1662	1350

**Figure C2.** Map of Roseate Tern Breeding Populations in the Western Caribbean.





**Figure C3.** Map of Roseate Tern Breeding Populations in the Eastern Caribbean.



Updated information on the distribution and numbers of the roseate tern in Florida, Puerto Rico, and the U.S. Islands is provided in the following sections. Information on the distribution and numbers of the roseate tern in the remainder of the Caribbean is summarized in Appendix A, section A.1.

#### **2.3.1.2.1 Distribution and numbers in Florida**

The Florida Fish and Wildlife Conservation Commission (FFWCC) and the USFWS designated the roseate tern in Florida as a threatened species (USFWS 1993; R. Zambrano, pers. comm. 2010). Until the early 1970s, the Dry Tortugas were the primary roseate tern breeding area in Florida (Robertson 1978, p. 39). Predators and nesting failure due to storm surges from tropical storms probably led to the gradual shifting of this colony to the Florida Keys, with much of the activity occurring on spoil or otherwise denuded islands in the Key West area (Robertson 1978, p. 40). Zambrano *et al.* (2000) identified 12 roseate tern breeding areas in the mainland Florida Keys from 1974 to 1998: Grassy Key, Indian Key (fill), Key Haven, Key West (Tank Island), Key West Harbour, Key West (Truman Annex roof), Lower Matecumbe Bay, Marathon (condominium roof), Marathon (Casa Cayo condominium), Marathon (Vaca Rock), Missouri Key, and Pelican Shoal.

A high percentage of the Florida Keys roseate tern population typically occupies a single nesting location in any given year (Zambrano *et al.* 2000). Throughout the period of 1960 to 1970 the primary nesting location was in the Dry Tortugas. The colony then gradually diminished in size in the late 1970s. Concurrently, nesting shifted to a small number of sites in the middle and lower Florida Keys (Smith 1996). By about 1990, virtually no nesting occurred at the historical Tortugas sites, and the main nesting colony established itself at Pelican Shoal, a small sand and coral rubble island in the Lower Keys (Zambrano *et al.* 2000). Up to three additional active colonies (besides Pelican Shoal) have been recorded in any 1 year since the shift to the main Keys. However, these colonies all tended to be small and unstable. By 1999, only Pelican Shoal and a roof colony on the Marathon Government Center building remained (Zambrano *et al.* 2000). More disturbing than the reduction in the number of colonies is the decline in the overall population.

Between 2000 and 2005, the entire Florida breeding population was restricted to two sites: a ground and roof colony (R. Zambrano, pers. comm. 2010). Since the 1970's, the roseate tern has been declining in Florida. By 1997, only the Marathon Government Center (a roof colony) and Pelican Shoal (a ground colony) remained (R. Zambrano, pers. comm. 2010). However, the 2005 hurricanes set in motion a steep decline in the nesting population and overall breeding productivity (R. Zambrano, pers. comm. 2010). Tropical Storm Arlene, and Hurricanes Dennis, Rita, and

Wilma were responsible for eroding and completely removing the ground colony site in 2005. After 2005, Pelican Shoal was submerged under 1 to 2 feet of water and no longer available as a nesting site. The disappearance of the ground colony caused the population to fragment into four smaller, and in some cases, less suitable sites (R. Zambrano, pers. comm. 2010). In the spring of 2006, FFWCC biologists attempted to provide the birds displaced from Pelican Shoal with an alternative nesting area (R. Zambrano, pers. comm. 2010). In cooperation with the National Park Service (NPS), biologists placed plastic tern decoys along with a sound system and speakers broadcasting tern calls on Long Key at Dry Tortugas National Park in April, before the terns arrived for nesting. These techniques, known as “social attraction”, have been used around the world to attract colonially nesting birds to nesting areas and to restore seabird colonies. In 2007 and 2008, 39 and 47 roseate tern pairs, respectively, nested at Long Key. FFWCC and NPS will continue using social attraction methods at Dry Tortugas National Park until it is determined that roseate terns have permanently established themselves there (R. Zambrano, pers. comm. 2010).

From the mid-1980s to 2005, roseate terns were rarely observed in Key West NWR and nesting habitat appeared to be non-existent (T.J. Wilmers, USFWS, pers. comm. 2010). With the passage of Hurricane Wilma in 2005, a small, barren island (Wilma Key) of sand, shell, and broken coral developed in Key West NWR (Wilmers and Lyons 2008). Roseate terns were attracted to the site, but were met with an onslaught of human and human-related harassment caused by dogs, para-surfers, and the hitting of golf balls. Harassment may have occurred every day and probably many nights as well. Between June 2006 and May 2009, up to 71 roseate terns were observed during surveys of the island. In 2006, three roseate tern pairs nested at this site, though unsuccessfully (T.J. Wilmers, pers. comm. 2010). The birds nested there in both 2006 and 2007. By the latter year, erosion had reduced the island by about a third. Nesting did not occur in 2008 (the island by then had narrowed considerably) or 2009, by which time most was flooded at high tide (T.J. Wilmers, pers. comm. 2010).

Subsequent to 2005, breeding locations were largely redistributed. Some of the erosion of suitable nesting sites due to the 2005 storms may have been partly ameliorated by depositions elsewhere, although the net effect was a decline in habitat and overall nest abundance. The Dry Tortugas National Park, a former roseate tern breeding site, was once again used for nesting, as was the new sand bar, Wilma Key) (Wilmers and Lyons 2008; R. Zambrano, pers. comm. 2010). In addition, a new roof colony was discovered at Key Colony in 2007 (R. Zambrano, pers. comm. 2010). Roseate terns were also found nesting among least terns (*Sternula antillarum*) on a condominium rooftop in Marathon Key in 2008, where FFWCC biologists located and monitored one roseate tern nest (R.

Zambrano, pers. comm. 2010). Roseate terns did not successfully nest on Wilma Key in 2007 (they first nested on this island in 2006), presumably due to high levels of human disturbance. Roseate terns were again observed on Wilma Key in 2008. USFWS staff posted the area “No trespassing” and erected symbolic fencing on the island, and buoys around it in order to prevent disturbance and encourage nesting (Wilmers and Lyons 2008). However, roseate terns did not nest at this site in 2008, and in 2009, 27 nests were reported on Bush Key, Dry Tortugas (R. Zambrano, pers. comm. 2010).

Some nesting resumed at the Marathon Government Center rooftop after 2005 (R. Zambrano, pers. comm. 2010). Based on limited observations, birds in this colony appeared to be abandoning and re-nesting at various intervals, so it was difficult to estimate colony size based on nest counts alone; it was unclear how many pairs were successful in their nesting attempts for this same reason. Subjectively, however, it was clear that the Marathon Government Center colony was larger in 2008 than in 2007 (R. Zambrano, pers. comm. 2010). A sample of chicks was captured, banded, and released onsite. There were an estimated 81 roseate tern nests at the Marathon Government Center in 2009 (R. Zambrano, pers. comm. 2010).

Since the 2005 hurricanes, the Florida roseate tern population has experienced a 71 percent reduction in the number of nests (R. Zambrano, pers. comm. 2010). The population averaged only 75 ( $SD \pm 1$ ) nests, and a low number of chicks fledged from these colonies. Only 28 and 13 chicks fledged in 2006 and 2007, respectively. Colony productivity, measured as the number of fledged chicks per breeding pair, did not decline drastically after the hurricanes but it continues to be low compared to previous years (R. Zambrano, pers. comm. 2010).

Approximately 300 pairs of breeding roseate terns were estimated between Marathon Key and the Dry Tortugas, Florida, by 1996; although none had nested in the Dry Tortugas for over 10 years (USFWS 1999). The maximum number of adult pairs at Pelican Shoal, estimated from counts of incubating adults and total adults present, has averaged about 300 pairs annually since roseate terns were first documented on the island in 1987. However, the peak count in 1999 was only 170 pairs. The Marathon Government Center roof harbored a maximum of 9 pairs in 1999, down from an estimated 25 pairs in 1997 (Zambrano *et al.* 2000). In 2000, the total number of nesting roseate terns in Florida was 324 pairs, and during the period of 2000 to 2005, the overall population averaged 261 ( $SD \pm 91$ ) nests per year (Zambrano 2001). Using social attraction techniques (decoys and recorded tern vocalizations) in the Dry Tortugas, FFWCC and NPS biologists found 33 and 39 roseate tern nests in 2006 and 2007, respectively (R. Zambrano, pers. comm. 2010). In 2009, 81 roseate tern

nests were counted on Marathon Government Centre roof, and 27 nests in the Dry Tortugas (R. Zambrano, pers. comm. 2010).

A probable nesting record for the mainland of Florida is that of six pairs of terns that nested on a rooftop at Pompano Beach, southeast Florida, in 1969 (McGowan 1969, Stevenson and Anderson 1994). They were reported as common terns, but would have been far out of range for that species and the location on a rooftop is characteristic of roseate terns in the Florida Keys. There is a long history of roseate terns being misidentified as common terns in the Caribbean, so that all breeding records of “common terns” there are suspect except for those in Aruba, Bonaire, Curaçao and the islands off Venezuela (Nisbet 2002, Lee 2009).

#### **2.3.1.2.2 Distribution and numbers in Puerto Rico**

The main island of Puerto Rico is surrounded by many groups of small islands and cays, and the following locations harbor roseate tern colonies at present: Northwest Cays (Cayo Mera and adjacent unnamed cays), Culebra archipelago (Cayo Molinos, Cayo Ratón, Cayo Yerba, and Punta Soldado), Southwestern Cays (Guayanilla - Cayo Guayanilla, and La Parguera - Cayo El Palo, Cayo Media Luna East, Cayo Media Luna West, Cayo San Cristóbal, Cayo Turrumote, and Cayo Turrumote II) and Vieques Island (Saliva 2009).

Although every year roseate terns occur around the Culebra archipelago, they have not nested there in consecutive years since the mid-1990's (Saliva 2009). The number of nesting roseate terns in the archipelago has been steadily declining from a high of over 300 pairs in 1988 to an average of 15 to 25 pairs in the mid 1990's. Between 2000 and 2004, roseate tern estimates from boat surveys have not surpassed 15 breeding pairs (Saliva 2009). Previous observations of roseate terns banded at Culebra but breeding in the Virgin Islands, suggest that the Culebra population may have shifted to the larger colonies of the Virgin Islands (Saliva 2009).

Roseate terns are not common nesters in Vieques Island, where they usually appear in small foraging flocks on the southeast coast in late May (Saliva 2009). Their breeding activities around Vieques Island are not well documented, because nesting occurred at the easternmost tip of the island, where U.S. Navy military activities precluded monitoring. Although reports indicate that roseate terns have nested in Vieques Island prior to 1990, it was not until 2001, when up to 10 pairs were reported nesting at the easternmost point in mid July (Saliva 2009). Since roseate terns in the Caribbean are notorious colony-shifters, there is the possibility that these late nesters were birds that had abandoned a nesting area in nearby Punta Soldado, in the Culebra archipelago, in late June (Saliva 2009). This colony in Vieques Island, however, was abandoned shortly

thereafter; further supporting that it may have been re-nesters. The species has not been observed nesting in Vieques Island again.

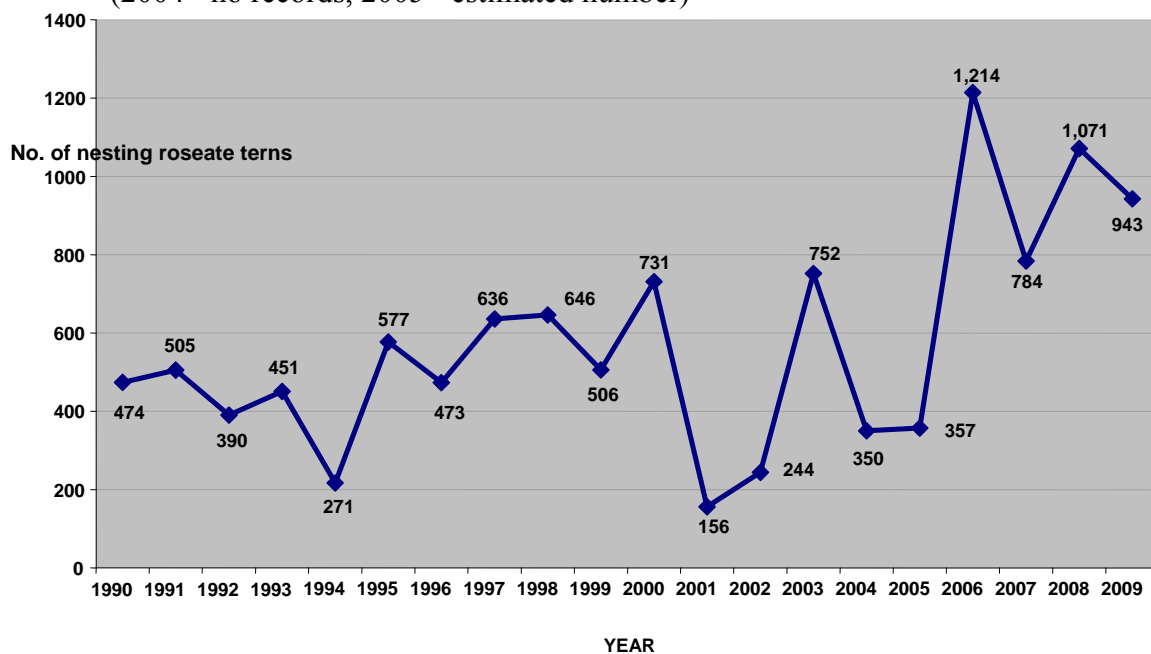
A section of the Uritas Reef on the south coast emerged to form “Cayo Guayanilla”, an unnamed, small coralline/sandy island about 1 km east of Punta Verraco, between Guayanilla and Tallaboa, Puerto Rico (Saliva 2009). Traditionally, roseate tern colonies at Cayo Guayanilla have been relatively small (high of 20 in 1997), and only one pair nested between 2000 and 2004 (Saliva 2009). Shifting of colony sites between birds from La Parguera (to the west) and Cayo Guayanilla is evident from observations of roseate terns banded in La Parguera nesting at Cayo Guayanilla. The reason for such low roseate tern nesting activity at Cayo Guayanilla is not clear; however, the diminishing size of the island may play a role in the selection of larger breeding areas in La Parguera (Saliva 2009).

USFWS biologists have conducted surveys of the Puerto Rico roseate tern populations since 1990, and total nest counts for the Southwestern Cays subpopulation have been recorded for this period (Figure C4). The number of nesting pairs during the period 1990 to 2000 fluctuated from a low of 217 to a high of 731 nesting pairs in 1994 and 2000, respectively. Significant fluctuations were observed between 2001 and 2009, although an increasing trend is apparent. Although no colony counts were conducted in 2001 and 2004 at the Southwestern Cays, nesting roseate terns were observed at Cayo Turrumote II in 2001, and several hundred pairs were estimated from boat observations in 2004. Colonies on the Southwestern Cays appeared to be stable, with similar numbers of nesting pairs as in 2002 and record numbers of fledglings in 2003 (85 percent estimating from one fledged chick per nest). The Cayo Guayanilla colony did not form in 2004, probably due to the loss of over 90 percent of nesting habitat from surge erosion. Estimates from boat surveys at colonies on the Southwestern Cays suggest a stable number of nesting pairs in 2004 compared to 2003 (550 to 650 pairs). On-the-ground nest counts at colonies on the southwest in 2005 showed half the number of nesting terns (~350 pairs) compared to the stable numbers observed from 1990 to 2003 (mean of about 600 pairs). Reasons for these declines are not known.

The number of roseate tern nesting at Culebra Island has been decreasing for the past 15 years with no nesting in 2003. Reasons for this decline and nest site abandonment are not known. Banded birds have been observed in Culebra at colonies in the U.S. Virgin Islands, so colony shifting is occurring. Colonies in cays off northwest Puerto Rico appeared to have increased in 2003, but this was based only on estimates from aerial surveys (no ground-truthing). On-the-ground nest counts in those colonies are needed. Overall, the status of the population appears to be stable from

2002 to 2003. No nesting occurred at Culebra between 2003 and 2004. Colonies in cays off northwest Puerto Rico appear to have increased in 2004, based on off-site colony estimates (less than 75 pairs). All roseate tern colonies in Puerto Rico showed a dramatic decrease in number of nesting pairs in 2005, compared to previous years. Nesting at Culebra has been declining from about 30 pairs in the mid 1990's to no nesting in 2003, 2004, and 2005. Colonies in cays off northwest Puerto Rico appeared to have decreased in 2005 compared to 2004, based on off-site colony estimates.

**Figure C4.** Counts of Roseate Terns breeding in Southwestern Puerto Rico (2004= no records, 2005= estimated number)



In 2006, USFWS biologists reported a relatively high number of pairs at Culebra (100 pairs from a normal 15 pairs or less), and in Cayo Mera, Northwestern Cays, approximately 75 to 100 pairs were reported. Number of nests in the southwest population increased in 2006 to 1,214; 784 nests in 2007; and 1,071 nests in 2008. Colonies in the Northwest were estimated at 200 to 250 pairs in 2008. Roseate tern colonies at Culebra Island have not been censused since the mid 1990s. In 2009, colonies in the Northwest Cays were estimated at about 300 pairs, and a total of 943 roseate tern nests were counted during peak breeding in the Southwestern Cays.

### 2.3.1.2.3 Distribution and numbers in the U.S. Virgin Islands

In recent years, roseate terns have nested at 17 different sites in the U.S. Virgin Islands: Booby Rock, Carval Rock, Congo Cay, Cricket Rock, Dog Island, Flanagan Island, Flanagan Rocks, Flat Cay, Kalkun Cay, LeDuck Island, Little Flat Island, Pelican Cay, Ramgoat Cay, Rata Island, Saba Island, Shark Island, Turtledove Cay, and Two Brothers (Pierce 2001). The top five islands, ranked in order of frequency of occupation, are Kalkun Cay, Shark Island, Saba Island, LeDuck Cay, and Pelican Cay. Table C2 summarizes nest counts at these 17 sites over 15 years, 1987 to 2001 (Pierce 2001, and unpubl. data). The data show considerable year-to-year movement among sites, with the average site occupied for only 4 of 17 years and none occupied for more than 12 of 17 years. Total numbers of nests over all sites varied among years from 481 to 2,258 pairs, with the mean of 1,252 only 55 percent of the maximum (Table C3).

**Table C3.** Clutch-sizes of roseate terns at the Parguera colonies, southwestern Puerto Rico, 1990 to 1994 (from Shealer 1995, Table 6.4).

Year	Census dates	Clutch-size			Total nests	Mean $\pm$ se
		c/1	c/2	c/3		
1990	18 June	343	130	1	474	1.28 $\pm$ 0.02 <sup>a</sup>
1991	3 June	16	86	3	105	1.88 $\pm$ 0.04 <sup>b</sup>
1992	28 to 29 May	322	68	0	390	1.17 $\pm$ 0.02 <sup>c</sup>
1993	27 May, 3 June	71	198	5	274	1.76 $\pm$ 0.03 <sup>b</sup>
1994	31 May	43	29	0	72	1.40 $\pm$ 0.05 <sup>a</sup>
Total		795	511	9	1,315	1.402 $\pm$ 0.012

<sup>a, b, c</sup> Entries with different letters are significantly different from each other.

Pierce (2009) estimated the roseate tern breeding population in the U.S. Virgin Islands at 500 to 1,000 pairs for 2004. Nest counts likely underestimated the actual nesting population, because nests were often hidden, or placed in positions too precarious to observe directly (Pierce (2001). Nest counts of inaccessible colonies were estimated, based on the number of adults in the colony, by observation from a boat or using binoculars from a vantage point on the main island (Pierce (2001). Colonies in the U.S. Virgin Islands have not been monitored since 2005; although DPNR staff has conducted periodic boat colony estimates. Estimates from boat surveys at colonies in the U.S. Virgin Islands



suggested stable numbers of roseate tern nesting pairs in 2005 (~1000 pairs) (Pierce, DPNR, pers. comm., 2005). The large year-to-year variation in numbers of nests at colonies in the U.S. Virgin Islands led to the idea that exchange of breeders occurs between the U.S. Virgin Islands, British Virgin Islands, and Puerto Rico (namely, Culebra archipelago). The occurrence of large numbers of breeding birds in the U.S. Virgin Islands (>1,000 pairs) has coincided with lower numbers in the British Virgin Islands and vice versa (Pierce 1996); therefore, the combined total roseate tern population in the British Virgin Islands and U.S. Virgin Islands is estimated at about 2,500 pairs (Pierce 2009). In the U.S. Virgin Islands, most of the roseate tern nesting cays are located off St. Thomas and St. John; are difficult to access, and are uninhabited (Pierce 2009).

#### **2.3.1.2.4 Survival and dispersal**

Shealer *et al.* (2005) estimated the annual survival rates of breeding adult roseate terns in the Caribbean, based on 233 birds color-banded at the Parguera colonies in 1991 to 1992 and 116 resightings at these and other sites in 1992 to 1994. Using mark-recapture methodology, the initial estimate of the annual survival rates of adults was 0.62, which was adjusted to 0.70 to correct for band losses and then to 0.71 to 0.80 to take account of emigration. Even the high end of this range is lower than the most recent estimate of 0.835 for the Northeast population (Spendelov *et al.* 2008). This is contrary to the expectation that survival rates would be higher in tropical than in temperate populations (Nisbet and Ratcliffe 2008).

Shealer *et al.* (2005) also estimated juvenile survival, based on 71 near-fledglings color-banded in the Parguera colonies in 1991 and 9 resightings in 1993 to 1994. They estimated survival from fledging to age 2 as  $0.44 \pm 0.11$ , survival from age 2 to age 3 as  $0.71 \pm 0.45$ , from which survival from fledging to age 3 was calculated as 0.31. Because of the small sample sizes, these estimates have a wide range of uncertainty, but they are valuable as one of only two estimates of juvenile survival for tropical roseate terns (Nisbet and Ratcliffe 2008). They are similar to the values reported by Lebreton *et al.* (2003) for the Northeast population in non-hurricane years, for which the mean value for survival to age 2 was  $0.415 \pm 0.031$  (see section 2.3.1.2.5 for the Northeast population).

Based on the same mark-resighting data, Shealer *et al.* (2005) also estimated dispersal rates among three groups of colonies: Parguera (southwestern Puerto Rico), Culebra (northeastern Puerto Rico) and the U.S. Virgin Islands farther to the east. Emigration from Culebra Island was high (7 of 10 birds over the course of the study), and immigration to Culebra Island was limited to 2 of 31 birds from the U.S. Virgin Islands.

Chronic breeding failure apparently was responsible for the extremely high emigration estimate obtained for the Culebra Island colonies, and presumably was the cause of the complete abandonment of this region by terns in 1997 (Shealer *et al.* 2005a). Excluding the data for Culebra Island, Shealer *et al.* (2005) estimated annual dispersal rates among the other sites to be in the ranges 0.02 to 0.13: these rates were comparable to those among colonies in the Northeast (Lebreton *et al.* 2003: see section 2.3.1.2.6 for the Northeast population).

Breeding success, therefore, appeared to correlate positively with breeding-site fidelity in roseate terns (Shealer *et al.* 2005a). The low emigration rate estimated for terns from southwestern Puerto Rico colonies might be explained by absolute distances among the three regions, characteristics of the nesting habitat, or a combination of both of these factors (Shealer *et al.* 2005a). Except for southwestern Puerto Rico, all known breeding colonies east of the Dominican Republic are located on elevated cays (up to 100 m above sea level) with volcanic or clay substrate and sparse to thick vegetation cover. In southwestern Puerto Rico, roseate terns nest exclusively on exposed coralline cays, barely above the high water mark, and with little or no vegetation cover (Shealer *et al.* 2005a). Therefore, habitat specificity may influence movement decisions by breeding adults, but further study is warranted (Shealer *et al.* 2005a).

The survival estimates for adults from both the Northeast and Caribbean populations suggest a high rate of annual mortality relative to other species of marine birds, including terns (Shealer *et al.* 2005a, Stenhouse and Robertson 2005). Since little adult mortality is witnessed at breeding colonies, and wintering ranges of the two populations overlap, the collective findings are consistent with the interpretation that most mortality to adults in both populations occurs outside of the breeding season.

#### **2.3.1.2.5 Breeding performance in Puerto Rico**

Shealer (1995) studied the breeding performance of roseate terns at the Parguera colonies from 1990 to 1994. See section 2.3.1.6, below, for a description of those colony sites and their habitats. Roseate terns arrive at La Parguera in late April to early May and egg-laying begins around mid to late May. Breeding success varied markedly among years. Annual mean clutch-sizes varied from 1.17 to 1.86 with an overall mean of 1.40 (Table C3). Mean egg volume was about 19.2 ml, with no significant variation among years or between 1-egg and 2-egg clutches (Shealer 2009, Table 6.5). Hatching success was very high (>93 percent) in 1 year, but near-zero in another year because most of the birds deserted the colony in the third week of incubation. Productivity (chicks raised to fledging per

pair) varied from 0.01 to 1.56, with an overall mean between 0.31 and 0.42 (Table C4).

Douglas (2000) studied roseate tern breeding in the same colonies at La Parguera in 1999. He reported that roseate terns synchronized their breeding activities in small groups that clustered together; with the earliest nesters selecting the most favorable habitat (center of the colony) and late-nesting groups selecting peripheral areas. The earliest egg-laying dates reported were separated from the latest egg-laying dates by approximately 11 days. The first chicks gained limited flight capability 22 days after the first eggs hatched, and by day 44 to 46, the majority of adults and juveniles had departed southwestern Puerto Rico (Douglas 2000). Therefore, roseate terns departed southwestern Puerto Rico just 3 weeks after fledging and at this time, juveniles were still dependent upon their adults. This phenology is similar to that observed in the northeastern United States, where chicks fly at age 22 to 28 days and remain dependent upon adults for at least 8 weeks following fledging.

Mean clutch-sizes at the two La Parguera colonies in 1999 were 1.13 and 1.74 (Douglas 2000). The mean volume of single egg clutches was 19.37 ml (se = 0.15, n = 35) (Douglas 2000). Defining the linear growth rate (LGR) as the slope of a line fitted to mass data between ages 3 to 14 days (following Nisbet *et al.* 1995 and Nisbet *et al.* 1998; Ramos *et al.* 2001) the mean value of LGR at the Turrumote II colony (4.0 g/day) was lower than those recorded in the Northeast population, where LGR of A-chicks (first-hatched chicks) at Bird Island, Massachusetts, ranged from 5.97 to 6.54 g/day, and LGR of B-chicks (second-hatched chicks) ranged from 4.70 to 6.30 g/day (Nisbet *et al.* 1995). LGR of A-chicks at Falkner Island, Connecticut, ranged from 5.68 to 6.09 g/day, and LGR of B chicks ranged from 4.66 to 6.44 g/day (Nisbet *et al.* 1995). The average fledging mass of roseate tern chicks in southwestern Puerto Rico was 81 g, or 77 percent of mean adult mass; which may be considered an at-risk low mass for a roseate tern fledgling from the Northeastern population (Douglas 2000). The average asymptotic mass (a predictor of post-fledging survival) of roseate tern chicks in the northeastern United States is higher than that of Caribbean roseate terns (Douglas 2000). At Bird Island, the average asymptotic mass of A-chicks ranged from 100.3 to 103.2 g, and that of B-chicks ranged from 90.4-99.8 g; whereas at Falkner Island they ranged between 94.5 to 101.1 g and 85.1 to 98.1 g, respectively (Nisbet *et al.* 1995). From the combined data, the asymptotic mass of A-chicks in the northeastern United States ranges approximately 86 to 94 percent of mean adult mass, and the asymptotic mass of B-chicks ranges approximately 77 to 91 percent (Douglas 2000). Four of the Puerto Rico roseate tern chicks did fledge at high mass (95 to 101 g), indicating that they are capable of growing larger if adequate food is available (Douglas 2000).

**Table C4.** Reproductive success of roseate terns at the Parguera colonies, southwestern Puerto Rico, 1991–1993 (from Shealer 1995, Table 6.7).

Year	Colony site	Nests	Hatching success	Productivity (chicks fledged/pair) (mean $\pm$ se)
1991	Media Luna W	71	93.4 percent	1.56 $\pm$ 0.06
1992	Turumote	154	0.6 percent	0.01
1992	Media Luna E	236	unknown	0.42
1993	San Cristobal	131	unknown	0.45 to 0.90
1993	Turumote II	371	unknown	0.10 to 0.20
Total		963		0.31 to 0.42

#### 2.3.1.2.6 Breeding performance in the U.S. Virgin Islands

Table C5 summarizes data on mean clutch-sizes at 17 colony sites in the U.S. Virgin Islands from 1987 to 1998 (from J. Pierce, U.S. Virgin Islands Division of Fish and Wildlife, unpubl. data). Mean values over all sites varied markedly among years, from 1.16 in 1994 to 1.69 in 1991. There was considerable variation among sites within years, although clutch-sizes at the larger colonies were generally similar and tended to track the variations in the overall mean (Table C5).

Douglas (2001) also studied roseate tern breeding at LeDuck Island, U.S. Virgin Islands, in 2001. He reported a mean clutch size of 1.39 eggs (se = 0.03, n = 335), which was similar to the mean value of 1.40 recorded by Shealer (1995), and was intermediate between the values of 1.13 and 1.74 recorded by Douglas (2000) in Puerto Rico colonies. The mean volume of single egg clutches was 18.99 ml (se = 0.15, n = 82) at LeDuck, similar to those recorded in previous years in Puerto Rico (see previous section). The mean volume of single egg clutches (18.99 ml) was less than the mean volume of first eggs in two-egg clutches ( $t(133) = 2.28$ ,  $p < 0.05$ ) and greater than the volume of second eggs ( $t(133) = 3.98$ ,  $p < 0.001$ ) (Douglas 2001). The volume of eggs within two egg clutches differed significantly ( $t(104) = 6.16$ ,  $p < 0.001$  ( $x = 19.5$  vs.  $x = 18.0 \text{ cm}^3$  ( $1.1 \text{ in}^3$ ))). Hatching success ranged from 0.32 to 0.76 and averaged  $0.56 \pm 0.06$  at LeDuck, whereas average fledging success was  $0.56 \pm 0.07$  and the average productivity was  $0.35 \pm 0.07$  (Douglas 2001). This means that the

LeDuck colony of 1,400 nests successfully hatched 700 to 868 chicks and fledged 343 to 565 offspring (Douglas 2001).

Mean LGR of roseate tern chicks at LeDuck was 4.22 g/day (se = 0.18, n = 103) (Douglas 2001). This was similar to the growth rate recorded for roseate terns in southwestern Puerto Rico during 1999 (4.0 g/day  $\pm$  0.42, n = 119; Douglas 2000). In both years, the chick growth data included all intervals between weighings for all chicks in the samples during the linear phase of growth, irrespective of age (Douglas 2001). The highest growth rate was attained during the first 5 days (Fig. C5). Fledging measurements were also similar between cays of southwestern Puerto Rico in 1999 and LeDuck in 2001. However, the average mass at fledging was 86 percent of the mean adult mass compared to 77 percent mean adult mass recorded at La Parguera in 1999 (Douglas 2001).

Table C5. Average clutch size of Roseate Terns in the U.S. Virgin Islands, 1987–1998.

Nesting Colony	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Kalkun	1.13	•	•	1.37	“a”	1.20	“a”	1.14	1.48	•	1.75	1.49
Shark	1.28	•	1.24	•	1.68	1.44	1.67	•	1.29	1.37	•	1.15 “b”
Pelican	1.14	1.73	•	•	•	•	1.55	•	1.58	•	•	•
Flat	1.27	•	•	•	•	•	•	1.35	•	•	1.89	•
Booby Rock	1.43	•	•	•	•	•	•	•	•	•	•	•
Flanagan Rocks	1.28	1.61	•	•	•	1.16	•	•	•	•	•	•
Saba	•	1.62	1.35	1.31	1.75	•	•	•	“a”	1.59	“a”	“a”
Flanagan	•	1.64	•	•	•	1.20	•	•	•	1.17	•	•
Cricket	•	1.32	•	•	•	•	•	•	1.23	•	•	•
Dog	•	1.64	1.42	•	•	•	•	•	•	•	•	•
Rata	•	•	•	1.64	•	1.30	•	•	2.14	•	1.39	•
Two Brothers	•	•	•	1.50	•	•	•	•	•	•	•	“c”
LeDuck	•	•	•	•	1.66	•	•	•	•	•	1.53	1.43
Carval Rock	•	•	•	•	•	1.06	•	•	•	•	•	•

Congo	•	•	•	•	•	•	•	•	•	•	•	•
Little Flat	•	•	•	•	•	1.06	•	•	•	•	•	•
Ramgoat	•	•	•	•	•	•	•	•	•	•	1.64	•
Weighted mean (d)	1.224	1.625	1.314	1.384	1.692	1.205	1.609	1.160	1.458	1.410	1.592	1.444

“a”: no estimate of clutch size possible due to inaccessible location of nests

“b”: numerous broken eggs

“c”: re-nesting after Congo colony failure; mostly 1-egg nests

“d”: mean calculated by weighting each value in proportion to the number of nests in the colony in that year.

### 2.3.1.2.7 Breeding performance in Florida Keys

Zambrano (2001) measured growth rates of roseate tern chicks at Marathon Cay and Pelican Shoal, Florida) in 2001. LGR for A-chicks was 4.55 g/day at Marathon Cay and 5.04 g/day at Pelican Shoal (Zambrano 2001). These figures are higher than those reported from other tropical colonies of roseate terns, but lower than temperate roseate terns (Zambrano 2001). Ramos *et al.* (2001) found an LGR of 4.07 g/day in the Seychelles population. Chicks infested with ticks in the Seychelles showed much lower LGR (0.24 g/day). In contrast, mean values of LGR for A-chicks in temperate roseate tern populations are as high as 6.54 g/day (Zambrano 2001). Similarly, the highest asymptotic mass recorded in Zambrano's study (90.3 g), using a minimum of two measurements, was at least 5.9 g less than the lowest reported asymptotic mass for roseate tern A-chicks in the northeast United States (Nisbet *et al.* 1998; Zambrano 2001). Additionally, chicks in Florida did not exhibit a well-defined asymptotic mass period as has been found in the population in the northeastern United States (Nisbet *et al.* 1998). It appears that Florida roseate terns exhibit a more linear growth up until the time of fledging (Zambrano 2001). The slower growth and lower asymptotic mass of Florida roseate terns, when compared to the Northeast U.S. population, is consistent with other studies comparing growth and energetics of temperate and tropical seabirds. It is presumed that the slower growth of tropical birds is due to the lower availability of food and the resulting lower rate of energy intake (Zambrano 2001).

Clutch size range from one to three eggs (Zambrano 2001, Table 7). Modal clutch size was two eggs in 2000 and one egg in 2001. Zambrano (2001) found a clutch size independence between colony sites in 2000 ( $\chi^2 = 0.04$ ,  $p = 0.83$ ) but dependent in 2001 ( $\chi^2 = 6.16$ ,  $p = 0.01$ ). The colony site dependence in 2001 might be due to the large number of re-nesters. Furthermore, clutch size was dependent between years when colonies were pooled ( $\chi^2 = 259.88$ ,  $p < 0.83$ ) (Zambrano 2001). Marathon colony hatching success was 0.84 chicks hatched/nest (50 percent) and 1.07 chicks hatched/nest, (76.9 percent) respectively (Zambrano 2001). Pelican Shoal data was not available. Marathon colony fledged 15 chicks in 2000 and 17 in 2001. Pelican Shoal fledged 150 chicks in 2000 and 32 in 2001, despite the greater number of nests in the second year (Zambrano 2001).

Zambrano's study shows that a roof colony had equal or greater annual reproductive success compared with a ground colony. Marathon productivity was 0.42 chicks fledged/nest in 2000 and 0.61 chicks fledged/nest in 2001. Pelican Shoal productivity was 0.45 in 2000 and 0.06 in 2001. Pelican Shoal low productivity in 2001 may possibly be underestimated due to the large number of re-nesting birds (Zambrano 2001). A higher estimate for Pelican Shoal productivity could be



calculated if fledged chicks per breeding pair is used, instead of fledged chicks per nest (Zambrano 2001).

Zambrano (2001) reported that the linear growth rates of Florida A-chicks in 2001 were higher than those at other tropical roseate tern colonies (*e.g.*, Seychelles), but lower than temperate populations. Chicks in Florida did not exhibit a well-defined asymptotic mass period as has been found in Northeast U.S. populations; but rather a more linear growth until the time of fledging (Zambrano 2001). Colony productivity was similar for the Marathon (roof) (0.42) and Pelican Shoal (ground) (0.45) colonies, respectively; but it was significantly higher on Marathon (0.61) than on Pelican Shoal (0.06) in 2001 (Zambrano 2001).

#### **2.3.1.2.8 Sex-ratio**

There is no direct evidence on the sex-ratio of Caribbean roseate terns. At La Parguera, 3-egg clutches comprised 0.7 percent of all clutches enumerated in 1990 to 1994 (Table C34). This is smaller than the 2 to 5 percent prevalence of ‘super-normal’ clutches in the Northeast population, but this has to be evaluated in relation to the fact that the mean clutch-size is smaller in the Caribbean (Tables 3, C3 and C5). In the 2 years when mean clutch-sizes at La Parguera were comparable with those in the Northeast, the frequency of 3-egg clutches was 2.1 percent. This suggests that female-female pairs occur at a frequency of at least 2 percent in the population, although in most years they would lay only one egg each and would not be detected through the occurrence of ‘supernormal’ clutches.

#### **2.3.1.2.9 Population dynamics and metapopulation structure**

Shealer *et al.* (2005) speculate that higher pre-breeding survival of Caribbean birds may be offset by lower clutch/brood sizes and higher variance in annual reproductive success, relative to temperate-zone populations. In western North Atlantic colonies, mean clutch sizes usually range from 1.6 to 1.9 eggs/nest, whereas means typically range from 1.1 to 1.8 eggs in Caribbean colonies (Shealer *et al.* 2005a). In the absence of predation, reproductive success in North American colonies is usually higher than in Caribbean and other tropical colonies, where roseate terns are prone to suffer heavy egg predation, ectoparasite infestation, and to abandon colonies during the breeding season (Shealer *et al.* 2005a). Therefore, the net result of higher nestling mortality and lower pre-breeding mortality (Caribbean) compared to lower nestling mortality and higher pre-breeding mortality (North America) may be such that actual recruitment into the breeding population at age 3 is similar (Shealer *et al.* 2005a). Despite distinct life-history differences between North American and Caribbean roseate terns during the breeding season, the adult survival rates suggest that these two populations are regulated by high adult

mortality, which most likely occurs during the non-breeding season when their ranges overlap and they are exposed to similar environmental conditions (Shealer *et al.* 2005a). Whether high adult mortality is a result of human predation, density-dependent effects on food resources, or some other cause remains to be determined and conservation efforts directed toward this endangered species only during the breeding season may be insufficient to aid its recovery (Shealer *et al.* 2005a; Monticelli *et al.* 2008).

Populations of Caribbean roseate terns are regulated by complex factors that are not well-understood, and previous experience at the colony site, *i.e.* predation events or colony-level reproductive success, does not appear to influence the choice of nesting location (Pierce 2001).

#### **2.3.1.3 Genetics, genetic variation, or trends in genetic variation:**

No genetic studies of the Caribbean population have been reported. See section 2.3.1.3 for the Northeast population for studies of the populations in the temperate North Atlantic Ocean and other parts of the world

#### **2.3.1.4 Taxonomic classification or changes in nomenclature:**

There have been no changes in the taxonomic classification or nomenclature of roseate terns. Both the Northeast and Caribbean populations remain classified in the subspecies *Sterna d. dougallii*. Lashko (2004) confirmed that this subspecies is distinct from *S. d. gracilis* of the Indo-Pacific region, but she did not evaluate the Caribbean birds. See section 2.3.1.4 for the Northeast population for changes at the generic level.

#### **2.3.1.5 Spatial distribution, trends in spatial distribution, or historic range:**

There have been no recent changes in spatial distribution of Caribbean roseate terns: the distribution delineated in the various chapters of Bradley and Norton (2009) and summarized in their Table 29.6 is essentially the same as that delineated by Nisbet (1980) and van Halewyn and Norton (1984). However, there have been no recent records from Las Aves, Los Roques, Curaçao, Bonaire, or islands off Belize and Honduras, where nesting roseate terns had been recorded prior to 1960: it is not clear whether this reflects lack of recent surveys or a contraction in the south and west of its Caribbean range.

#### **2.3.1.6 Habitat or ecosystem conditions:**

Roseate terns are uncommon over much of their range; and in temperate regions, they generally nest under cover (Burger and Gochfeld 1988). In northwestern Europe and northeast North America, roseate terns generally nest under vegetation or in rock crevices, but in the Caribbean their nests are exposed on

coral rocks, while in Australia they often nest on bare sand with no vegetation cover (Del Hoyo *et al.* 1996). In the Caribbean, where they rarely breed on large islands, roseate terns nest primarily on small offshore islands, rocks, cays, and islets (Burger and Gochfeld 1988; Hoffman *et al.* 1993; USFWS, 1993; Saliva 2000). They have been reported nesting near vegetation or jagged rock, on open sandy beaches, close to the water line on narrow ledges of emerging rocks, or among coral rubble (Saliva 2000). In Puerto Rico and the Virgin Islands, roseate terns may choose a suitable nesting location 1 year and ignore it in other years; and the same islands may be used in successive years (Pierce 1996; Douglas 2000). Colonies are highly susceptible to disturbance in the early stages of egg laying, and may relocate within the breeding season, often joining an established group of nesters. Late-formed colonies are small (<100 pairs) and rarely raise young to fledging (Pierce 1996). Shelter for chicks may be found near nests (*e.g.*, short vegetation, rocks, crevices, debris); however, most nests are completely exposed (USFWS, 1993).

The Southwestern Cays of La Parguera form an array of more than 20 islands that lies between 5 m and 3 km off the south coast of Lajas and Cabo Rojo (Saliva 2009). These islands are from 0.5 to several hectares in size, composed mostly of red mangrove (*Rhizophora mangle*) and coralline deposits (although the larger ones have volcanic substrate); where roseate terns nest with sandwich and cayenne terns (*Thalasseus eurygnatha*) (Saliva 2009). Usually, eggs are laid on fine coralline substrate, with or without nearby vegetation and coral rubble (Saliva 2009).

Roseate terns arrive in the Culebra archipelago from migration in late April and begin nesting in mid to late May (Saliva 2009). They have nested at Cayo Ratón, Cayo Yerba, Cayo Molinos, and Culebra Island (Punta Soldado), but since 2000, Cayo Molinos have been the only cay used by roseate terns. Most nest-sites at Cayo Ratón and Cayo Yerba are close to vegetation (*Cyperus* sp.), whereas nest-sites at Cayo Molinos and Culebra Island are on volcanic rock with no vegetative cover (Saliva 2009).

Culebra Roseate terns do not show strong nest-site fidelity, and adults would arrive at potential breeding sites and may abandon these shortly thereafter (Saliva 2009). The reasons for colony abandonment are not clear, but the presence of peregrine falcons and red-tailed hawks (*Buteo jamaicensis*) during the period of colony-site selection in early May at Culebra (possibly attracted by the large masses of sooty terns (*Onychoprion fuscatus*) that arrive in late April) may deter roseate terns from nesting (Saliva 2009). If successful breeding occurs, adult and young roseate terns depart Culebra around late July.

Roseate terns at La Parguera in southwestern Puerto Rico, nest on bare, offshore coralline cays, with little or no vegetation; and during unusual wet periods in the dry season, ephemeral herbaceous vegetation grows on these cays before roseate terns arrive for breeding (Saliva, USFWS, pers. obs.). When this occurs, large

numbers of roseate terns select vegetated areas for nesting. In 1999, after one of such rainy events, when coastal strand vegetation grew at both Turrumote II and Media Luna East cays in La Parguera, Douglas (2000) found that roseate terns preferred to nest in coastal strand vegetation. At Media Luna East, 92 percent of nests were located near this vegetation, and on average, the vegetation was within 3.8 cm of the nest. The densest aggregations of nesting roseate terns were found in this vegetation at both colonies (Douglas 2000). Furthermore, at Turrumote II, hatching success was highest in the vegetated portion of the colony, and predation was much lower. These data suggest that Caribbean roseate terns will select vegetated habitat when it is available (Douglas 2000). Furthermore, the availability of cover probably enhances productivity and reduces thermal stress. At both colonies, those roseate terns that did not nest in vegetation tended to seek shelter among rocks or along drift logs and other debris (Douglas 2000). The presence of cover influenced the arrangement of nests; where cover was abundant the roseate terns nested in clusters, but where cover was sparse, the nest arrangement was linear and diffuse. For example, at Turrumote II, the south side of the cay was sheltered from wave action by a fringing reef, but the north side was exposed to beach erosion and during heavy seas, waves wrapped around the lee side of the island and surged onshore (Douglas 2000). Vegetation was abundant on the south side of the cay and roseate terns nested in a relatively tight cluster. By contrast, vegetation was sparse on the north side of the cay, and roseate tern nests were sparsely arranged along plant vines, small drift logs, or on open sand (Douglas 2000).

Although reports indicate that roseate terns have nested in Vieques Island prior to 1990, it was not until 2001 when up to 10 pairs were reported nesting at the easternmost point in mid-July on jagged, volcanic substrate without vegetation cover (Saliva 2009). It is not known to what extent U.S. Navy bombing practices around the time when roseate terns were selecting nest-sites (late April to early May) deterred adults from selecting Vieques Island for nesting (Saliva 2009).

The composition and size of Cayo Guayanilla change every year, depending on storms and heavy surf that erode this small island; making it an unstable substrate for nesting seabirds. Sometimes the island, barely above sea level, would submerge completely for months at a time and re-emerge later (Saliva 2009). Vegetation (grass, probably *Sporobolus* sp.) had been growing on this island until 2003, when erosion caused by heavy surge removed most of it. By the summer of 2004, the island had been reduced to a few square meters with no vegetation (Saliva 2009). Usually, eggs are laid on fine coralline substrate, with or without nearby vegetation (Saliva 2009).

The volcanic Northwest Cays are offshore rocks and cays (14), frequently over washed by heavy north swells, which form jagged depressions on the limestone rock that provide nesting habitat for some seabird species (Saliva 2009). Vegetation is only found on the larger cays (>0.5 ha). A few of these are connected to the main island through a thin strip of volcanic material. The cays

that are used by seabirds, however, are separated from other land and access is difficult unless sea conditions allow climbing over the jagged edge. Two species of seabirds nest there: roseate tern and bridled tern (*Onychoprion anaethetus*) (Saliva 2009). Roseate terns arrive to the breeding areas in early May, and nesting begins in mid to late May (Saliva 2009). They shift colony sites among years, but generally always use from 1 to 3 particular cays for breeding. Young fledge in mid-July and leave the breeding areas with their parents in late July or early August. Population estimates from aerial surveys show a steady increase in colony size from 75 pairs before 2000, to close to 200 pairs in 2004 (Saliva 2009). These colonies, however, have not been visited late in the breeding season to estimate fledging success, but marine biologists working offshore have observed flocks of adult and young roseate terns foraging offshore in early August 2003, which suggests that colonies in the Northwest Cays successfully produce fledglings (Saliva 2009).

In the U.S. Virgin Islands, the nesting islands differ in their topography and habitat characteristics, but generally most are small, steep, rugged and not easily accessible (Pierce 2001). Habitats on the cays are varied, but all are included in the subtropical dry forest (Ewel and Whitmore 1973).

In Florida, throughout the period of the 1960s to 70s, the primary roseate tern nesting location was in the Dry Tortugas, but the colony gradually diminished in size in the late 1970s (Zambrano 2001). Concurrently, nesting shifted to a small number of sites in the middle and lower Florida Keys, and by 1990 virtually no nesting occurred in the historical Dry Tortugas sites and the main nesting colony established itself at Pelican Shoal (Zambrano 2001). Nesting sites in Florida include small sand and coral rubble islands, and tar and gravel-covered rooftops on two-story high buildings. The paucity of islands with suitable nesting substrate and minimal human disturbance appear to have contributed to population reduction and shifting from historical breeding grounds (Zambrano 2001).

#### **2.3.1.7 Other:**

##### **2.3.1.7.1 Predation**

Colonies of roseate terns in Florida, Puerto Rico, and the U.S. States Virgin Islands are vulnerable to predation by laughing gulls, peregrine falcons, red-tailed hawks, yellow-crowned night herons (*Nyctanassa violacea*), fire ants (*Solenopsis* spp.), land crabs (*Gecarcinus ruricola*), sally light-foot crabs (*Grapsus grapsus*), and hermit crabs (*Coenobita clypeatus*) (Pierce 2009; Saliva 2009). On low-lying cays, they are preyed upon by ruddy turnstones (*Arenaria interpres*) and American oystercatchers (*Haematopus palliatus*) (Saliva 2000; Pierce 2009; Saliva 2009). Other potential predators include magnificent frigatebirds (*Fregatta magnificens*), and cattle egrets (*Bubulcus ibis*) (Saliva 2000; R.

Zambrano 2010, pers. comm.). Fire ants are found on all offshore cays and can enter pipped eggs or kill newly hatched chicks (Pierce 2009; Saliva, USFWS, pers. obs. 1999). Predation by fire ants and predation by gulls appear to be the most important source of mortality for roseate tern eggs and chicks in the U.S. Virgin Islands (Pierce 2001).

In comparing nest-site selection in colonies at the Culebra Archipelago, Puerto Rico, with that at New York colonies, Burger and Gochfeld (1988) found that roseate terns in Culebra nested closer to other roseate terns (conspecifics), farther from vegetation which was shorter, with less cover around their nests, and with greater visibility indices than those nesting in New York. They attributed these differences to the kind and nature of predators present at the two sites, and the absence of common terns to provide early warning and anti-predator defense in Puerto Rico (Burger and Gochfeld 1988). Roseate terns in Culebra nest in single-species colonies and one of its main chick predators, land crabs, hide during the day at the base of clumps of vegetation.

Burger and Gochfeld (1988) attributed the differences on nest-site selection at two Puerto Rico colonies to the presence of large populations of land crabs on Cayo Ratón, but not on Cayo Molinos. Cayo Molinos had Sally-lightfoot crabs, which only came to the edges of the roseate tern colony to drag dead chicks into the water to eat; but Burger and Gochfeld (1988) failed to see them within the colony or dragging live chicks. However, they observed land crabs on Cayo Ratón dragging away two live roseate tern chicks (1 to 2 days old), and they found seven chicks already dragged into crevices by crabs which had already eaten the legs and feet of some (Burger and Gochfeld 1988). The crabs came out from under piles of rocks, and dragged the squeaking chicks under the rocks with them. Burger and Gochfeld (1988) had expected that parents could defend their chicks against such attacks, but 6 of 15-banded 1- to 3-day-old chicks disappeared overnight in one plot heavily infested with the crabs. Thus, nesting more in the open with sparse nearby vegetation and vertical rocks (without crevices and overhangs) may reduce land crab predation while still providing some protection from solar radiation and aerial predators such as laughing gulls. Roseate terns selected nest sites with some vegetation cover, no doubt to provide additional shade or predator protection for young chicks (Burger and Gochfeld 1988).

Shealer and Burger (1992) suggested that reproductive success was low for a roseate tern colony in southwestern Puerto Rico in 1990 because of the effect of predators. Although they observed only two instances of avian predation, chicks disappeared between colony censuses and only 16 of 81 chicks fledged. Additionally, colony defense by adult terns reduced the amount of time available for foraging, and thus prevented chicks from receiving as much food as they would have in the absence of predator

disturbance (Shealer and Burger 1992). Despite the highest adult aggression levels towards intruders, most chick mortality occurred during the hatching period, and they suspected land crab predation. Although the small tern colony that they studied responded appropriately to the most conspicuous types of intrusion (*e.g.*, hawks), they were unable to defend successfully against land crabs; and the persistent disturbance caused by laughing gulls may have facilitated crab predation on young roseate tern chicks (Shealer and Burger 1992).

Of all roseate tern colonies in Puerto Rico, the land crab is only found at Cayo Ratón, Culebra, where it is abundant. Shealer and Burger (1992) occasionally observed small groups of roseate terns hovering over something on the ground and assumed that these disturbances were caused by land crabs. Although Shealer (1993) has never witnessed direct predation by sally light-foot crabs on roseate tern chicks, he removed a live roseate tern chick from under a rock and found that its abdomen had been cut open and its viscera exposed; and he suspected that this injury was caused by a sally light-foot crab. Hermit crabs, which are abundant in virtually all cays in southwestern Puerto Rico and prey upon hatchling brown pelicans (*Pelecanus occidentalis*) (Saliva, USFWS, pers. obs.), may also be responsible for attacking very young roseate terns from under the rocks. Roseate tern chicks begin to make their way toward the edge of the island after the parents stop brooding them (approximately 5 days of age). The chicks spend most of their time hiding under rocks to keep cool, emerging only when a parent returns with food (Shealer 1993). Chicks are rarely seen again until they near fledging, when they emerge to strengthen their wings in preparation for flight (approximately 20 days of age). Thus, for a 2-week period, chicks may be vulnerable to crab predation (Shealer 1993). Although adult terns are usually aggressive towards crabs that occasionally travel across the surface of the island, they are probably unable to detect predation that occurs under the coral rocks (Shealer 1993).

In 1997, Shark Island was selected for a crab removal experiment in the U.S. Virgin Island because roseate terns nesting on Shark Island had rarely fledged young, and crab predation on nestlings was suggested as a possible cause. Rats were eliminated as a possible cause in 1998, after no rats were caught in traps (Pierce 2001). The island was small enough to feasibly trap and remove land and hermit crabs. After several days of baiting, fewer than half a bucket-full of hermit crabs was caught on Shark Island. This was less than was expected and no land crabs were trapped (Pierce 2001). During the crab baiting activities on Shark Island, Pierce (2001) discovered that crabs weren't as prevalent as previously thought and that large fire ant colonies were abundant on the island, suggesting that the ants may be a bigger management problem than crabs. Fire ants could be the cause for colony abandonment because they can enter eggs

during hatching and kill the emerging chick (Spendelow 1982; Saliva 2000). Since the time from pipping to hatching is about 24 hours, a large fire ant infestation could eliminate successful hatching (Pierce 2001).

Laughing gulls were the primary predators of roseate tern chicks on Pelican Shoal, Florida, during the 2000 breeding season (Zambrano 2001). The exact number of chicks predated during the nesting season is unknown. However, during 21 days of observations from the blind at various times of the day, gulls took 8 chicks out of 12 attempts. During one single observation period, two gulls predated 4 chicks in less than 30 minutes. Most of the chicks were estimated to be less than 10 days old (Zambrano 2001). Mobbing by adults varied from none to a few individuals chasing the gulls out of the colony. One to three gulls were usually responsible for attacks on the colony. No behavioral observations were conducted in 2001, and the time spent on the island was too brief to detect avian predation (Zambrano 2001). No laughing gulls were ever observed approaching the roofs of the Marathon Government Center. In addition, there was no evidence of avian or mammalian predation of roseate or least tern chicks on the roof during both years. Ruddy turnstones were observed at the ground and roof colony and are most likely responsible for 9 eggs found pierced at Marathon. Hermit crabs were observed preying on abandoned eggs at Pelican Shoal (Zambrano 2001).

In 1994, abandonment of the Saba Island (U.S. Virgin Islands) colony occurred because of red-tailed hawk predation on adults and/or laughing gull predation on eggs (Pierce 1996). Numerous broken eggs were found in the colony, and hawks were flushed twice from a tree near the colony. This colony apparently moved to Kalkun Cay, as that colony increased in size shortly after the Saba Island colony was abandoned (Pierce 1996). The Shark Island colony was abandoned in 1995 and again in 1996, though in both years the colony looked normal during the egg counts (Pierce 1996). Something occurred either during late incubation or early egg hatching that caused complete egg failure, but visits to Shark Island at this time provided no evidence as to the cause (no bodies of dead chicks were found). Shark Island is located near a heavily used tourist area, and, although signs are posted around the perimeter of the island, human disturbance is probable (Pierce 1996). The widely distributed laughing gull routinely feeds on food discarded by humans and thus benefits from human activities; and because gulls of this species often prey on the eggs and young of tern species, including the roseate tern, their burgeoning populations are of concern (Pierce 2009). In the U.S. Virgin Islands, roseate terns can be found in both monospecific colonies or in mixed species colonies with other terns or laughing gulls (Pierce 2001). The laughing gull is the only gull that nests in the region. Gull populations have been artificially increased in many areas, creating conflicts between



terns and gulls. The uncovered garbage at the St. Thomas landfill have given gulls unlimited food supplies, causing increased predation pressure upon seabird colonies (Pierce 2001). On Pelican Island, the proximity of a large and aggressive laughing gull colony appears to be the major factor affecting roseate tern nest success (Pierce 2001).

The Flat Cay (U.S. Virgin Islands) roseate tern colony was unsuccessful in 2000, when most breeding roseate terns apparently abandoned the island sometime during the late incubation or early chick-rearing period (Pierce 2001). Only two chicks fledged from this colony. The reason for the abandonment is unknown, but the terns have historically not done well on Flat Cay. A possible explanation is that yellow-crowned night herons depredated the eggs or young chicks (Pierce 2001). Night herons are often seen roosting on the rocks at Flat Cay. The other tern species that nest on Flat Cay do so under the cover of thick vegetation (sooty terns and laughing gulls) or hide their nests (noddy, (*Anous stolidus*), and bridled terns), apparently affording some protection from marauding night herons (Saliva and Burger 1989). In the U.S. Virgin Islands, with the exception of rats, introduced mammals are not a major threat to seabird breeding success. Rats have been eradicated on several cays where roseate terns nest, and efforts are ongoing to continue eradication at several others (Pierce 2009).

In 1998, roseate terns nested for the first time in rat-infested Congo Cay (273 nests), but all the eggs were promptly depredated by rats within three days (Pierce 2009).

Large roseate tern colonies tend to be more successful than smaller colonies, likely as a result of more effective mobbing behavior against aerial predators (Pierce 2009). Gull predation was intense at LeDuck Island throughout the breeding season in 2001, but roseate terns were successful at repelling these attacks early in the breeding season, due to their large numbers and synchronous breeding behavior (Pierce 2001). Hundreds of terns, acting as a cohesive social unit, mobbed gulls. Once chicks hatched and migrated away from nesting areas, gulls were much more successful at depredating eggs and chicks. This was because defense of the colony was no longer a cohesive effort, and late breeding terns were affected in this case by reduced group defense response to gull attacks (Pierce 2001). The availability of suitable hiding places close to the nest site may influence nest success by decreasing the chance of predation.

At Culebra archipelago colonies, predation appears to be a major cause of egg and chick mortality (Shealer and Burger 1992). Shealer (1993) identified five species of potential avian predators at the Cayo Ratón colony at Culebra in 1990, but of those only the laughing gull was seen in the act of capturing chicks. Laughing gulls occasionally visit the La

Parguera terneries, but have never been seen to be successful in capturing either eggs or chicks, as terns respond to them vigorously and drive them away. Roseate terns have been observed mobbing and diving on various heron species, but no predation by any heron on tern eggs or chicks has ever been seen there (Shealer 1993).

Colony abandonment is very common among roseate terns in southwestern Puerto Rico, and they may switch colony locations within a breeding season (Saliva 2009). Colony switching may be linked to over washing of nests by heavy surf and tropical storms, and egg predation by ruddy turnstones and American oystercatchers. Ruddy turnstones constantly patrol colonies, and working in pairs or trios, opportunistically harass adult roseate terns to break and eat eggs (Saliva 2009). The larger American oystercatchers, undeterred by mobbing or incubating adult terns, walk through the colony and snatch eggs at leisure. Additionally, since American oystercatchers nest on these cays, their presence among nesting roseate terns may result in predation of eggs and young, disruption of incubation, and colony abandonment (Saliva 2009).

The major predator of roseate terns at La Parguera seems to be the ruddy turnstone, which preys primarily on sandwich tern eggs (Shealer 1993). However, in 1992, turnstones destroyed a colony of 154 roseate and 112 sandwich tern nests on Cayo Turrumote. Shealer (1993) has observed groups of up to eight turnstones surround an incubating tern and force it off its nest, then prey on the eggs. Douglas (2000) suggested that predation was the most important cause of roseate tern nest failure in southwestern Puerto Rico, and shorebirds were the most important predators. Ruddy turnstones were the most persistent egg predator, which scavenged around the perimeter of the colonies, moving frequently between roseate nests and the shoreline, and monitored individual nests; returning repeatedly to the same nests and checking its status. They walked almost to within reach of the incubating tern and attempted to peer underneath at the nest contents (Douglas 2000). During the heat of the day, some roseate terns vacated nests temporarily to cool off by dipping in the ocean, or to join foraging flocks near the coastline in the evening. It was these temporary abandonment that made the nests most vulnerable. Sometimes attacks occurred very rapidly, within less than a minute of the adult's departure, and the incubating adult returned within less than 2 minutes to find its egg punctured. Ruddy turnstones punctured eggs at Media Luna East and were frequently mobbed by roseate and sandwich terns (Douglas 2000).

A small number of ruddy turnstones (5 to 7) could wreak havoc in a short time, and they were most successful at predating nests at the perimeters of the colony and less successful at predating nests in the center of the colony (Douglas 2000). A ruddy turnstone attempting to walk within the colony

was assaulted from all sides. Thus, a nest with neighbors was defended by several roseate terns. When one of these nests was vacated temporarily, neighbors drove off ruddy turnstones that attempted to predate the nest. However, a nest without neighbors might only be defended by the incubating adult (Douglas 2000). A linear arrangement of nests exposed roseate terns to greater predation pressure than a clustered arrangement (Douglas 2000). This was because ruddy turnstones worked from the edges of the colony inward, and a linear arrangement of nests had the effect of placing every nest on the edge of the colony. By contrast, as many as four neighbors defended nests in a clustered arrangement (Douglas 2000). Ruddy turnstones that attempted to walk among these nests were attacked from two sides at once. Nests in clustered arrangements were as close as 1 ft. 11 in. apart, placing the ruddy turnstone within easy striking distance of incubating terns. This tended to discourage the incursions of ruddy turnstones and reduced their success at taking eggs (Douglas 2000).

Douglas (2000) found that fledging success differed between colony sites and between colony areas at southwestern Puerto Rico colonies in 1999. The Media Luna East colony failed completely. Not a single chick hatched from the 207 nests (Douglas 2000). Nest survival was highly variable at the Turrumote II colony (plot 1= 28 percent, plot 2= 52 percent, plot 3= > 80 percent). Both plots 1 and 2 were heavily predated, and plot 1 was inundated by storm driven waves. Plot 3 closely fringed a sandwich tern colony, and roseate terns on this plot nested in a clustered configuration and in close proximity to each other. Plot 3 had 93 viable nests on 17 June and no evidence of predated eggs. Predation was the primary cause of nest failures at both colonies. At Turrumote II, predation was linked to failures at 22 percent and 43 percent of nests on plots 1 and 2, respectively. At the Media Luna East colony, predation was documented at 98 nests or 47 percent of the colony. However, the ultimate fate of many chicks and eggs was unknown at both colonies. Eggs disappeared between nest checks, probably carried off by either oystercatchers or laughing gulls. Fifty-five percent of nests at Media Luna East disappeared in this way (Douglas 2000).

#### **2.3.1.7.2 Disease and parasites**

Ramos *et al.* (2001) examined the effect of an infestation by the hard tick *Amblyomma loculosum* on the growth and survival of roseate terns in the Seychelles. Nestling growth, fledging age, and survival seemed to be significantly reduced by only one female tick (Ramos *et al.* 2001). Ramos *et al.* (2001) found a linear growth rate of 4.07 g/day in the Seychelles roseate tern population, compared to chicks infested with ticks (0.24g/day). During a successful breeding season, nestling mortality due to ticks is likely to be additive and may increase in the event of several

consecutive successful breeding seasons. Because roseate terns are very sensitive to variations in food levels, a moderate tick infestation may speed up nestling mortality and have detrimental consequences on breeding success, if a period of food shortage coincides with the peak of tick infestation (Ramos *et al.* 2001).

Ectoparasite infestation has been suggested to explain the pattern of colony movement for roseate terns in the Caribbean, but evidence to support this hypothesis is lacking. The best evidence to date is that colony site selection is a response to proximity of a good food supply (Hoffman *et al.* 1993, Pierce 2001).

Based on the above, disease or predation should be considered a current threat to the species. The magnitude of threat of disease and predation on the Caribbean roseate tern is high, and the immediacy of threat to the species is imminent, because predation by avian predators, ants, rats, and crabs results in colony abandonment or significant mortality of egg, chicks, and adult roseate terns. In addition, there is evidence that ectoparasite infestations may result in significant mortality of young roseate terns.

#### **2.3.1.7.3 Post-fledging dispersal, staging, migration and winter quarters**

To assess whether or not the decline in the Caribbean population of roseate terns might be an artifact of birds moving between colonies from year to year, a color-banding program was initiated in 1991 for roseate terns breeding in Puerto Rico, on cays off La Parguera and at Cayo Molinos, and in the U.S. Virgin Islands (2001). The USFWS led the color-banding scheme, which was part of a cooperative study among several northeastern U.S. breeding colonies in an attempt to monitor patterns in annual movement (Pierce 2001). Between 1991 and 1993, 356 adult roseate terns were color-banded in Puerto Rico, and of the 233 adults banded in 1991 and 1992, 78 (33.5 percent) were re-sighted or recaptured at breeding colonies (Shealer 1993); see CA section 2.3.1.2.4. Seven of the 78 birds re-sighted were missing one color-band, resulting in an average yearly rate of band loss of 9.0 percent. Most re-sightings of color-banded birds were at the original colony of banding. However, one bird banded on May 22, 1992, at Parguera was re-trapped on a nest at Cayo Molinos, Culebra, on June 9, 1993 (Shealer 1993). A second bird banded on June 9, 1992, at Cayo Media Luna, Parguera, was found dead on June 3, 1993, at Cayo Turrumote II, Parguera (Shealer 1993). A color-banded adult was seen on June 23, 1993, at Green Cay, near Jost Van Dyke, British Virgin Islands; this bird was banded in 1992 at a colony in St. Thomas (Pierce, DPNR, pers. comm. 1993). On June 6 and 11, 1992, Shealer (1993) observed a color-banded juvenile roseate tern at Cayo

Turumote that had been banded as a chick on June 26, 1991, at LeDuck Island, St. John. Several banded roseate terns have been recaptured whose origins were unknown at the time of recapture; most consisted of chicks banded at Parguera colonies by J. Colón that were recaptured as adults breeding at the same location, as well as one bird trapped on the nest at Parguera was banded in 1983 at St. Thomas (Shealer 1993) (this 10-year-old bird is the oldest yet recovered for the Caribbean population). Preliminary evidence suggests, however, that although Caribbean roseate terns frequently shift colony sites within a specific breeding location, they are extremely area-tenacious in their choice of a breeding location from year to year. Although a few birds do move (Shealer 1993), only 3 of the 10 nesting islands used by roseate terns in 1993 were used again in 1994, a 70 percent colony-site turnover rate (Shealer 1994).

Adult roseate terns led juveniles on training flights and foraging trips from the Turumote II colony in La Parguera (Douglas 2000). Typically, the adult flew high above the sea surface (30 to 60 ft). The juvenile followed 15 to 20 ft behind the adult, remaining close to the sea surface. Adults and chicks maintained vocal contact during the entire foraging trip. If a juvenile lost contact with its adult, it returned to the colony (Douglas 2000). Adult roseate terns were observed leading juveniles away from the breeding colony at Turumote II towards a small sand bar, where adults and juveniles were consistently recorded loafing and roosting. Many roseate tern adults were observed carrying fish to this sand bar to feed juveniles, and leading juveniles on foraging trips from the sand bar (Douglas 2000). Adult roseate terns continued to feed and care for juveniles after they departed the breeding colony, and at least up until the date when they departed southwestern Puerto Rico in early August (Douglas 2000). At La Parguera, the rate of kleptoparasitism was high (Douglas 2000), but at LeDuck even attempts at kleptoparasitism were rarely observed in 2001 (Douglas 2001). This suggests that chick food provisioning may have been better at LeDuck. Alternatively, the physical settings of the two sites may promote and deter kleptoparasitism: La Parguera has low cobble cays that are relatively open and exposed, whereas LeDuck is many times larger than the Turumote II colony, has higher relief above the sea, and has grassy hillsides (Douglas 2001).

Hays *et al.* (1999) reported recoveries of banded roseate and common terns netted at Mangue Seco, Bahia, Brazil. The proportions of roseate and common terns recovered at Mangue Seco from States in the United States are what one would expect given the numbers of banded birds estimated to have survived (Hays *et al.* 1999). In contrast, a significantly higher than expected number of Caribbean roseate terns were recovered at Mangue Seco. I. Nisbet and J. Spendelov (unpubl. data) estimated that by 1999, 65 percent of the U.S. population of roseate terns was banded, but only 13 percent of the roseate terns netted at Mangue Seco (December

1996 to February 1997) were banded (Hays *et al.* 1999). Mangue Seco appears to be a more important area for birds from the Caribbean than for those from the Northeast population, and, therefore, there are probably additional sites where roseate terns from the Northeast, and possibly the Caribbean, occur along the South American coast (Hays *et al.* 1999). The finding that roseate terns from the Caribbean and Northeast populations are found together during the non-breeding season raises the possibility that birds from one population might accompany birds from the other to their breeding grounds. Interestingly, four birds wearing color band combinations put on in the Caribbean have been reported in the United States (Hays *et al.* 1999). Three were adults banded in southwestern Puerto Rico: one banded June 8, 1991, was observed by J. Zingo on Falkner Island, Connecticut, on July 31, 1993; two others banded May 22, 1992, and June 11, 1992, were observed by Hays on Great Gull Island, New York, on May 22, 1997, and May 18, 1994, respectively. A fourth bird banded as a hatching year bird at LeDuck, St. Johns, U.S. Virgin Islands, on June 26, 1991, was observed by Hays on Great Gull Island May 18, 1994 (Hays *et al.* 1999). However, as noted in Northeast section 2.3.1.3, all these sightings were made by reading bands by telescope and have not been confirmed by trapping despite intensive trapping of the Northeast breeding population (Spendelov *et al.* 2008). Although several of these birds were reported within nesting areas, none was actually seen at a nest. Hence, the extent of interchange of breeding individuals among the populations, if any, remains conjectural.

The color-banding program has yielded information on regional inter-colony movement and wintering distribution of subpopulations of Caribbean roseate terns. Re-sightings of banded birds in the nesting colonies have confirmed that movement of birds between southwest Puerto Rico and Culebra, the U.S. Virgin Islands, and the British Virgin Islands exists from one breeding season to the next (Pierce 2001, Shealer *et al.* 2005a). Banded birds have been documented switching from Shark Island (U.S. Virgin Islands) to La Parguera (Puerto Rico), from La Parguera to LeDuck Island (U.S. Virgin Islands), from Flanagan Island (U.S. Virgin Islands) to Green Cay (British Virgin Islands), from Culebra to Green Cay, and from LeDuck Island, Rata Cay, Shark Island (all U.S. Virgin Islands) to Green Cay (British Virgin Islands) (Pierce 2001). Nisbet (1984) suggested that Caribbean and northeastern birds mingle throughout the winter anywhere from Guyana to Brazil, where most banded recoveries have been recorded. Banding data have revealed that at least some of the Virgin Islands and Puerto Rico roseate terns spend the non-breeding season near Mangue Seco, Brazil. The recoveries in Brazil of roseate terns from the Northeast and Caribbean populations show the two populations mix to some degree during the non-breeding season (Table 2) (Hays *et al.* 1999). The fact that all the recovered birds were banded as adults may indicate that young and adult birds go to different

areas in the non-breeding season. In 2001, during banding activities on LeDuck Island, U.S. Virgin Islands, a roseate tern banded by researchers in Mangue Seco, Brazil, during the non-breeding season was trapped on a nest at LeDuck Island (Pierce 2001); highlighting the importance of South America for wintering Caribbean roseate terns.

Overall, the concentration of terns at Mangue Seco includes the largest known gathering of roseate terns along the South American coast and is the first reported area where numbers of birds from the Northeast and Caribbean populations are found together (Hays *et al.* 1999). It is also the first place in the Western Hemisphere where common terns from the Azores have been found during their non-breeding season. Mangue Seco's importance to roseate terns from the New World, as well as common terns from the Azores, is thus clear. The area warrants protection and every effort should be made to prevent development that would adversely affect the terns (Hays *et al.* 1999).

The migratory pathway of Caribbean birds is not known, but the route is almost certain to be 2,000 to 4,000 km (1,243 to 2,485 mi) shorter than the route taken by the northeastern (North America) population (Shealer *et al.* 2005a).

Although scientists believe that the majority of roseate terns in North America and the Caribbean winter in northern South America (Nisbet 1984; Hays *et al.* 1999; Hayes and Bodnar 2009), small flocks may remain near some Caribbean islands. J. Saliva (pers. obs. 1992) reported a flock of about 55 roseate terns and a common tern approximately half a mile off Mary Point, St. John, on October 8, 1992. Most roseate terns in that flock were adults in winter plumage. He also observed a flock of some 65 roseate terns, 1 common tern, and 2 black terns feeding about 3 km (1.86 mi) off the coast of Ponce, Puerto Rico, on October 9, 1992. Roseate terns in that group were mostly adults in winter plumage, although several had tail streamers, bright orange legs, and black bills. In January 1993, J. Saliva (pers. obs. 1993) observed, on several occasions, roseate terns in a mixed group of roosting sandwich terns, brown pelican, snowy egrets, and ruddy turnstones at Mayagüez, Puerto Rico.

#### **2.3.1.7.4 Food and foraging**

In the Caribbean, roseate terns feed on a variety of fish species such as dwarf herring (*Jenkinsia lamprotaenia*), thread herring (*Opisthonema oglinum*), halfbeak (*Hyporhamphus unifasciatus*), young mackerel, small squid, ballyhoo (*Hemiramphus brasiliensis*), Atlantic silversides (*Menidia menidia*), Spanish sardines (*Sardinella aurita*), bigeyed scad (*Selar crumenophthalmus*), Atlantic flying fish (*Cypsedurus melanurus*), and Atlantic bumper (*Chloroscombrus chrysurus*) (USFWS 1993, Hoffman *et*

al. 1993, Zambrano 2001), usually when predatory species such as jacks (*Caranx* sp.), tarpon (*Megalops atlanticus*), and bonito (*Sarda sarda*) chase prey fish near the sea surface (J. Saliva, pers. obs.). Local abundance of small schooling marine fish may vary from year to year, and roseate terns seem to be attracted to areas of peak prey abundance (Pierce 2001). Adults feeding chicks do not regurgitate, but return to the colony carrying a single fish in their beaks. Therefore, proximity and abundance of prey during the chick-rearing period may enhance chick survival (Pierce 2001).

In Puerto Rico, roseate terns commonly feed over schools of predatory fish that drive prey fish to the surface (Shealer and Burger 1993), a situation that creates an ephemeral and unpredictable food source. Thus, it is critical for a young roseate tern to learn quickly to find scattered food patches (Shealer and Burger 1995). In general, adult roseate and sandwich terns foraging at a lagoon in Cayo Turrumote, southwestern Puerto Rico, were more proficient in prey capture than the 1-year-old birds (Shealer and Burger 1995). One-year-old roseate terns caught fewer fish per attempt than the other three age classes, whereas both 1-year-old roseate and sandwich terns caught fewer fish per unit time than their respective adults (Shealer and Burger 1995). One-year-old terns of both species also required more wing beats per fish captured than adults. One-year-old sandwich terns were 90 percent as successful as adults were, but 1-year-old roseate terns were only 79 percent as successful as adults of the species (Shealer and Burger 1995). Young sandwich terns may develop skills in capturing and handling prey before prey locating abilities are perfected, while roseate terns may first learn to locate prey and then gradually improve capture and handling techniques (Shealer and Burger 1995).

Shealer and Burger (1995) and Shealer (1998) studies on the diet of roseate and sandwich terns in southwestern Puerto Rico indicate that the primary prey base for adults of both species consisted of only two species of fish (dwarf herring and sardines). These two species comprised over 65 percent of the prey items delivered by each tern species to chicks in all 3 years of this study (1991 to 1993). Despite this high degree of specialization, roseate and sandwich terns fed on prey in different proportions, and indices of feeding overlap were low prior to chick hatching (Shealer 1998). The diets of adult terns consisted of prey items in different proportions than were fed to chicks, and the sizes of prey fed to chicks increased consistently as chicks aged, although there were yearly differences in the magnitude of this trend (Shealer 1998). Although yearly and colony-site differences existed, the diversity of prey items roseate terns feed to chicks did not seem to differ substantially between temperate and tropical colonies (Shealer 1998).



Shealer and Burger (1995) and Shealer (1998) found that prey delivered to chicks appeared to be more diverse than prey eaten by adult roseate terns, and possibly sandwich terns. In 1992, the diversity of prey fed to roseate tern chicks in Puerto Rico was extremely low at 1.74. In 1992, both roseate and sandwich terns relied much more heavily on dwarf herring and less heavily on sardines, suggesting that sardines were less available around the local area than in 1991 or 1993 (Shealer 1998). Roseate terns in 1992 experienced a dismal breeding season: 23 percent fewer birds attempted to breed compared to 1991, mean clutch size was the lowest ever recorded for this population, and reproductive success averaged only 0.26 chicks/ pair (Shealer 1998). Sandwich terns in 1992 were not as drastically affected by the apparent reduction in sardines. They delivered to chicks a higher diversity of prey in 1992 than in 1991, and experienced moderate reproductive success. These findings suggest that sandwich terns were able to adjust to depletion in their preferred prey, but that roseate terns were less able to use alternative prey resources (Shealer 1998).

Shealer and Burger (1995) and Shealer (1996) found evidence that roseate terns specialized on a biotic factor, namely the presence of predatory fish, to make prey available to them. Despite the high capture success in inshore shallows, roseate terns foraged primarily in blue-water habitats as opposed to shallow lagoons. Clearly, this was because mean food capture rate for roseate terns in blue-water flocks over predatory fish was four times as high as in any other situation (Shealer 1996). Even though capture success was low and the fish were smaller in blue-water habitats, profitability (mean mass of fish ingested per unit time) to roseate terns was high. Thus, foraging roseate terns may rely heavily on predatory fish in the Caribbean (Shealer 1996). Roseate terns experience a much higher prey-capture rate when foraging over predatory fish than all other situations, and inshore abundance of larger predatory fishes may coincide with the terns' breeding season. Since 1991, the first predatory fish activity around the roseate tern colonies was between May 15 and 25 (Shealer 1996), in close parallel to laying of first eggs of the season for roseate terns. Therefore, there may be a causal relationship between the inshore arrival of predatory fishes and the onset of egg laying in terns. However, additional years of data are needed to confirm such a relationship (Shealer 1996). Predatory fish themselves have been shown to depress foraging success of roseate terns in mixed flocks (Shealer 1996), and Ramos (2000) suggested that the single most important foraging strategy for roseate terns and other inshore feeding species on Aride Island, Seychelles, appears to be in association with predatory fish. Breeding failures and periods of low chick food delivery were connected with the absence of predatory fish and small loose feeding flocks (Ramos 2000). Ramos (2000) stated that, in order to assist the conservation of roseate terns on Aride Island, information on the ecology and movement

patterns of predatory fish are needed, because the relative importance of predatory fish versus other factors such as prey recruitment and oceanographic characteristics for foraging roseate terns is unknown.

#### **2.3.1.7.5 Human disturbance**

Douglas (2000) conducted a simple test to determine if roseate tern trapping and banding activities would result in nest abandonment. A sample of 38 roseate tern nests was monitored, where eggs were set up on their blunt end to indicate that the nests had already been trapped, and 28 of these nests hatched (Douglas 2000). The average date of trapping (for those nests that hatched) was the 19<sup>th</sup> day of incubation, and all except one of these nests was trapped after the 17<sup>th</sup> day of incubation. The one exception was a nest trapped on the 13<sup>th</sup> day of incubation. A few of the nests were abandoned because of trapping activities, since eggs were found in the same position days after the nests had been trapped. Clearly, no incubating adult returned to settle on these nests (Douglas 2000). However, most trapped adults returned to tending their nests.

Social attraction techniques and stronger colony protection are being implemented with limited success at roseate tern colonies in the Florida Keys, but this may not be enough to prevent the complete extirpation of this species from Florida (R. Zambrano, pers. comm. 2010).

### **2.3.2 Five-factor analysis:**

The five-factor analysis presented in this section is limited to the threats within the jurisdiction of the United States. The following paragraph summarizes what little information is available regarding threats to roseate terns outside the United States.

#### **Threats in other countries**

The available data on roseate tern populations throughout the Caribbean indicate that this species is primarily threatened by human disturbance, egg collection, and predation by introduced mammals (see Appendix A, section A.2). Although many Caribbean islands have laws to protect wildlife, including roseate terns, enforcement of these laws is practically absent, and all but one location (St. Kitts and Nevis) report egg collection as one of the most important causes of colony failure and mortality.

##### **2.3.2.1 Factor A. Present or threatened destruction, modification or curtailment of its habitat or range:**

At present time, we are not aware that present or threatened destruction, modification, or curtailment of the Caribbean roseate tern habitat or range is currently occurring in areas under U.S. jurisdiction. Sites of roseate tern breeding colonies on Puerto Rico and the U.S. Virgin Islands are publicly-owned and designated for conservation. Roseate terns nesting at Long Key, Dry Tortugas,

are administered by the NPS, and USFWS staff posted “No trespassing” signs, erected symbolic fencing in roseate tern nesting areas, and installed buoys on Wilma Key, Key West NWR, in order to prevent disturbance and encourage tern nesting (Wilmers and Lyons 2008). At Marathon Government Center in the Florida Keys, roseate terns nest in artificial substrate on a building rooftop, but we are not aware of plans to modify the rooftop materials or layout of appliances.

Sea level rise and hurricanes/storms are discussed under Factor E-Other natural or manmade factors. An excess of plant growth (either exotic or native species) does not appear to be a factor affecting roseate nesting activity. Most Caribbean colonies have little or sparse vegetation, and during rainy years when certain herbs survive the dry conditions at the colony sites, roseate terns nest under their shade. Therefore, we believe that this factor is not a current threat for the Caribbean roseate tern in these areas.

#### **2.3.2.2 Factor B. Overutilization for commercial, recreational, scientific, or educational purposes:**

In this section, we limit the discussion to overutilization for commercial purposes, such as selling large numbers of birds or eggs at market, and not local or subsistence consumption. Egging for local consumption is discussed in Factor E.

The 1987 final listing rule stated that a major cause for the declining number of roseate terns since the 1950s may be the trapping and netting of wintering terns for human consumption along the northeastern coast of South America (Nisbet 1984). Because Caribbean roseate terns are thought to winter in the same area as birds from the Northeast population, they would be similarly at risk from such activities; however, there is little information about the current level of trapping and netting in the wintering areas. Although some terns (predominantly common terns) are known to have been taken (or found dead) by Brazilian fisherman, it is not known if the practice is widespread enough to have population-level effects.

Overutilization is not known to be a threat to the species on either the breeding or wintering grounds, but the ecology and threats to the species in the wintering grounds remain poorly studied (Hays *et al.* 1999).

At the present time, we are not aware that overutilization of this species for commercial, recreational (*e.g.*, hunting), scientific, or educational purposes has occurred, or is currently occurring. This factor does not appear to be a current threat for the Caribbean roseate tern.

#### **2.3.2.3 Factor C. Disease or predation:**

There is no direct evidence that Caribbean roseate terns are significantly affected by disease or parasitism. However, it has been reported recently that roseate terns at another tropical site, the Seychelles, are seriously affected by tick parasitism (Ramos *et al.* 2001; see CA section 2.3.1.7.2). This might be a significant threat in other tropical areas, including the Caribbean, and deserves further study.

Predation is known to be a major factor affecting reproductive success in many Caribbean colonies (see CA section 2.3.1.7.1). Important predators include laughing gulls, peregrine falcons, red-tailed hawks, ruddy turnstones, rats, and land crabs. These predators commonly prey on roseate tern eggs and chicks, and the observations summarized in CA section 2.3.1.7.1 suggest that together they reduce the average reproductive success of Caribbean roseate terns to well below the level that could be achieved in the absence of predation. Predation also appears to be responsible for many of the shifts among sites that are characteristic of Caribbean roseate terns. Although there is little evidence that the frequency or intensity of predation are increasing or have been exacerbated by human activity (except as noted for laughing gulls in the U.S. Virgin Islands), there is a clear opportunity for more intensive predator management to effect a considerable increase in average reproductive success.

Disease (ectoparasitism) and predation are considered current threats to the species. The magnitude of threat of ectoparasitism and predation on the Caribbean roseate tern is high, and the immediacy of threat to the species is imminent, because predation by avian predators, ants, rats, and crabs results in colony abandonment or significant mortality of eggs, chicks, and adult roseate terns. In addition, there is evidence that ectoparasite infestations may result in significant mortality of young roseate terns.

#### **2.3.2.4 Factor D. Inadequacy of existing regulatory mechanisms:**

This factor focuses on adequacy of non-ESA regulatory mechanisms *if* the protections of the ESA were to be removed. Federal protection is offered to the Caribbean roseate tern under the MBTA (50 CFR Part 21). According to this statute, migratory birds, their parts, nests, or eggs may not be possessed, imported, exported, bartered, and offered for sale, purchase, or barter without a valid permit issued pursuant to the provisions of the MBTA.

In Florida, the roseate tern is designated by the FFWCC as State threatened. FFWCC also designated Pelican Shoal, a small island off Boca Chica Key, a “Critical Wildlife Area” on July 13, 1990, providing maximum protection to the colony under Florida State law and allowing the site to be closed to human activities between May 1 and August 31. In addition, roseate terns nesting on

Federal lands in Florida, such as Long Key, Dry Tortugas, and Wilma Key, Key West NWR (Wilmer and Lyons 2008), are protected irrespective through agency regulations irrespective of the ESA.

In 1999, the Commonwealth of Puerto Rico approved Law #241 known as the “Nueva Ley de Vida Silvestre de Puerto Rico” (New Wildlife Law of Puerto Rico). The purpose of this law is to protect, conserve, and enhance both native and migratory wildlife species within its jurisdiction, regulate permits, regulate hunting activities, and regulate exotic species, among others. In 2004, the Puerto Rico Department of Natural and Environmental Resources approved the “Reglamento para Regir el Manejo de las Especies Vulnerables y en Peligro de Extinción en el Estado Libre Asociado de Puerto Rico” (Regulation 6766: To regulate the management of threatened and endangered species in Puerto Rico). The Caribbean roseate tern is included in this list of protected species and designated as “vulnerable”. Under this regulation, Article 2.06 prohibits collecting, cutting, removing, among other activities, listed animals and plants within the jurisdiction of Puerto Rico. Additionally, the roseate tern is listed as threatened by the U.S. Virgin Island Territorial Government under Code, Title 12 – Chapter 2; Protection of Indigenous, Endangered and Threatened Fish, Wildlife and Plants of the Endangered and Indigenous Species Act of 1990. The purpose of this Chapter is to protect, conserve and manage indigenous fish, wildlife and plants, and endangered or threatened species for the ultimate benefit of all Virgin Islanders, now and in the future. Section 105 of this Chapter prohibits the harassment, injury or killing, or the attempt to do the same, or sell or offer for sale any specimen, or parts or products of an endangered or threatened species.

Based on: (1) The presence of Federal, State, Commonwealth, and Territorial laws and regulations protecting the Caribbean roseate tern; (2) the absence of evidence indicating any lack of enforcement of these laws and regulations to protect this species; and (3) the absence of nesting colonies in North Carolina, South Carolina, and Georgia, inadequacy of existing regulatory mechanisms does not appear to be a threat to the Caribbean roseate tern population.

#### **2.3.2.5 Factor E. Other natural or manmade factors affecting its continued existence:**

##### **Disturbance and predation by humans**

Many tern species breed along coasts and inland waterways, and they have thus had a long history of interactions with man (Del Hoyo *et al.* 1996). Over much of this mutual history, tern eggs, chicks, and adults have been exploited for food, fashion, and information. Throughout history, bird eggs have provided an easily accessible, high-protein, low-cost food source; and exploitation of tern eggs has been recorded for most species, and for most areas of the world (Del Hoyo *et al.* 1996). Even today, and despite legal restrictions, eggging is a problem for many tern species, because colonies are often in places that are hard to monitor and eggs

are often highly prized. People in the West Indies, who collect their eggs while those of species that are more common are ignored, attribute aphrodisiac qualities to roseate and sooty tern eggs (Del Hoyo *et al.* 1996).

The simultaneous decline of the roseate tern in Europe and North America is at least partially attributable to harvesting on the wintering grounds in West Africa and northern South America, where large numbers of birds are taken (Del Hoyo *et al.* 1996). In tropical regions, the conservation efforts are at an early stage, for eggging is still widespread, even where law ostensibly protects terns (Del Hoyo *et al.* 1996). Human residential, commercial, and recreational activities in proximity to roseate tern colonies are a significant source of disturbance to breeding terns (Saliva 2000). Although terns can habituate to some human disturbance, it does nonetheless cause chicks to run from nesting ledges or may keep adults off their nests, allowing predators to steal eggs or exposing eggs to lethal temperatures (Saliva 2000).

Human disturbance at colony sites continues to threaten roseate tern nesting populations, primarily in Puerto Rico and Florida. Limited and declining nesting habitat, tropical storms, and human disturbance have contributed to the precipitous decline of the roseate tern in Florida (Hughes, pers. comm. 2008; R. Zambrano, pers. comm. 2010). Human disturbance, such as camping, has been a problem at Pelican Shoal (USFWS 1999). Boaters and recreational users and their pets disrupted another colony, feral cats preyed upon chicks that fell off roof colonies, and building and air-conditioning repairs on the roof colonies led to nest abandonment (R. Zambrano, pers. comm. 2010). Additionally, remaining white pebble roofs in the Keys continue to be converted to tar or other unsuitable surfaces. Social attraction techniques and stronger colony protection likely ameliorated declines, but efforts may not be enough to prevent the complete extirpation of breeding roseate terns in Florida (R. Zambrano, pers. comm. 2010).

In 2001, at least 100 nests on Pelican Shoal, Florida, were either destroyed directly or were abandoned because of a sea turtle attempting to nest on the island (Zambrano 2001). Based on the tracks, the species of turtle was a loggerhead (*Caretta caretta*) or a hawksbill (*Eretmochelys imbricata*), with the loggerhead being the most common in this area. The sea turtle made several attempts to dig a nest cavity but was unable to due to the hard coral rubble present under the sandy portions of the island (Zambrano 2001). A single sea turtle can nest up to 12 times in one season. This sea turtle emerged five nights during the tern breeding season and caused damage on four of those occasions. Two of these emergences took place during June 11 and 12, 2001, prior to the hatching of the first group of nests of the season (Zambrano 2001).

The more exposed nest sites of Caribbean roseate terns make eggs and young chicks vulnerable to predators when the parents are off the nest, and in the Virgin Islands the nest sites are precariously situated on islands that are located near heavily used tourist areas, or that are subject to predation from other animals

(Pierce 2009). These colonies are readily accessible to illegal egg collecting, as well as the unintentional and purposeful disturbance of breeding colonies, which often results in colony abandonment (Pierce 1996, Pierce 2009). Despite the relative inaccessibility of nesting sites on offshore cays to human and mammalian predators, the breeding seabird populations of the U.S. Virgin Islands remain threatened by a variety of factors. Excessive human disturbance of nesting adults results in overheating of eggs and small chicks, sometimes provoking abandonment by adults (Pierce 2009).

The Division of Fish and Wildlife of the U.S. Virgin Islands government has maintained sanctuary signs on the important seabird cays to limit foot traffic into the seabird colonies and to inform the public of conservation restrictions, but their effectiveness is questionable (Pierce 2009). Another large colony was found in 1997 on LeDuck with 783 nests. After the nesting season was over, it was reported that St. John fishermen had taken "baskets full of eggs" from LeDuck (Pierce 2001). This eggging event on LeDuck explains the greater than 50 percent reduction in colony size that was observed when researchers returned to the island to band chicks (Pierce 2001). Although Federal and local legislation prohibit eggging in the U.S. Virgin Islands, it has been a serious problem for open ground nesters like sooty and roseate terns, and some eggging probably still occurs (Pierce 2009). Eggging events are difficult to resolve because they usually occur in remote colonies with no witnesses and with little evidence left at the colony site (Pierce 2001).

Disturbance and predation by humans appears to be the single most detrimental factor to roseate tern reproductive success in Puerto Rico (Shealer 1993). Human predation occurs in the form of eggging. Coastal towns near roseate tern colonies in southwestern Puerto Rico highly prize roseate tern eggs in particular, due to their relative rarity, and regard them as an aphrodisiac (Shealer 1993). One myth that exists with some of the locals—one that is particularly damaging to roseate terns—is that the first eggs laid by the birds are infertile (Shealer 1993). By taking all the eggs from the first clutch, the locals believe that they are actually helping to promote the propagation of the species. This belief is false, and is, in fact, the opposite of what really occurs (Shealer 1993). The first clutch in birds is the one with the highest viability; roseate terns sometimes will re-lay if the first clutch is lost, but egg fertility declines with subsequent nesting attempts (Shealer 1993).

Eggging was strongly suspected at the roseate tern colony at Cayo Molinos in the Culebra archipelago in 1990 when, after returning to an active roseate tern colony in Cayo Molinos, Shealer (1993) found it deserted and with no signs of dead chicks, broken eggs, or eggshell fragments to suggest animal predation. In 1992, two young men in a boat landed on an offshore resting site for terns near Cayo Turrumote. They waded ashore carrying buckets and chased all the terns away. They then searched the area for a few minutes and, finding nothing, returned to their boat and left (Shealer 1993). In 1993, at Cayo San Cristóbal, poachers apparently took approximately 86 roseate tern eggs from 49 nests on 31 May.

Shealer (1993) inspected the site on June 1 and found empty nests with no signs of disturbance. All the missing eggs were in the same concentrated vicinity. In 1994, human disturbance and destruction at roseate tern colonies in southwestern Puerto Rico continued (Shealer 1994). On June 12, 1994, two boats, each containing two people, arrived on Cayo Turrumote. The boaters appeared to be local anglers, as they were cast netting for baitfish within the inner lagoon close to the roseate tern colony. After 20 minutes, two teenagers in one boat landed on the north side of the island and began walking toward the tern colony. As they walked, one of the boys began picking up rocks and throwing them at the birds circling overhead, as well as at birds that were on the ground incubating eggs. The other boy began picking up eggs and shaking them, then smashing them in his hand or throwing them on the ground and breaking them. This destruction continued for 14 minutes. Shealer (1994) directly observed them destroying six sandwich tern eggs and four roseate tern eggs but suspected that they destroyed many more.

Although eggging of roseate tern nests may still be a problem in Puerto Rico and the Virgin Islands, incidental or intentional harassment or disturbance of nesting terns may be more detrimental (Shealer 1993). On May 21, 1991, Shealer (1993) found at Cayo Turrumote, a historical tern nesting site near Parguera, five people with a reef-drilling rig in the lagoon. The drilling team was unaware of the potential damage they had caused. During the time they were on the reef, Shealer (1993) observed several roseate and sandwich terns circling overhead and giving alarm calls. Terns did not colonize the island in 1991, possibly due to the disturbance early in the season. In 1992, Shealer (1993) lived on Cayo Turrumote from May 20 to July 14. During this period, 68 people in 17 parties landed on the island. Most caused no harm or disturbance to the nesting terns; however, 6 of the 17 (35 percent) parties would have been potential problems had they been allowed to roam the island, because in these six instances, the people were walking toward the tern colony where the terns preferred to nest on the reef crest (Shealer 1993). On May 31, two people walked around the island undisturbed, and after they departed, it was found that they had inadvertently crushed all the eggs in three roseate tern nests (Shealer 1993).

Other potential problems that Shealer (1993) observed were children brandishing sticks and striking out at terns diving upon them, and people standing in the ternery for long periods at a time during hot days, which kept the birds from shading their eggs and chicks.

Regular monitoring of roseate terns nesting on Cayo Turrumote I in 1993 indicated that human visitation was heavier than in 1992 (Shealer 1993). Roseate terns nested late on Turrumote I in 1993, but all 40 nests were destroyed or abandoned after the long Fourth of July weekend holiday. Over half the nests had eggshell fragments and dead embryos, and the rest of the eggs were completely missing from the nests (Shealer 1993).



The trapping and killing of roseate terns and other tern species for food at roosting sites at their wintering quarters has been reported and may be an important component of population regulation in this species (Shealer *et al.* 2005b). The main winter quarters of roseate terns are along the northern coasts of South America extending in the west from the Pacific coast of Colombia to eastern Brazil; primarily Guyana (Nisbet 1984). Roseate terns in these areas have been shot or trapped intensively for sale at the local markets (Nisbet 1984). Nisbet (1984) found a change in the pattern of winter recoveries, associated with intensive trapping of roseate terns for food in a limited area in eastern Guyana. Between 1968 and 1977, about 1.2 percent (118 of 10,037) of the roseate terns banded as chicks or juveniles in the Northeast were recovered in their first year of life in this area (Nisbet 1984). The number of banded adults at risk during this period was about 3,800 (2317 banded as adults between 1965 and 1977, plus about 15 percent of the 9,872 chicks banded between 1965 and 1975). Of these, 68 (1.8 percent) were recovered as adults in Guyana between 1968 and 1977. Thus, trappers in eastern Guyana appear to have trapped about 1 percent of the juveniles and 2 percent of the adults banded at this period (Nisbet 1984). Taken alone, this is insufficient to account for the decline of 30 to 40 percent in the breeding population during this period (Nisbet 1980). However, these figures demonstrate the impact that a single trapper can have on a limited population with concentrated winter quarters, thus indicating the species' vulnerability to systematic trapping in more than a few locations (Nisbet 1984). The extent of roseate tern mortality due to trapping for food in northern South America during the wintering migration is currently unknown and merits further evaluation.

### **Competition**

Competitive interactions between roseate terns and other seabird species foraging upon the same food resource have been suggested as potentially affecting chick food provisioning in roseate terns (Shealer and Burger 1993, Ramos 2000). Shealer and Burger (1993) studied the competitive interaction of roseate terns foraging in mixed-species flocks of noddy terns, bridled terns, and brown boobies (*Sula leucogaster*). The frequency of prey capture attempts by roseate terns showed a significant negative relationship to the number of brown noddies in the flock, while no such relationship was apparent in monospecific flocks of roseate terns (Shealer and Burger 1993). Roseate terns changed foraging strategy (from diving to surface picking), and showed significantly less prey capture attempts, in mixed-species flocks than monospecific flocks (Shealer and Burger 1993). Even though Shealer and Burger (1993) detected a measurable effect of roseate terns foraging in flocks with brown noddies, they did not believe that it represented a significant factor.

### **Kleptoparasitism**

Inter- and intra-specific kleptoparasitism has been reported for tropical roseate terns (Ramos 2000; Douglas 2000). Roseate tern adults must deliver food items

very quickly or risk kleptoparasitism. For example, on June 28, 1999, a roseate tern adult landed with a needlefish and was on the ground for less than 15 seconds when six other adult roseate terns converged and attacked it, attempting to take the fish. Such observations were rare, and most successful food deliveries were accomplished quickly ( $R = 0.68 \pm 0.04$  sec.,  $n = 34$ ) (Douglas 2000). If a food delivery took longer, the threat of kleptoparasitism increased. Rarely could an adult proceed leisurely with longer food deliveries, though one successful delivery extended over 4.6 seconds. If another adult or chick interfered with the feeding, the handoff was aborted, and the adult would fly away from its chick and return to attempt the handoff a second or third time (Douglas 2000). Interference usually increased the time required to make a food delivery ( $5 \leq 34.35 \pm 3.38$  sec.,  $n = 16$ ). Douglas (2000) observed 54 deliveries in which theft was attempted, and it prolonged or interrupted food deliveries in all but four instances. Sometimes the adult would attempt to lead its chick away from the source of interference; and once chicks learned to fly, even over short distances, they would establish contact with their adults by chasing them in flight. The chick would land, and the adult would follow, feeding the chick as it landed (Douglas 2000). Other adult roseate terns attempted to steal food deliveries by aerial pursuit, by landing in front of the chick as a handoff was about to occur or by attempting to swoop down upon the chick as it was handling the food item (Douglas 2000). Other roseate tern chicks attempted to steal food deliveries by converging upon the point of delivery and intercepting the handoff, or by trying to grab the fish while the chick was handling its prey item. Upon completing a delivery, the adult roseate tern frequently attacked the closest chick or adult, chasing it away. This reduced the potential for kleptoparasitism, reduced interference, and facilitated food handling for the chick (Douglas 2000).

### **Wind turbines**

In 2001, the USFWS was contacted by a private developer to evaluate the effect of the proposed construction and operation of 25 1.65 MW wind turbines on Punta Verraco, Cerro Toro, and Punta Ventana in southern Puerto Rico on several listed species, including the roseate tern (USFWS 2006). The USFWS concluded that the project would not result in direct or indirect effects to nesting, roosting, and feeding grounds of roseate terns, and only two roseate terns are expected to be injured or killed in 40 years of the project operation (USFWS 2006).

### **Climate change and stochastic events**

Climatic factors determining the availability of food sources affect the fledging success and survival of roseate terns. Underwater sea currents and sea temperatures influence the production and timing of arrival of juvenile fish, the main source of food for young terns near breeding areas. Storm surge and abrupt tidal changes dramatically affect the habitat within low-lying islands where some roseate tern colonies are located, resulting in nest abandonment or failure to nest. One nesting area in southern Puerto Rico was reduced in size in 2004, and

disappeared underwater in 2005. Storms during the 2005 hurricane season reduced the size of the main nesting area in the Florida Keys to about half its original size.

Given the paucity of islands with suitable nesting substrate and minimal human disturbance, the sole population of the Caribbean roseate tern in the continental United States appears dangerously vulnerable to any one of a number of human-induced or natural stochastic events (Zambrano 2001). For instance, the lowered breeding number in 1999 might have been a result of Hurricane Georges, which passed directly over Pelican Shoal the previous year (after the breeding season); the hurricane essentially reduced the nesting substrate by half. Subsequent to Hurricane Georges, there has been further erosion of the island from indirect hurricanes and strong winter storms (Zambrano 2001). The small surface area of Pelican Shoal and its low relief appear to make nests of roseate terns very vulnerable to being washed away by extreme high tides and storms. Tropical waves and massive storms (*e.g.*, hurricanes) are quite common during the summer months, and waterspouts are also prevalent in the Florida Keys during the breeding season (Zambrano 2001).

*Summary.* We believe that the magnitude of threat from other natural or manmade factors to the Caribbean roseate tern is high, and the immediacy of threat is imminent. Human recreational activities in proximity to roseate tern colonies and visitation to breeding colonies are a significant source of disturbance to breeding terns, and egg collection continues to be a major source of egg loss and colony desertion in many Caribbean roseate tern colonies. Intentional and un-intentional harassment of nesting roseate terns by tourists or anglers results in significant egg or chick mortality and colony abandonment. Hurricanes and storms, Inter- and intra-specific competition for nesting areas or food resources, and indirect effects of urbanization and development also threaten this species. In addition, the expected sea level rise and temperature increase from climate change is likely to result in significant reduction or elimination of low-lying nesting areas and changes in fish abundance and distribution (both prey and predatory fish species).

## **2.4 Synthesis: Caribbean population**

Roseate terns have a nearly worldwide nesting distribution. South of New Jersey, United States, roseate terns have been reported in the States of Georgia and Florida. In the Caribbean, nesting colonies have been reported on the Bahamas Islands; Bermuda Islands; Greater Antilles (Cuba, Hispaniola (Dominican Republic and Haiti), Jamaica, and Puerto Rico); Lesser Antilles (Anguilla, British Virgin Islands, French Antilles (Guadeloupe and outer islands, St. Martin, and St. Bartholomew), Grenadines, Grenada, Martinique, U.S. Virgin Islands (St. John and St. Thomas), St. Kitts and Nevis, St. Lucia, and St. Vincent); Turks and Caicos islands; and islands in the southern Caribbean [Netherland Antilles (Aruba, Bonaire, Curacao, St. Maarten, Saba, and St. Eustatius) and Trinidad and Tobago].

The Caribbean population of the roseate tern was listed as threatened in 1987, and its recovery plan was approved on September 24, 1993. The USFWS considers the Caribbean roseate tern population in U.S. territories to be improving, because the Florida population and the second largest population in southwestern Puerto Rico have been slowly increasing since 1960 and 1990, respectively, and the largest population in the U.S. Virgin Islands appears to be stable. In contrast, the available data on roseate tern populations throughout the Caribbean (excluding Florida, Puerto Rico, and the Virgin Islands) indicate that most colonies are relatively small, decreasing in size, or abandoning historic sites.

In Florida, Puerto Rico, and the U.S. Virgin Islands, habitat loss or degradation, over-utilization, and inadequacy of existing regulatory mechanisms are not threats to roseate terns. The magnitude of threat of disease and predation on this species is high, and the immediacy of threat to the species is imminent, because predation by avian predators, ants, rats, and crabs results in colony abandonment or significant mortality of egg, chicks, and adult roseate terns. In addition, there is evidence that ectoparasite infestations may result in significant mortality of young roseate terns. Other natural or manmade factors continue to threaten Caribbean roseate tern populations. Human recreational activities in proximity to roseate tern colonies and visits to breeding colonies are a significant source of disturbance to breeding terns, and egg collection for subsistence continues to be a major source of egg loss and colony desertion in many roseate tern colonies. Intentional and unintentional harassment of nesting roseate terns by tourists or anglers results in significant egg or chick mortality and colony abandonment. Hurricanes and storms, inter- and intra-specific competition for nesting areas or food resources, and indirect effects of urbanization and development also threaten this species. The expected sea-level rise and temperature increase from climate change could result in significant reduction or elimination of low-lying nesting areas and changes in fish abundance and distribution (both prey and predatory fish species).

Elsewhere in the Caribbean, egg collection for local food consumption is the main threat to roseate tern populations. Additional threats include development of tourism in coastal regions, collection of unfledged young and adults for food, hunting or fishing for terns with hooks to collect their metal bands, building of causeways to access cays and islands (which facilitate introduction of feral mammals), intentional and unintentional human disturbance, oil pollution, over-fishing, and agricultural development. Based on the above, the Caribbean roseate tern continues to meet the definition of a threatened species, *i.e.*, it is considered likely to become endangered in the foreseeable future throughout all or a significant portion of its range unless recovery efforts continue.

### **3.0 RESULTS–NORTHEAST POPULATION**

**3.1 Recommended Classification:** Retain as endangered; no change in status is warranted. However, since the listing of the “species” (*i.e.*, the population of *S. dougallii dougallii* that breeds in the U.S. and Canadian North Atlantic) pre-dated the 1996 DPS policy, the listing should be revisited.

Rationale: During the past 20+ years, a comprehensive program entailing colony stewardship, nesting habitat enhancement, predator management and population monitoring has been

implemented at all major and most of the smaller breeding colonies. Despite these efforts, the Northeast population has only briefly exceeded 4,000 (peak season) nesting pairs (1999 to 2000), and only three colonies have consistently supported 200 or more roseate nesting pairs. Thus far, the goal of reaching 5,000 breeding pairs distributed among six large colonies has not been met, although the productivity goal of 1.0 chicks per pair appears achievable, as productivity at major colonies often exceeds 1.0 chicks/pair (Nisbet and Spindel 1999, RTRT 2009). Recent data suggest a regionwide reduction in productivity since 2000, for unknown reasons, but productivity at the major colonies has still exceeded 1.0 in most years. Despite good productivity, however, the rangewide population in the Northeast declined 25 percent (1,000 breeding pairs) between 2000 to 2009. The delisting objective (increase population to historic high of 8,500 pairs) clearly has not been achieved. In addition, threats, mainly nesting habitat loss due to erosion, unknown causes of adult and juvenile mortality, and predation on eggs, chicks and adults from a suite of avian and mammalian predators, continue to affect the Northeast population throughout its range.

**3.2 New Recovery Priority Number:** Retain as a priority 3; no change is needed.

**3.3 Listing and Reclassification Priority Number:** 6

Rationale: We acknowledge the merits of clarifying the listing to recognize the Northeast population as a DPS. Priority for formal recognition of this entity (and the Caribbean population) (as described in 48 FR 43098) is 6 on a scale of 1 to 6, indicating that: (1) The proposed change would have low management impact, and (2) the action is not petitioned. Formal recognition of the two units would change little in terms of regulatory impact, as the best scientific information available continues to indicate that both the Northeast and Caribbean populations should remain classified as endangered and threatened, respectively.

### **3.0 RESULTS–CARIBBEAN POPULATION**

**3.1 Recommended Classification:** Retain as threatened; no change in status is warranted. However, since the listing of the “species” (*i.e.*, the population of *S. dougallii dougallii* that breeds in Puerto Rico, Florida, the U.S. Virgin Islands and elsewhere in the Caribbean) pre-dated the 1996 DPS policy, the listing should be revisited.

Rationale: The recovery objectives for reclassification and for delisting have not been met. Regarding the first objective, coordinated programs to maintain, protect, and enhance roseate tern populations in the Caribbean have not been established due to logistical difficulties to convene all countries involved as well as differences in political and governmental procedures to protect natural resources among different governments. In addition, proper mechanisms to protect roseate terns from human and animal predation are lacking in most Caribbean countries, so recreational and commercial activities and egg collection continue largely unabated. Harvesting of large numbers of birds in northern South America also continues to be a factor. The main winter quarters of Puerto Rico and U.S. Virgin Islands roseate terns are along the northern coast of South America, and the extent of roseate tern mortality due to trapping for food in northern South America during the wintering migration is currently not known and merits further evaluation.

Neither has the second recovery objective of maintaining a stable or increasing Caribbean roseate tern population for at least 5 consecutive years been met. In fact, the status of this species in many Caribbean countries is not known; thus, it is not possible to assess whether the Caribbean roseate tern population is increasing or decreasing. Only colonies in Puerto Rico and the U.S. Virgin Islands are visited regularly to obtain actual or estimated breeding population numbers.

**3.2 New Recovery Priority Number:** Retain as 3; no change is needed.

**3.3 Listing and Reclassification Priority Number:** 6 (see Rationale under NE section 3.3)

## **4.0 RECOMMENDATIONS FOR FUTURE ACTIONS**

### **NORTHEAST POPULATION**

1. The many conservation activities on-going for the roseate tern in the Northeast should continue.
  - a. These include the monitoring of all breeding colonies to assess the number of nesting pairs, their hatching success and nesting productivity; the management of competitors and predators that compete with roseate terns for limited nesting space or cause loss of eggs, chicks or adults; enhancing nesting habitat through the placement of artificial nest boxes or other structures; and taking measures to avoid habitat degradation from the incursion of invasive exotic and native plant species that can cover sparsely vegetated nesting areas with rank or dense vegetation that is unsuitable for tern nesting.
  - b. Banding and color-banding studies that add meaningfully to our understanding of meta-population dynamics, individual longevity, nest site fidelity, identification of migration and wintering areas, age at first breeding, juvenile and adult survival rates and other demographic parameters should continue. These studies may provide data essential to our understanding of roseate tern population dynamics.
2. Immediate measures to ensure the viability of nesting habitats at key island colony sites in the Northeast should be taken before erosion and rising sea levels cause further reduction in habitat suitability, carrying capacity and productivity of nesting pairs. A detailed project report and environmental assessment has been prepared for the restoration of Bird Island (USACOE 2005), and alternatives to protect the shoreline of Ram Island have been identified (ACRE 2009), but substantive actions to implement projects are needed.
3. New studies and technologies are needed to identify why the adult breeding population of the roseate tern in the Northeast has declined (2000 to 2009), despite generally good nesting productivity and the absence of significant mortality of adults during most breeding seasons.
4. For all nesting sites in the Northeast and particularly for colonies in the warm water sub region, the vulnerability to sea level rise and over washing by coastal storms should be assessed, and where feasible, plans developed to mitigate further loss of habitat. At other

sites, where there are either natural or man influenced changes to the coastal geomorphology of nesting islands, shoreline protection opportunities may not be feasible or desirable. At these locations, alternative nesting sites nearby should be evaluated and their suitability to support nesting roseate terns through the removal of competing species, habitat improvement or other means should be investigated.

5. Measures to address the features of the revetment at Falkner Island that are inimical to successful nesting and chick rearing by roseate terns should be mitigated. Specifically, the interstitial spaces within the revetment that are in proximity to nesting sites of roseate terns, wherein chicks, adults and fledglings may become trapped, should be filled with crushed stone or other material. Secondly, source material suitable to sustain the spit at the north end of the island should be provided or an alternative means should be identified to avoid the further loss of the tern nesting habitat there.
6. Geolocators have made possible the first technology-based means to track the phenology and general migratory path followed by roseate terns during their annual cycle. Results to date should be evaluated and a determination made whether continuation of this study would provide additional information to benefit the recovery program.
7. A better understanding of the habitats used by roseate terns during the post-breeding staging period should be developed and the factors that are limiting the use of preferred sites should be addressed.
8. Very little is known about the distribution and ecology of roseate terns during migration and wintering, and there is no information of any kind about causes of death. Much more information is needed about factors limiting survival in the winter quarters to allow formulation of effective conservation measures.

## **CARIBBEAN POPULATION**

1. Fire ant control would be an inexpensive, effective way to improve nesting success. Fire ant poison should be broadcast in the nesting areas on these islands prior to arrival of the terns. It would be advisable to formulate plans for this work in consultation with a fire ant specialist and an avian toxicologist to develop a protocol that results in maximum kill of fire ants with a minimum risk of toxicity to the birds. Ideally, this method could be executed up to a month prior to breeding and would effectively depress fire ant parasitism through the end of June.
2. Crab removal and relocation should be considered at colonies with abundant crab populations.
3. Rats should be eradicated from islands used by roseate terns.
4. Laughing gull control should be considered in areas where gulls specialized in egg-robbing are detected, through aversive conditioning or gull colony-level control.

5. Enhance nesting habitat. Roseate terns in the U.S. Virgin Islands seem to prefer open sites with some type of shelter near the nest. Vegetation encroachment in nesting areas should be controlled by removing excessive vegetation cover. Artificial nest shelters can be provided in those areas where natural shelter is not available. The number of shelters provided would depend on the number of nesting terns, the location of nests, and availability of natural shelter.
6. A regional approach to conduct annual surveys should be implemented. Because of the yearly shifting of colonies, monitoring roseate terns requires a regional approach. Simultaneous annual nest counts in the U.S. Virgin Islands, British Virgin Islands, and Puerto Rico should be conducted to accurately establish the population status of this species for the Puerto Rico Bank. Communication and cooperation between researchers in this region is essential to detect the population trends for this species.
7. Continued banding increases the likelihood of recoveries of banded birds. Banding provides information about migration, longevity, and factors affecting the species on the wintering grounds. Banding of adults is also necessary to determine the extent, if any, of dispersal and inter-colony movement. Linkages between breeding and wintering sites should promote the establishment of more holistic and efficient management plans.
8. Prevent human disturbance. The posting of breeding areas will alert the public that the area is an endangered species nesting area and that foot traffic is prohibited. The preparation and placing of warning signs to prevent humans from entering the nesting areas or poaching eggs, for instance, is not a law enforcement issue.
9. Recreational use of cays and islands used by roseate terns, particularly in Florida, should be restricted. Islands where potential conflict between human use and tern nesting is expected should be monitored early in the roseate tern-breeding season for potential nesting activity. If nesting activity is strongly suspected or imminent, these islands should be temporarily closed to the public during the incubation period. After chick hatching, the islands may be opened to the public with warning signs posted to prevent human incursion into the colony area.
10. Protect nesting colonies from poaching. The illegal take of eggs by humans is by far the primary limiting factor for roseate terns in the Virgin Islands. Federal and local law enforcement officer patrolling of nesting areas is crucial during the 3-week window when colonies are most vulnerable.
11. Additional research is needed on the genetics of the Caribbean metapopulation, as well as colony-site fidelity, to determine the degree of intermixing between colonies. Population models using estimates of annual productivity and annual adult survival rates, to determine the long-term status of the population, should be considered.
12. Long-term data on annual reproductive success and chick growth rates are needed to properly assess the importance of Florida colonies to the total Caribbean population



13. Roseate terns in the Seychelles are seriously affected by tick parasitism (see section 2.3.1.7.2, above). Ectoparasites could be a significant threat in other tropical areas, including the Caribbean, and deserves further study.
14. The extent of roseate tern mortality due to trapping for food in northern South America during wintering and migration is currently not known, and merits further evaluation.
15. Monitor prey and predatory fish populations in waters off southwestern Puerto Rico between April and August for a period of 10 years. Schools of juvenile fish, followed by predatory fish, appear in southwestern waters each spring. Even if prey fish species are abundant, they are only accessible to roseate terns when predatory fish species drive the smaller fish to the surface and within reach of roseate terns. Therefore, both healthy prey and predatory fish populations are necessary for roseate terns to breed successfully. Fish populations may be affected by overfishing and climatic changes that regulate their population size and distribution. Monitoring these fish populations provide early warning that climatic changes may imminently affect roseate tern recovery.

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**U.S. FISH AND WILDLIFE SERVICE FIVE-YEAR REVIEW**

Species: Roseate tern (*Sterna dougallii*)

**Current Classification for Northeast Population:** Endangered

**Current Classification for Caribbean Population:** Threatened

**Recommendation Resulting from the Five-Year Review:** No change

**Review Conducted by:**

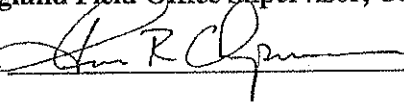
Michael Amaral, New England Field Office

Jorge E. Saliva, PhD., Caribbean Ecological Services Field Office

**FIELD OFFICE APPROVAL:**

New England Field Office Supervisor, U.S. Fish and Wildlife Service

Approve

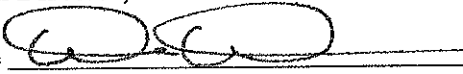


Date 27 September 2010

**REGIONAL OFFICE APPROVAL:**

Regional Director, U.S. Fish and Wildlife Service Region 5

Approve




Date 9/30/10

**REGIONAL OFFICE CONCURRENCE:**

Regional Director, U.S. Fish and Wildlife Service Region 4

Concur

  
Acting ARD-JES

Date 9/29/2010



## APPENDIX A: STATUS, TRENDS AND THREATS TO CARIBBEAN ROSEATE TERNS IN AREAS OUTSIDE U.S. JURISDICTION

### A.1 Status, numbers and trends

Bermuda: The last confirmed nesting of roseate terns in the Bermuda Islands was in the 1840s (Madeiros 2009).

Bahamas: An estimated 560 to 800 pairs of roseate terns nested in the Bahamas Islands at Cay Sal Bank, Great Bahama Bank, Eastern Islands, and Southern Islands during 1994 to 2004 (Hallett 2009) (Fig. C2).

Turks and Caicos Islands: In the Turks and Caicos Islands, the roseate tern is an uncommon summer and winter visitor (Pienkowski 2009) (Fig. C2). Pienkowski (2009) reports an estimated 200 nesting pairs on Fish Cay, and additional breeding activity at Little Sand Cay and Salt Cay (unknown numbers).

Cuba: Roseate terns are considered rare summer residents in Cuba, where the nesting population has been estimated at between 40 to 50 pairs at four breeding sites: La Vela Cay, Hicacos Peninsula, Mono Grande Cay, and Ballenatos Cay (Jiménez *et al.* 2009) (Fig. C2).

Jamaica: In Jamaica, small numbers of roseate terns have been observed roosting or nesting at South Cay, Pelican Cay, Sand Bank Cay, and South West Cay (Haynes-Sutton 2009, pp. 70 and 72) (Fig. C2). The largest count of nesting roseate terns in Jamaica was 70 pairs in the 1960s, but more recent sporadic counts (1997 to 2005) have shown less than five nests and about 18 adult birds (Haynes-Sutton 2009).

Hispaniola: The roseate tern is an uncommon regular spring and fall transient, and former local breeder, in Hispaniola (Keith 2009) (Fig. C2). It is known to have bred at two Dominican Republic localities in 1927 to 1931: Isla Beata and San Lorenzo Cay. However, there are no documented breeding records since that period (Keith 2009). There are 18 known band returns at Hispaniola from sites between Nova Scotia and New York in North America (Keith 2009), suggesting that wintering roseate terns from the northeast North America populations using Hispaniola as a stopover site during migration.

British Virgin Islands: Roseate terns have been reported nesting at Carrot Rock, Cooper Island, Cockroach and Dog Isles, Fallen Jerusalem, Green Cay, Guana Island, Indian Rocks, Round Rock, Seal Dog Islands, and Watson's Rock (Schreiber and Pierce 2009) (Fig. 3). Recent annual surveys have shown that they regularly shift colony sites between the U.S. Virgin Islands and the British Virgin Islands. Pierce (2001) reported a minimum of 350, and a maximum of 1,755 pairs, for the British Virgin Islands. Schreiber and Pierce (2009) estimated that between 600 and 2,000 pairs of roseate terns breed in the British Virgin Islands, with numbers varying each year as the birds shift around.

Anguilla: At least 210 pairs were recorded nesting on a sandy spit at the eastern end of Scrub Island in 2000 (Holliday and Hodge 2009). It had also been recorded breeding on Sombrero in 1964 and 1985.

French West Indies (Guadeloupe Archipelago and Martinique): In Martinique, an incomplete survey in 2006 estimated 250 nesting pairs of roseate terns at Pain de Sucre (Sainte Marie) and about 150 to 200 pairs in Islet Petit-Pinon; the largest count east of the Puerto Rico Bank (Lemoine *et al.* 2009; Dubief and Leblond 2010) (Fig. C3). About 80 pairs were recorded at two sites in the Guadeloupe Archipelago (Dubief and Leblond 2010).

Netherlands Antilles (St. Maarten, Saba, and St. Eustatius): Roseate terns have been reported in the Netherlands Antilles, but it is not known if nesting occurs (Collier and Brown 2009a) (Fig. C3).

St. Kitts and Nevis: The roseate tern is an irregular breeder in small numbers (e.g., six pairs in 2004) and it is threatened by dogs, cats, rats, mice (*Mus musculus*), mongoose (*Herpestes* spp.), and African green monkeys (*Cercopithecus* spp.) (Collier and Brown 2009b) (Fig. C3).

Antigua, Barbuda, and Redonda: The species appears to nest sporadically and in small numbers; but no recent breeding surveys have been conducted (Sylvester *et al.* 2009) (Fig. C3).

St. Lucia: The roseate tern is an irregular breeder in small numbers (up to 75 pairs in 1992 and 2000); where the regular presence of humans on the many offshore islands during the nesting season, and sporadic eggging, disturb nesting seabirds and cause nest failure (Anthony and Dornelly 2009) (Fig. C3).

St. Vincent, the Grenadines, and Grenada: These islands support small, scattered colonies of about 15 pairs of roseate terns; although earlier accounts suggest that the species was “fairly common” (Frost *et al.* 2009) (Fig. C3). The extent to which egg collection occurs at present in St. Vincent, the Grenadines, and Grenada is unknown, but there are reports that poaching still occurs in the Grenadines (Frost *et al.* 2009).

Trinidad and Tobago: Seabird colonies are generally restricted to the smaller, uninhabited satellite offshore islands, particularly off the coast of Tobago (Hayes and Bodnar 2009) (Fig. C3). About 50 adults (25 pairs) were seen nesting on a vegetated islet off Petite Tacarib Bay, and another small group may have been nesting on a rock near Huevos Island, both on the north coast of Trinidad, in May 2002 (Hayes and Bodnar 2009). Numbers of nesting roseate terns in Tobago have varied from about 10 to 200 pairs; with the most recent count of 111 pairs nesting on a rock off Courland Point in May 2001 (Hayes and Bodnar 2009).

Aruba, Curaçao, and Bonaire (formerly Netherlands Lesser Antilles): The largest aggregations of seabirds in these islands are found in Lago Reef, Aruba; where fluctuating small numbers of roseate terns have been recorded since 1979 (van Halewyn 2009) (Fig. C2). The most recent breeding population count for roseate terns known to van Halewyn (2009) was 52 pairs of roseate terns in 2001. However, A. del Nevo (unpublished data) reports up to 65 pairs in 2005 to 2009. This represents a decrease from 71 to 112 pairs in the 1980s (Gochfeld *et al.* 1994). No

roseate tern nesting activity has been documented recently for Curaçao and Bonaire, but the species nested at several sites on both islands between 1960 and 1983 (Debrot *et al.* 2009).

Islands off Venezuela: During the 1950s, Roseate Terns were recorded breeding on five islands in the Los Roques archipelago and in “large numbers” on two islands in the Las Aves archipelago (Nisbet 1980). They are now thought to be extirpated from Los Roques (Esclasans *et al.* 2009); there is no recent information from Las Aves.

## A.2 Threats

Cuba: Increased development of tourism in coastal regions; collection of eggs and unfledged young for food; hunting or fishing for terns with hooks to collect their metal bands; and building of causeways to access cays and islands (which facilitate introduction of feral animals), threaten the roseate tern and other nesting seabirds in Cuba (Jiménez *et al.* 2009).

Jamaica: Introduced predators, egg collection, fishing camps on seabird cays, and other forms of habitat disturbance have been identified as the major threats to Jamaican seabirds, including roseate terns (Haynes-Sutton 2009). The majority of Jamaican nesting and roosting seabird colonies are less than 3 meters above mean sea level at the highest point and, therefore, particularly vulnerable to global warming and the associated projected sea level rise (Haynes-Sutton 2009).

Hispaniola: Seabirds, and most other birds, have little effective protection in Hispaniola. None of the laws, regulations, or protected area designations, either national or international, has much impact in a practical sense since there is no law enforcement at all in Haiti, and not much in the Dominican Republic (Keith 2009). Nearly all seabird colonies located anywhere near human populations continue to be regularly robbed of eggs, even at Isla Beata, which is designated a Ramsar Biosphere Reserve and a Dominican Republic national park (Keith 2009).

Bahamas: Hallett (2009) cites the following threats to the species in the Bahamas: loss of breeding habitat for tourist development, intentional and unintentional human disturbance, poaching of eggs and birds, predation by feral animals, and deleterious human effects on the species' food chain (such as pollution and over-fishing).

Turks and Caicos Islands: Loss or disturbance of coastal habitat in Salt Cay, and egg collection on the Sand Cays (largely by illegal immigrants), threaten nesting roseate terns in the Turks and Caicos (Pienkowski 2009).

Aruba, Curaçao, and Bonaire (formerly Netherlands Lesser Antilles): In Aruba, van Halewyn (2009) considered that close proximity to sites of heavy pollution from an oil refining plant, growing tourism development, and egg collecting, placed these breeding colonies at considerable risk. However, the security zone around the refinery currently makes Lago Reef a de facto sanctuary and other species of terns have flourished there. A. del Nevo (personal communication) considers human disturbance to be the main threat to roseate terns in Aruba, because in recent years they have settled first in sites outside the refinery where they are subject to heavy human disturbance, moving to Lago Reef later in the season for re-nesting. In Curaçao

and Bonaire, Debrot et al. (2009) suggested that roseate terns may be particularly vulnerable to local extirpation when large colonies of other species that they nest with (mixed species colonies) are disrupted by recreational disturbance from tourists. Oil pollution and cat predation also threaten roseate terns in Curaçao (Debrot *et al.* 2009).

Anguilla: The main threats to this species are disturbance of nesting pairs by tourists, development of the privately owned islands, and introduction of rats (Holliday and Hodge 2009).

French West Indies (Guadeloupe Archipelago and Martinique): The main threats are considered to be predation by black rats (*Rattus rattus*), mongooses (*Herpestes* spp.) and peregrine falcons (*Falco peregrinus*), human disturbance during the nesting season, and collection of eggs, chicks and adults (Leblond 2009, Lemoine *et al.* 2009, Dubief and Leblond 2010).

St. Kitts and Nevis: The roseate tern is threatened by dogs, cats, rats, mice (*Mus musculus*), mongoose, and African green monkeys (*Cercopithecus* spp.) (Collier and Brown 2009b).

St. Lucia: The regular presence of humans on the many offshore islands during the nesting season, and sporadic eggging, disturb nesting seabirds and cause nest failure (Anthony and Dornelly 2009).

St. Vincent, the Grenadines, and Grenada: The extent to which egg collection occur at present in these islands is unknown, but there are reports that poaching still occurs in the Grenadines (Frost *et al.* 2009).

Trinidad and Tobago: The major seabird colonies in Trinidad and Tobago are legally protected as wildlife sanctuaries. However, despite legal protection of seabirds within wildlife sanctuaries, law enforcement is lax and poaching of eggs still occurs (Hayes and Bodnar 2009).