RUFA RED KNOT
BACKGROUND INFORMATION AND THREATS ASSESSMENT

SUPPLEMENT TO
Endangered and Threatened Wildlife and Plants; Final Threatened Status for the Rufa Red Knot (*Calidris canutus rufa*)
[Docket No. FWS–R5–ES–2013–0097; RIN AY17]

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SUMMARY

TERMINOLOGY: Throughout this document, "rufa red knot," "red knot," and "knot" are used interchangeably to refer to the subspecies *Calidris canutus rufa*. "*Calidris canutus*" and "*C. canutus*" are used to refer to the species as a whole or to birds of unknown subspecies. References to other particular subspecies are so indicated by use of the Latin name. "Winter" is consistently used to refer to the nonbreeding period of the red knot life cycle when the birds are not undertaking migratory movements, typically December to February, although this period is actually summer in the Southern Hemisphere. Likewise, although the seasons are reversed in the Southern Hemisphere, "spring" is used throughout to refer to the nonbreeding period of the red knot life cycle when the birds are undertaking northbound migratory movements and "fall" is used to refer to the nonbreeding period when the birds are undertaking southbound migratory movements.

INTRODUCTION: The rufa red knot (*Calidris canutus rufa*) is a medium-sized shorebird that migrates annually between its breeding grounds in the central Canadian Arctic and several wintering regions, including the Southeast United States (Southeast), the Northeast Gulf of Mexico, northern Brazil, and Tierra del Fuego at the southern tip of South America. During both the northbound (spring) and southbound (fall) migrations, red knots use key staging and stopover areas to rest and feed. Another subspecies, *Calidris canutus roselaari*, breeds in western Alaska and on Wrangel Island in eastern Russia (Carmona *et al.* 2013, p. 169; Buehler and Baker 2005, p. 498) and winters on the Pacific coast from northern Mexico through Panama and possibly farther south (D. Newstead pers. comm. February 13, 2014; Carmona *et al.* 2013, pp. 171, 175). The nonbreeding ranges of these two subspecies are known to overlap in a few locations, and may overlap more broadly. However, geolocator data confirm the existence of distinct breeding areas for the *rufa* and *roselaari* subspecies (D. Newstead pers. comm. February 13, 2014; L. Niles pers. comm. January 4, 2013; Newstead *et al.* 2013, p. 56; Niles *et al.* 2012a, pp. 197–200; Niles *et al.* 2010a, pp. 125–126). The rufa red knot's typical life span is at least 7 years (J. Parvin pers. comm. March 14, 2014; Niles *et al.* 2008, p. 28), with the oldest known wild bird at least 21 years old as of 2014 (Bauers 2014; Jordan 2014). Age of first breeding is at least 2 years (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014; Harrington 2001, p. 21).

BREEDING: The red knot breeds in the central Canadian Arctic, from the islands of northern Hudson Bay to the Foxe Basin shoreline of Baffin Island, and west to Victoria Island (Niles *et al.* 2008, pp. 15–16; Morrison and Harrington 1992, p. 73). Potential breeding habitat extends farther north the southern Queen Elizabeth Islands (Niles *et al.* 2008, p. 16). The extent to which rufa red knots from different wintering areas mix on the breeding grounds, and therefore potentially interbreed, is poorly known (Harrington *et al.* 1988, p. 443). Red knots generally nest in dry, slightly elevated tundra locations, often on windswept slopes with little vegetation. Breeding areas are located inland, but near arctic coasts. Nests may be scraped into patches of mountain avens (*Dryas octopetala*) plants, or in low spreading vegetation on hummocky (characterized by knolls or mounds) ground containing lichens, leaves, and moss. After the eggs hatch, red knot chicks and adults quickly move away from high nesting terrain to lower, freshwater wetland habitats. On the breeding grounds, the red knot's diet consists mostly of terrestrial invertebrates such as insects and other arthropods (Niles *et al.* 2008, p. 27; Harrington 2001, p. 11).

Pair bonds form soon after the birds arrive on breeding grounds, in late May or early June, and remain intact until shortly after the eggs hatch (Niles *et al.* 2008, p. 25–26; Harrington 2001, p. 16). Female rufa red knots lay only one clutch (group of eggs) per season, and, as far as is known, do not lay a replacement clutch if the first is lost. The usual clutch size is four eggs, though three-egg clutches have been recorded. The incubation period lasts approximately 22 days from the last egg laid to the last egg hatched, and both sexes participate equally in egg incubation. Young are precocial, leaving the nest within 24 hours of hatching and foraging for themselves (Niles *et al.* 2008, p. 27). Females are thought to leave the breeding grounds and start moving south soon after the chicks hatch in mid-July. Thereafter, parental care is provided solely by the males, but about 25 days later (around August 10) males also abandon the newly fledged juveniles and move south. Not long after, they are followed by the juveniles (Niles *et al.* 2008, p. 14). Breeding success of High Arctic shorebirds such as *Calidris canutus* varies dramatically among years in a somewhat cyclical manner. Two main factors seem to be responsible for this annual variation: abundance of arctic lemmings (*Dicrostonyx torquatus* and *Lemmus sibericus*) (by indirectly affecting predation pressure on shorebirds) and weather (Piersma and Lindström 2004, pp. 63–64; Blomqvist *et al*. 2002, p. 149; Summers and Underhill 1987, p. 169). Growth rate of *C. canutus* chicks is very high compared to similarly sized shorebirds nesting in more temperate climates and is strongly correlated with weather-induced and seasonal variation in availability of invertebrate prey (Schekkerman *et al*. 2003, p. 332).

NONBREEDING RANGE: Geolocator and resightings data show definitively that the *rufa* nonbreeding range includes the entire Atlantic and Caribbean coasts of South America and the Caribbean islands; Chiloé Island on the central Pacific coast of Chile; the Pacific coast of Panama; the North American Gulf and Atlantic coasts from Tamaulipas, Mexico through Quebec, Canada; the interior of South America; and the interior of the United States and Canada west at least as far as the Great Plains (Bimbi *et al.* 2014, pp. 29–31; S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8 and 12, 2014; Newstead 2014a, p. 19; D. Newstead pers. comm. May 8, 2014; Niles 2014; J. Parvin pers. comm. March 13, 2014; Newstead *et al.* 2013, pp. 55–57; Burger *et al.* 2012b, p. 107; Niles 2012a; Niles *et al.* 2012a, entire; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, entire; Niles *et al.* 2008, p. 19; B. Paxton pers. comm. November 9, 2008; Buehler 2002, p. 42; Morrison and Harrington 1992, p. 77). *Calidris canutus roselaari* also occurs in certain parts of this established *rufa* nonbreeding range. Best available data are limited but suggest that the nonbreeding ranges of *C.c. roselaari* and *C.c. rufa* overlap, at least in Texas during spring and in Panama during winter (D. Newstead pers. comm. May 13, 2014; D. Newstead pers. comm. February 13, 2014; D. Newstead pers. comm. February 11, 2014; D. Newstead pers. comm. August 20, 2012). However, geolocator and resightings data provide strong evidence that *Calidris cantus* on the Pacific coast from northeastern Russia to Las Garzas, Mexico are the *roselaari* subspecies, and we conclude from the best available data that the rufa red knot does not occur in this region of the Pacific (D. Newstead pers. comm. February 13, 2014; Carmona *et al.* 2013, entire; J. Buchanan pers. comm. January 9, 2013).

WINTERING: Wintering areas for the rufa red knot include the Atlantic coasts of Argentina and Chile (particularly the island of Tierra del Fuego that spans both countries), the north coast of Brazil (particularly in the State of Maranhão), the Northwest Gulf of Mexico from the Mexican State of Tamaulipas through Texas (particularly at Laguna Madre) to Louisiana, and the Southeast United States from Florida (particularly the central Gulf coast) to North Carolina (Newstead 2014a, p. 19; Newstead *et al.* 2013, p. 55; L. Patrick pers. comm. August 31, 2012; Niles *et al.* 2008, p 17). Smaller numbers of knots winter in the Caribbean, and along the central Gulf coast (Alabama, Mississippi), the mid-Atlantic, and the Northeast United States (eBird.org 2014; Russell 2014, p. 4; Burger *et al.* 2012b, p. 6; A. Dey pers. comm. November 19, 2012; H. Hanlon pers. comm. November 22, 2012; Niles *et al.* 2012a, entire; L. Patrick pers. comm. August 31, 2012; Morrison and Harrington 1992, p. 77). *Calidris canutus* is also known to winter in Central America, northwest South America, and along the Pacific coast of South America, but it is not yet clear if all these birds are the *rufa* subspecies (Carmona *et al.* 2013, entire). Winter area fidelity appears to be high, with minimal movement of birds among wintering regions (Georgia Department of Natural Resources (GDNR) 2013; BandedBirds.org 2012; Schwarzer *et al.* 2012, p. 729; Niles *et al.* 2008, pp. 9, 55; Harrington *et al.* 1988, p. 441). Researchers often distinguish between those rufa red knots that winter the farthest south (in Argentina and Chile) and therefore undertake the longest-distance migrations ("southern-wintering"), from those that winter farther north in northern Brazil and the Southeast ("northern-wintering"), with some notable physiological and ecological differences between the two groups (B. Harrington pers. comm. November 14, 2013).

MIGRATION BIOLOGY: Each year some red knots make one of the longest distance migrations known in the animal kingdom, traveling up to 19,000 mi (30,000 km) annually. Red knots undertake long flights that may span thousands of miles without stopping. As *Calidris* *canutus* prepare to depart on long migratory flights, they undergo several physiological changes. Before takeoff, the birds accumulate and store large amounts of fat to fuel migration and undergo substantial changes in metabolic rates. In addition, the leg muscles, gizzard (a muscular organ used for grinding food), stomach, intestines, and liver all decrease in size, while the pectoral (chest) muscles and heart increase in size. Due to these physiological changes, *C. canutus* arriving from lengthy migrations are not able to feed maximally until their digestive systems regenerate, a process that may take several days. Because stopovers are time-constrained, *C. canutus* requires stopovers rich in easily digested food to achieve adequate weight gain (Niles *et al.* 2008, pp. 28–29; van Gils *et al.* 2005a, p. 2609; van Gils *et al.* 2005b, pp. 126–127; Piersma *et al*. 1999, pp. 405; 412) that fuels the next migratory flight and, upon arrival in the Arctic, also fuels a body transformation to breeding condition (Morrison 2006, pp. 610–612). At some stages of migration, very high proportions of entire shorebird populations may use a single migration staging site to prepare for long flights. High fractions of the red knot's rangewide population can occur together at a small number of nonbreeding locations, leaving populations vulnerable to loss of key resources (Harrington 2001, p 22). For example, Delaware Bay provides the final Atlantic coast stopover for a significant majority (50 to 80 percent) of the red knot population making its way to the arctic breeding grounds each spring (Clark *et al*. 2009, p. 90; Brown *et al.* 2001, p. 10). Individual red knots show moderate fidelity to particular migration staging areas between years (French Guiana Regional Scientific Council for Natural Heritage (CSRPN) 2013; Duerr *et al.* 2011, p. 16; Watts 2009a; Harrington 2001, pp. 21–22).

SPRING MIGRATION: Well-known spring stopover areas along the Atlantic coast include Río Gallegos, Península Valdés, and San Antonio Oeste (Patagonia, Argentina); Lagoa do Peixe (eastern Brazil, State of Rio Grande do Sul); Maranhão (northern Brazil); the Southeast United States (e.g., the Carolinas to Florida); the Virginia barrier islands (United States); and Delaware Bay (Delaware and New Jersey, United States) (A. Dey pers. comm. April 21, 2014; Wallover *et al.* 2014, p. 6; GDNR 2013; South Carolina Department of Natural Resources (SCDNR) 2013, p. 36; Cohen *et al.* 2009, p. 939; Niles *et al.* 2008, p. 19; González 2005, p. 14). However, large and small groups of red knots, sometimes numbering in the thousands, may occur in suitable habitats all along the Atlantic and Gulf coasts from Argentina to Massachusetts (Niles *et al.* 2008, p. 29).

Although a few birds may depart before the end of January, the main red knot movement north from Tierra del Fuego occurs in February. The northward migration through South America is typically rapid, with only brief stopovers (Niles *et al*. 2008, p. 15), although longer stops in Argentina (17 to 22 days) have been reported (Musmeci *et al.* 2012, pp. 359–360). Birds moving north from Argentina typically arrive in Brazil in April (Scherer and Petry 2012, p. 46; Niles *et al.* 2008, p. 29). Departure from Brazil tends to occur in the first half of May (Niles *et al.* 2010a, p. 126; Niles *et al*. 2008, pp. 15, 29). Many knots marked in Argentina and Chile are seen on the Atlantic coasts of Florida, Georgia, South Carolina, and North Carolina during, but not before, May (B. Harrington pers. comm. November 14, 2013; GDNR 2013; SCDNR 2013, p. 31). Available data indicate that red knots wintering in the Southeast use at least two different spring migration routes—coastal (moving north along the coast to the mid-Atlantic before departing for the Arctic) and inland (departing overland for the Arctic directly from the Southeast coast) (Bimbi *et al.* 2014, pp. 29–30; SCDNR 2013, p. 38; Niles *et al.* 2012a, pp. 197–200; Harrington 2005a, p. 1; Morrison and Harrington 1992, p. 77).

FALL MIGRATION: Departure from the breeding grounds begins in mid-July and continues through August. Females are thought to leave first, followed by males and then juveniles (Niles *et al.* 2008, pp. 14–15; Harrington 2001, p. 6). Adult *Calidris canutus* pass through stopover sites along the migratory route earlier in years with low reproductive success than in years with high reproductive success (Blomqvist *et al*. 2002, p. 149). Along the U.S. Atlantic coast, southbound red knots start arriving in July. Numbers of adults peak in mid-August and most depart by late September, although geolocators and resightings have shown some birds (especially northern-wintering knots) stay through November (Wallover *et al.* 2014, p. 6; Niles *et al.* 2012a, pp. 197–200; Harrington *et al.* 2010b, p. 357; Harrington 2001, p. 2). Well-known fall stopover sites include southwest Hudson Bay (including the Nelson River delta), James Bay, the north shore of the St. Lawrence River, the Mingan Archipelago, and the Bay of Fundy in Canada; the coasts of Massachusetts and New Jersey and the mouth of the Altamaha River in Georgia in the United States; the Caribbean (especially Puerto Rico and the Lesser Antilles); and the northern coast of South America from Brazil to Guyana (eBird.org 2014; Autoridad de Energía Eléctrica (Electric Energy Authority, or (AEE) 2013; Newstead *et al.* 2013, p. 57; Niles 2012a; D. Mizrahi pers. comm. October 16, 2011; Niles *et al.* 2010a, pp. 125–136; Schneider and Winn 2010, p. 3; Niles *et al.* 2008, pp. 30, 75, 94; B. Harrington pers. comm. March 31, 2006; Antas and Nascimento 1996, p. 66; Morrison and Harrington 1992, p. 74; Spaans 1978, p. 72). However, birds can occur all along the coasts in suitable habitats. In one study of northern-wintering red knots, the total time spent along the U.S. Atlantic coast (including spring, fall, and for some birds winter) averaged 218 days (range 121 to 269 days) (Burger *et al.* 2012b, p. 1), or about 60 percent of the calendar year.

MIDCONTINENTAL MIGRATION: Geolocator results from red knots wintering in Texas have shown that these birds typically use a central, overland flyway across the midcontinental United States, with birds departing Texas between May 16 and May 21 and using stopover areas in the Northern Great Plains and along southern Hudson Bay (Newstead *et al.* 2013, p. 58). Texas-wintering birds typically use a similar and direct interior flyway across the midcontinental United States during the southbound migration, using a southbound stopover site on the south shore of Hudson Bay (Nelson River Delta to James Bay). Geolocator results (Bimbi *et al.* 2014, pp. 29–31; Niles 2014; Newstead *et al.* 2013; Niles *et al.* 2012a, p. 197-200; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, pp. 125–128) have suggested that rufa red knots exhibit strong flyway fidelity (Newstead *et al.* 2013, p. 58) (i.e., not switching between Atlantic coast and midcontinental routes). However, newer geolocator data, as yet unpublished, do show some switching between these two flyways. Several Texas-wintering birds have been shown to use the "typical" midcontinental flyway in spring, but then follow a fall migration route along the U.S. Atlantic coast before returning Texas via the Gulf coast. To date, no known geolocator tracks from Texas birds have shown use of the Atlantic coast during spring migration, but some resighting data suggest that this may also occur (D. Newstead pers. comm. May 8, 2014). Even for the same individual bird, the actual routes and number of stopovers can vary considerably from year to year (D. Newstead pers. comm. May 8, 2014). In one study, red knots wintering in the Northwest Gulf of Mexico spent nearly the entire nonbreeding phase of their annual cycle (286 days, or 78.4 percent of the calendar year) on the Texas coast (Newstead *et al.* 2013, p. 55).

NONBREEDING HABITAT: Coastal habitats used by red knots in migration and wintering areas are similar in character (Harrington 2001, p. 9), generally coastal marine and estuarine (partially enclosed tidal area where fresh and salt water mixes) habitats with large areas of exposed intertidal sediments. Migration and wintering habitats include both high-energy ocean- or bay-front areas, as well as tidal flats in more sheltered bays and lagoons (Harrington 2001, p. 9). Preferred wintering and migration microhabitats are muddy or sandy coastal areas, specifically, the mouths of bays and estuaries, tidal flats, and unimproved tidal inlets (North Carolina Wildlife Resources Commission (NCWRC) 2013; Lott *et al.* 2009, pp. 18–19; Niles *et al.* 2008, p. 30; Harrington 2001, p. 8). Along the U.S. Atlantic coast, dynamic and ephemeral (lasting only briefly) features are important red knot habitats, including sand spits, islets, shoals, and sandbars, features often associated with inlets (Harrington 2008, p. 2; Harrington in Guilfoyle *et al.* 2007, pp. 18–19; Winn and Harrington in Guilfoyle *et al.* 2006, pp. 8–10). In many wintering and stopover areas, quality high-tide roosting habitat (i.e., close to feeding areas, protected from predators, with sufficient space during the highest tides, free from excessive human disturbance) is limited (CSRPN 2013; K. Kalasz pers. comm. November 26, 2012; L. Niles pers. comm. November 19 and 20, 2012; Kalasz 2008, p. 9). In nonbreeding habitats, *Calidris canutus* require sparse vegetation to avoid predation (Niles *et al.* 2008, p. 44; Piersma *et al.* 1993, pp. 338–339, 349).

Available information suggests that red knots use inland saline lakes as stopover habitat in the Northern Great Plains (Newstead *et al.* 2013, p. 57; North Dakota Game and Fish Department (NDGFD) 2013; Western Hemisphere Shorebird Reserve Network (WHSRN) 2012; Skagen *et al.* 1999). We have little information to indicate whether or not red knots may also utilize inland freshwater habitats during migration, but data suggest that certain freshwater areas may warrant further study as potential stopover habitats (C. Dovichin pers. comm. May 6, 2014; eBird.org 2014; Russell 2014, entire). Best available data indicate that small numbers of red knots sometimes use manmade freshwater habitats (e.g., impoundments) along inland migration routes (eBird.org 2014; Russell 2014, entire; Central Flyway Council 2013; NDGFD 2013; Oklahoma Department of Wildlife Conservation (ODWC) 2013; A. Simnor pers. comm. October 15, 2012).

NONBREEDING FOOD: Across all (six) subspecies, *Calidris canutus* is a specialized molluscivore, eating hard-shelled mollusks, sometimes supplemented with easily accessed softer invertebrate prey, such as shrimp- and crab-like organisms, marine worms, and horseshoe crab eggs (Piersma and van Gils 2011, p. 9; Harrington 2001, pp. 9–11). The mollusk prey is swallowed whole and crushed in the gizzard, which in *C. canutus* is the largest (relative to body size) among any shorebird species evaluated (Piersma and van Gils 2011, pp. 9–11). Large gizzards are among this species' adaptations to a mollusk diet, allowing *C. canutus* to grind the hard shells of its prey. *Calidris canutus* prefer thin-shelled to thick-shelled prey species because they are easier to digest and provide a more favorable meat to mass ratio (higher prey quality) (van Gils *et al.* 2005a, p. 2611; Harrington 2001, p. 11; Zwarts and Blomert 1992, p. 113). From studies of other subspecies, Zwarts and Blomert (1992, p. 113) concluded that *C. canutus* cannot ingest prey with a circumference greater than 1.2 in (30 millimeters (mm)). For rufa red knots, prey lengths of 0.16 to 0.79 in (4 to 20 mm) have been observed (Cohen *et al.* 2010b, pp. 359–360; González *et al.* 1996, p. 575). Foraging activity is largely dictated by tidal conditions, as *C. canutus* rarely wade in water more than 0.8 to 1.2 in (2 to 3 cm) deep (Harrington 2001, p. 10). Due to bill morphology, *C. canutus* is limited to foraging on only shallow-buried prey, within the top 0.8 to 1.2 in (2 to 3 cm) of sediment (Gerasimov 2009, p. 227; Zwarts and Blomert 1992, p. 113). Along the U.S. coast, *Donax* and *Mulinia* clams and blue mussel (*Mytilus edulis*) spat are key prey items. A prominent departure from typical prey items occurs each spring when red knots feed on the eggs of horseshoe crabs (*Limulus polyphemus*), particularly during the key migration stopover within the Delaware Bay. Delaware Bay serves as the principal spring migration staging area for the red knot because of the abundance and availability of horseshoe crab eggs (Clark *et al.* 2009, p. 85; Harrington 2001, pp. 2, 7; Harrington 1996, pp. 76–77; Morrison and Harrington 1992, pp. 76–77). In Delaware Bay, horseshoe crab eggs are a superabundant source of easily digestible food.

POPULATION TRENDS: After a thorough review of the best available population data, we conclude that we do not have sufficient reliable data on which to derive a precise rangewide population estimate for the rufa red knot. For example, there are no rangewide population estimates for fall migration or breeding areas because birds are too dispersed. However, we can reliably infer population trend information from some areas. We have high confidence in long-term survey data from two key red knot areas, Tierra del Fuego (wintering) and Delaware Bay (spring), showing declines of 70 to 75 percent over roughly the same period, since about 2000 (Dey *et al.* 2014, p. 2; Dey *et al.* 2011a, p. 2; Clark *et al.* 2009, p. 88; Morrison *et al*. 2004, p. 65; Morrison and Ross 1989, Vol. 2, pp. 226, 252; Kochenberger 1983, p. 1; Dunne *et al.* 1982, p. 67; Wander and Dunne 1982, p. 60). Data sets associated with lower confidence, from the Brazil wintering region and three South American spring stopovers, also suggest declines roughly over this same timeframe (Niles *et al.* 2008, pp. 58, 134; Baker *et al.* 2005, p. 12; González 2005, p. 14; Morrison and Ross 1989, Vol. 2, p. 183; Harrington *et al.* 1986, p. 50), however, more recently a substantial increase was documented in Brazil (Dey *et al.* 2014, p. 1). Emerging information from Virginia also suggests a decline relative to the 1990s (B. Watts pers. comm. August 22, 2014). We do not conclude that the Southeast wintering region has declined over this period despite some years of lower counts in Florida, due to the likelihood that the birds' usage shifts geographically within this region from year to year (Harrington 2005a, pp. 1, 15). In summary, the best available data indicate a sustained decline occurred in the 2000s, and may have stabilized at a relatively low level in the last few years. Attempts to evaluate long-term population trends using national or regional data from volunteer shorebird surveys and other sources have also generally concluded that red knot numbers have declined, probably sharply (National Park Service (NPS) 2013; Andres 2009; Morrison *et al.* 2006, pp. 71, 76–77).

LISTING FACTORS: Under section 4(a)(1) of the Endangered Species Act (the Act), we may list a species based on any of the following five factors: (A) the present or threatened destruction, modification, or curtailment of its habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) the inadequacy of existing regulatory mechanisms; or (E) other natural or manmade factors affecting its continued existence. We have evaluated each of these five factors.

FACTOR A: Threats to the red knot from habitat destruction and modification are occurring throughout the entire range of the subspecies. These threats include climate change, shoreline stabilization, and coastal development, exacerbated regionally or locally by lesser habitat-related threats such as beach cleaning, invasive vegetation, agriculture, and aquaculture. The subspecies-level impacts from these activities are expected to continue into the future.

Within the nonbreeding portion of the range, red knot habitat is primarily threatened by the highly interrelated effects of sea level rise, shoreline stabilization, and coastal development. The primary red knot foraging habitats, intertidal flats and sandy beaches, will likely be locally or regionally inundated as sea levels rise, but replacement habitats are likely to re-form along eroding shorelines in their new positions (U.S. Climate Change Science Program (CCSP) 2009b, p. 186; Scavia *et al.* 2002, p. 152). However, if shorelines experience a decades-long period of rapid sea level rise, high instability, and landward migration, the formation rate of new foraging habitats may be slower than the rate at which existing habitats are lost (Iwamura *et al.* 2013, p. 6). In addition, low-lying and narrow islands (e.g., in the Caribbean, along the Gulf and Atlantic coasts) may disintegrate rather than migrate, representing a net loss of red knot habitat (Chapter 5 <u>in</u> International Panel on Climate Change (IPCC) 2014, p. 15; Titus 1990, p. 67).

Superimposed on changes from sea level rise are widespread human efforts to stabilize the shoreline, which are known to exacerbate losses of intertidal habitats by blocking their landward migration. About 40 percent of the U.S. coastline within the range of the red knot is already developed, and much of this developed area is stabilized by a combination of existing hard structures and ongoing beach nourishment programs (Rice 2012a, p. 6; Titus *et al.* 2009, p. 5). Hard stabilization structures and dredging degrade and often eliminate existing intertidal habitats, and in many cases prevent the formation of new shorebird habitats (CCSP 2009b, pp. 99–100; Nordstrom 2000, pp. 20, 98–107). Beach nourishment may temporarily maintain suboptimal shorebird habitats where they would otherwise be lost as a result of hard structures or sea level rise (Nordstrom and Mauriello 2001, entire), but beach nourishment can also have adverse effects to red knots and their habitats (Defeo *et al.* 2009, p. 4; Rice 2009, entire; Peterson *et al.* 2006, entire; Peterson and Bishop 2005, entire; Greene 2002, p. 5). In those times and places where artificial beach maintenance is abandoned (e.g., due to constraints on funding or sediment availability), the remaining alternatives available to coastal communities would likely be limited to either a retreat from the coast or increased use of hard structures to protect development (CCSP 2009b, p. 87; Defeo *et al*. 2009, p. 7). The quantity of red knot habitat would be markedly decreased by a proliferation of hard structures. Red knot habitat would be significantly increased by retreat, but only where hard stabilization structures do not exist or where they get dismantled. Relative to the United States, little is known about development-related threats to red knot nonbreeding habitat in other countries. However, in some key international wintering and stopover sites, development pressures are likely to exacerbate habitat impacts caused by sea level rise (CSRPN 2013; WHSRN 2012; Niles *et al.* 2008, pp. 17, 19, 73, 97–98; Ferrari *et al.* 2002, p. 39).

Lesser threats to nonbreeding habitat include beach cleaning, invasive vegetation, agriculture, and aquaculture. The practice of intensive beach raking may cause physical changes to beaches that degrade their suitability as red knot habitat (Defeo *et al.* 2009, p. 4; Nordstrom and Mauriello 2001). Although not a primary cause of habitat loss, invasive vegetation can be a regionally important contributor to the overall loss and degradation of the red knot's nonbreeding habitat (U.S. Fish and Wildlife Service (USFWS) 2012a, p. 27; Defeo *et al.* 2009, p. 6). Agriculture and aquaculture are a minor but locally important contributor to overall loss and degradation of the red knot's nonbreeding habitat, particularly for moderate numbers of red knots that winter or stopover in Northeast Brazil where habitats were likely impacted by the rapid expansion of shrimp farming since 1998 (Carlos *et al.* 2010, entire).

Within the breeding portion of the range, the primary threat to red knot habitat is from climate change. With arctic warming, vegetation conditions on the breeding grounds are changing, which is expected to eventually cause the zone of nesting habitat to shift north and contract (Feng *et al.* 2012, pp. 1359, 1366; Meltofte *et al.* 2007, p. 35; Arctic Climate Impact Assessment (ACIA) 2005, pp. 991, 998). Studies have already documented changes in arctic vegetation (e.g, increases in peak "greenness" and plant biomass; advancing of the arctic tree line; increased shrub abundance, biomass, and cover; increased plant canopy heights; and decreased prevalence of bare ground (Summary for Policymakers in IPCC 2014, p. 32; Chapter 28 in IPCC 2014, p. 12)). Vegetation effects are likely exacerbated by loss of sea ice (Bhatt *et al.* 2010, pp. 1–2l; Meltofte *et al.* 2007, p. 36). Arctic freshwater systems, foraging areas for red knots during the nesting season, are particularly sensitive to climate change and are already being affected (ACIA 2005, p. 1012; Meltofte *et al.* 2007, p. 35). Unpredictable but profound ecosystem changes (e.g., changing interactions among predators, prey, and competitors) are also likely to occur. There are early warning signs that arctic ecosystems are already experiencing irreversible regime shifts (Summary for Policymakers in IPCC 2014, p. 12). We conclude that ecosystem changes in the Arctic are already underway and likely to continue, and that arctic ecosystems likely face much greater future change that may be abrupt and irreversible. Further, climate change is opening the Arctic to development such as oil and gas exploration, commercial shipping, tourism, and fishing (Niles 2013; National Research Council (NRC) 2013, p. 4; Smith and Stephenson 2013, p. 2; Astill 2012; Roach 2007).

FACTOR B: Threats to the red knot from overutilization for commercial, recreational, scientific, or educational purposes exist in parts of the Caribbean and South America. Specifically, legal and illegal hunting do occur. We expect mortality of individual knots from hunting to continue into the future, but at stable or decreasing levels due to the recent international attention to shorebird hunting, and due to new voluntary and regulatory hunting restrictions in some areas.

Legal and illegal sport and market hunting in the mid-Atlantic and Northeast United States substantially reduced red knot populations in the 1800s, and we do not know if the subspecies ever fully recovered its former abundance or distribution (Karpanty *et al.* 2014, p. 2; Cohen *et al.* 2008; Harrington 2001, p. 22). Neither legal nor illegal hunting are currently a threat to red knots in the United States, but both occur in the Caribbean and parts of South America (Harrington 2001, p. 22). Hunting pressure on shorebirds in the Lesser Antilles (e.g., Barbados, Guadeloupe) is very high (USFWS 2011e, pp. 2–3), but only small numbers of red knots have been documented on these islands, so past mortality may not have exceeded tens of birds per year (G. Humbert pers. comm. November 29, 2013). Red knots are no longer being targeted in Barbados or Guadeloupe, and other measures to regulate shorebird hunting on these islands are being negotiated (G. Humbert pers. comm. November 29, 2013; McClain 2013; USFWS 2011e, p. 2). Much larger numbers (thousands) of red knots occur in the Guianas, where legal and illegal subsistence shorebird hunting is common (CSRPN 2013; Niles 2012b; Ottema and Spaans 2008, p. 343). About 20 red knot mortalities have been documented in the Guianas (D. Mizrahi pers. comm. October 16, 2011; Harrington 2001, p. 22), but total red knot hunting mortality in this region cannot be surmised. As of 2013, shorebird hunting was unregulated in French Guiana (A. Levesque pers. comm. January 8, 2013; D. Mizrahi pers. comm. October 16, 2011). However, a ban on hunting all shorebird species has been proposed in French Guiana (CSRPN 2013), and the red knot was designated a protected species in October 2014 (C. Carichiopulo and N. de Pracontal pers. comm. October 10, 2014). Subsistence shorebird hunting was also common in northern Brazil, but has decreased in recent decades (Niles *et al.* 2008, p. 99).

We have no evidence that hunting was a driving factor in red knot population declines in the 2000s, or that hunting pressure is increasing. While only low to moderate red knot mortality is documented, additional undocumented mortality is likely. The findings of Watts (2010, p. 39) suggest that even moderate (hundreds of birds) direct human-caused mortality may begin to have population-level effects on the red knot. We do not have reliable information to reasonably know if hunting mortality is or was previously at this level in the Guianas, though we conclude it was likely much lower (tens of birds) in the Caribbean islands. In contrast, catch limits, handling protocols, and studies on the effects of research activities on survival all indicate that overutilization for scientific purposes is not a threat to the red knot (Niles *et al*. 2010a, p. 124; L. Niles and H. Sitters pers. comm. September 4, 2008; Niles *et al.* 2008, p. 100).

FACTOR C: From our review of best available data, we conclude that disease is not a threat to red knot populations. Predation pressures exacerbate other threats in some nonbreeding areas, but likely contribute little direct mortality. Natural cycles of high predation rates on the breeding grounds are not a threat to red knot populations, but disruption of these cycles from climate change, which may lead to prolonged periods of low productivity, is a threat to the red knot.

Red knots may be adapted to parasite-poor habitats and may, therefore, be susceptible to parasites when migrating or wintering in high-parasite regions (Piersma 1997, p. 623). However, we have no evidence that parasites have affected red knot populations beyond causing normal, background levels of mortality (D'Amico *et al*. 2008, pp. 193, 197; Harrington 2001, p. 21), and we have no indications that parasite infection rates or red knot fitness impacts are likely to increase. Therefore, we conclude that parasites are not a threat to the red knot. For the most prevalent viruses found in shorebirds within the red knot's geographic range (e.g., avian influenza, avian paramyxovirus), infection rates in red knots are low, and health effects are minimal or have not been documented (D. Stallknecht pers. comm. January 25, 2013; Maxted *et al.* 2012, pp. 322–323; Coffee *et al.* 2010, p. 484; Escudero *et al.* 2008, pp. 494–495; Niles *et al*. 2008, p. 101; D'Amico *et al.* 2007, p. 794). Therefore, we conclude that viral infections do not cause significant mortality and are not a threat to the red knot. However, we acknowledge an unlikely but potentially high-impact, synergistic effect among avian influenza, environmental contaminants, and climate change could produce a population-level impact in Delaware Bay.

Outside of the breeding grounds, predation is not directly effecting red knot populations despite some mortality (Niles *et al.* 2008, p. 28). At key stopover sites, however, localized predation pressures exacerbate other threats to red knot populations by pushing red knots out of otherwise suitable foraging and roosting habitats, causing disturbance, and possibly causing changes to stopover duration or other aspects of the migration strategy (Niles 2010a; Watts 2009b; Niles *et al.* 2008, pp. 101, 116; Lank *et al.* 2003, p. 303). In addition, predation pressure may induce sublethal physiological stress that can impact shorebird fitness (Clark and Clark 2002, p. 49). We expect the direct and indirect effects of predators to continue at the same level or decrease slightly over the next few decades.

Within the breeding range, normal 3- to 4-year cycles of high predation, mediated by rodent (e.g., lemming) cycles, result in years with extremely low reproductive output but do not threaten the survival of the red knot at the subspecies level (Niles *et al.* 2008, pp. 64, 101;

Meltofte *et al.* 2007, p. 20). It is believed shorebirds, such as red knots, have adapted to these cycles, therefore these natural cycles are not considered a threat to the red knot. What is a threat, however, is that these natural rodent/predator cycles are being disrupted by climate change, which may increase predation rates on shorebirds over the long term and have subspecies-level effects (Chapter 28 in IPCC 2014, p. 14; Fraser *et al.* 2013, pp. 13, 16; Brommer *et al*. 2010, p. 577; Ims *et al.* 2008, p. 79; Kausrud *et al.* 2008, p. 98). Disruptions in the rodent-predator cycle pose a substantial threat to the red knot, as they may result in prolonged periods of very low reproductive output (Meltofte *et al.* 2007, p. 22). Such disruptions have already occurred and may increase due to climate change (Chapter 28 in IPCC 2014, p. 14; Fraser *et al.* 2013, pp. 13, 16; Brommer *et al*. 2010, p. 577; Ims *et al.* 2008, p. 79; Kausrud *et al.* 2008, p. 98). The substantial impacts of elevated egg and chick predation on shorebird reproduction are well known (Smith and Wilson 2010, pp. 615, 621; Meltofte *et al.* 2007, p. 20), although the red knot's capacity to adapt to long-term changes in predation pressure is unknown (Meltofte *et al.* 2007, p. 34). The threat of persistent increases in predation in the Arctic may already be having subspecies-level effects (Fraser *et al.* 2013, p. 13) and is anticipated to increase into the future. Further, warming temperatures and changing vegetative conditions in the Arctic are likely to bring additional changes in the predation pressures faced by red knots, such as colonization by new predators from the south, though we cannot forecast how such ecosystem changes are likely to unfold.

FACTOR D: We have reviewed the adequacy of existing regulatory mechanism across the range of the red knot. In Canada, the Species at Risk Act provides protections for the red knot and its habitat, both on and off of Federal lands. The red knot is afforded additional protections under Canada's Migratory Birds Convention Act and by provincial law in four of the Provinces. Red knots are legally protected from direct take and hunting in several Caribbean and Latin American countries, but we lack information regarding the implementation or effectiveness of these measures. For many other countries, red knot hunting is unregulated, or we lack sufficient information to determine if red knot hunting is legal. We also lack information for countries outside the United States regarding the protection or management of red knot habitat, and regarding the regulation of other activities that threaten the red knot such as development, disturbance, oil spills, environmental contaminants, and wind energy development.

In the United States, the Migratory Bird Treaty Act and state wildlife laws protect the red knot from direct take resulting from scientific study and hunting. The Sikes Act, the National Park Service Organic Act, and the National Wildlife Refuge System Improvement Act provide protection for the red knot from habitat loss and inappropriate management on Federal lands. Section 404 of the Clean Water Act, the Rivers and Harbors Act, the Coastal Barrier Resources Act, the Coastal Zone Management Act, and State mechanisms regulate shoreline stabilization and development. State and local regulations provide varying levels of protection from impacts associated with beach grooming. Several Federal and State policies are in effect to stem the introductions and effects of invasive species, but collectively do not provide complete protection to the red knot from impacts to its habitats or food supplies resulting from beach or marine invaders or the spread of harmful algal species. Although threats to the horseshoe crab egg food resource remain, regulatory management of the horseshoe crab fishery under the Adaptive Resource Management (ARM) framework is adequate to address threats to the knot's Delaware Bay food supply from direct harvest. Regarding climate change, the U.S. Environmental Protection Agency (USEPA) has developed several initiatives related to greenhouse gasses

(GHGs). However, some of the USEPA's proposed GHG regulations are not yet final and, even when final, substantially greater reductions in GHGs would still be needed at multiple scales to reduce the magnitude of likely climate changes over the next several decades. Although we lack information regarding the overall effect of recreation management policies on the red knot, we are aware of a few locations in which beaches are closed, regulated, or monitored to protect nonbreeding shorebirds. Relatively strong Federal laws likely reduce risks to red knots from oil spills, but cannot fully abate the risk of oil spills and leaks. Similarly, Federal law and policy reduce the red knot's collision risks from new wind turbine development, but some level of mortality is expected upon buildout of the Nation's wind energy infrastructure.

FACTOR E: Based on our review of the best available scientific and commercial data, the red knot faces subspecies-level impacts from other natural and manmade factors that are already occurring and are anticipated to continue and possibly increase into the future.

Reduced food availability at the Delaware Bay stopover site due to commercial harvest of the horseshoe crab is considered a primary causal factor in the decline of rufa red knot populations in the 2000s (Escudero *et al.* 2012, p. 362; McGowan *et al.* 2011a, pp. 12–14; Niles *et al.* 2008, pp. 1–2; Baker *et al.* 2004, p. 875). Under the current management framework (the ARM), the present horseshoe crab harvest is not considered a threat to the red knot. However, continued implementation of the ARM is imperiled by lack of funding to support the requisite monitoring programs. With or without the ARM, it is not yet known if the horseshoe crab egg resource will continue to adequately support red knot population growth over the next decade. Notwithstanding the importance of the horseshoe crab and Delaware Bay, the red knot faces a range of ongoing and emerging threats to its food resources throughout its range, including small prey sizes from unknown causes (Escudero *et al.* 2012, pp. 359–362; Espoz *et al.* 2008, pp. 69, 74), warming water and air temperatures (Jones *et al.* 2010, pp. 2255–2256), ocean acidification (International Geosphere-Biosphere Programme (IGBP) *et al.* 2013, pp. 9, 16; NRC 2010b, pp. 68–69), physical habitat changes (Chapter 5 in IPCC 2014, p. 21; Rehfisch and Crick 2003, p. 88; Najjar *et al.* 2000, p. 225), possibly increased prevalence of disease and parasites (Ward and Lafferty 2004, p. 543), marine invasive species (Seebens *et al.* 2013, p. 782; Ruesink *et al.* 2005, pp. 671–674; Grosholz 2002, p. 22–23), and burial and crushing of invertebrate prey from sand placement and recreational activities (Sheppard *et al.* 2009, p. 113; Schlacher *et al.* 2008b, pp. 345, 348; Schlacher *et al.* 2008c, pp. 878, 882; Greene 2002, p. 24).

In addition, the red knot's life-history strategy makes this species inherently vulnerable to mismatches in timing between its annual cycle and those periods of optimal food and weather conditions upon which it depends (Galbraith *et al.* 2014, p. 7 and Supplement 1; Liebezeit *et al.* 2014, p. 2; Conklin *et al.* 2010, p. 4; Gill *et al.* 2013, p. 1; Hurlbert and Liang 2012, pp. 4–5; McGowan *et al*. 2011a, pp. 2, 16; Smith *et al.* 2011a, p. 575; Meltofte *et al.* 2007, p. 36). The red knot's sensitivity to timing asynchronies has been demonstrated through a population-level response, as the late arrivals of birds in Delaware Bay is generally accepted as a key causative factor (along with reduced supplies of horseshoe crab eggs) behind population declines in the 2000s (Baker *et al*. 2004, p. 878). The factors that caused delays in the spring migrations of red knots from Argentina and Chile are still unknown (Niles *et al*. 2008, p. 2), and we have no information to indicate if this delay will reverse, persist, or intensify in the future. Superimposed on the existing threat of late arrivals in Delaware Bay are new threats emerging due to climate change (Summary for Policymakers in IPCC 2014, p. 30; Root *et al.* 2013, pp. 85–88; Hurlbert and Liang 2012, p. 4), such as changes in the timing of reproduction for both horseshoe crabs and mollusks (Burrows *et al.* 2011, p. 652; Poloczanska *et al.* 2013, pp. 3–4; Smith *et al.* 2010b, p. 563; van Gils *et al.* 2005a, p. 2615; van Gils *et al.* 2005b, pp. 126–127; Philippart *et al.* 2003, p. 2171). Climate change may also cause shifts in the period of optimal arctic insect and snow conditions relative to the time period when red knots currently breed (Grabowski *et al.* 2013, p. 1097; McGowan *et al*. 2011a, p. 13; Smith *et al*. 2010a, p. 292; Tulp and Schekkerman 2008, p. 48; Meltofte *et al.* 2007, pp. 7, 25; Piersma *et al.* 2005, p. 270; Schekkerman *et al.* 2003, p. 340). The red knot's adaptive capacity to deal with numerous changes in the timing of resource availability across its geographic range is largely unknown (Liebezeit *et al.* 2014, pp. 1, 10; Grabowski *et al.* 2013, p. 1103; Meltofte *et al.* 2007, p. 34). A few examples suggest some flexibility in red knot migration strategies (D. Newstead pers. comm. May 8, 2014; Grabowski *et al.* 2013, pp. 1097, 1100–1103; Smith *et al*. 2010a, p. 292; González *et al.* 2006, p. 115; González *et al.* in International Wader Study Group (IWSG) 2003, p. 18), but differences between the annual timing cues of red knots (at least partly celestial and endogenous) (Liebezeit *et al.* 2014, p. 10; Conklin *et al.* 2010, p. 5; Gill *et al.* 2013, p. 1; McGowan *et al.* 2011a, p. 16; Cadée *et al.* 1996, p. 82) and their prey (primarily environmental) (Smith *et al.* 2010b, p. 563; Philippart *et al.* 2003, p. 2171) suggest there are limitations on the adaptive capacity of red knots to cope with increasing frequency or severity of asynchronies.

Other factors are likely to exacerbate the effects of reduced prey availability and asynchronies, including human disturbance (Burger and Niles 2013a, p. 23; Burger and Niles 2013b, p. 657; Escudero *et al.* 2012, pp. 358, 362), competition with gulls (Niles *et al.* 2008, p. 107; Burger *et al.* 2007, p. 1162), and behavioral changes from wind energy development (Kuvlesky *et al.* 2007, p. 2489). Additional factors are likely to increase the levels of direct red knot mortality, such as harmful algal blooms (HABs) (Newstead 2014a, p. 23; Anderson 2007, p. 2), oil spills (Anderson *et al*. 2012, p. 10; WHSRN 2012; Kalasz 2008, pp. 39–40; Niles *et al.* 2008, p. 98, 100), and collisions with wind turbines (D. Newstead pers. comm. March 5, 2013; Burger *et al*. 2012c, p. 370; Burger *et al.* 2011, p. 348; Watts 2010, p. 1; Kuvlesky *et al.* 2007, p. 2487). In addition to elevating background mortality rates, these three factors pose the potential for a low-probability but high-impact event if a severe HAB or major oil spill occurs when and where large numbers of red knots are present, or if a mass-collision event occurs at wind turbines during migration.

CUMULATIVE EFFECTS AND CONCLUSION: Red knots face a wide range of threats across their range on multiple geographic and temporal scales. The effects of some smaller threats may act in an additive fashion to ultimately impact populations or the subspecies as a whole (cumulative effects). Other threats may interact synergistically to increase or decrease the effects of each threat relative to the effects of each threat considered independently (synergistic effects). For example, reduced food availability has been shown to interact synergistically with asynchronies and several other threats, such as asynchronies, disturbance, predation pressure, and competition with gulls (Escudero *et al.* 2012, p. 362; Dey *et al.* 2011a, pp. 7, 9; Breese 2010, p. 3; Niles *et al*. 2008, p. 2; Atkinson *et al.* 2007, p. 892; Niles *et al.* 2005, p. 4; Baker *et al.* 2004, p. 878). We conclude that a number of threats are likely contributing to habitat loss, anthropogenic mortality, or both, and thus contribute to the red knot's threatened status, particularly considering the cumulative and synergistic effects of these threats, and that several key populations of this species have already undergone considerable declines.

SPECIES INFORMATION

SPECIES DESCRIPTION

The rufa red knot (*Calidris canutus rufa*) is a medium-sized shorebird about 9 to 11 inches (in) (23 to 28 centimeters (cm)) in length. (Throughout this document, "rufa red knot," "red knot," and "knot" are used interchangeably to refer to the *rufa* subspecies. "*Calidris canutus*" and "*C. canutus*" are used to refer to the species as a whole or to birds of unknown subspecies. References to other particular subspecies are so indicated by use of the Latin name.) The red knot migrates annually between its breeding grounds in the Canadian Arctic and several wintering regions, including the Southeast United States (Southeast), the Northeast Gulf of Mexico, northern Brazil, and Tierra del Fuego at the southern tip of South America (figures 1 and 3). During both the northbound (spring) and southbound (fall) migrations, red knots use key staging and stopover areas to rest and feed (figure 4). This annual cycle is described in more detail below.

The red knot is easily recognized during the breeding season by its distinctive rufous (red) plumage (feathers). The face, prominent stripe above the eye, breast, and upper belly are a rich rufous-red to a brick or salmon red, sometimes with a few scattered light feathers mixed in. The feathers of the lower belly and under the tail are whitish with dark flecks. Upperparts are dark brown with white and rufous feather edges; outer primary feathers are dark brown to black (Harrington 2001, p. 2; Davis 1983, p. 372). Females are similar in color to males, though the rufous colors are typically less intense, with more buff or light gray on the dorsal (back) parts (Niles *et al.* 2008, p. 13). Subtle subspecies differences (see Taxonomy, below) in breeding plumage have been described. The intensity of rufous coloration is paler in *Calidris canutus rufa* than in other subspecies (Tomkovich 1992, p. 20). Red knots have a proportionately small head, small eyes, and short neck, and a black bill that tapers from a stout base to a relatively fine tip. The bill length is not much longer than head length. Legs are short and typically dark gray to black, but sometimes greenish in juveniles or older birds in nonbreeding plumage (Harrington 2001, p. 2). Nonbreeding plumage is dusky gray above and whitish below. Juveniles resemble nonbreeding adults, but the feathers of the scapulars (shoulders) and wing coverts (small feathers covering base of larger feathers) are edged with white and have narrow, dark bands, giving the upperparts a scalloped appearance (Davis 1983, p. 372). Body mass varies seasonally, with lowest average mass during early winter (4.4 ounces (oz); 125 grams (g)) and highest mean values during spring (7.2 oz; 205 g) and fall (6.1 oz; 172 g) migration (Harrington 2001, p. 12).

RESEARCH METHODS

Best available data regarding the red knot are generated by several different research methods summarized here and referenced throughout this document. For example, in many wintering and stopover areas, red knots are routinely counted by surveyors in aircraft or on the ground. In some areas, data from both ground and aerial counts are available. Although the results can be similar, data from these different methods should be compared with caution. Laursen *et al.* (2008) compared aerial and ground survey methods and found that, across all waterbird species evaluated, significantly greater numbers were identified from the ground than from aircraft. For shorebirds that were numerous and widespread and occurred in flocks (and therefore similar to red knots), there was a moderate correspondence between the two methods—

aerial surveys detected greater than 55 percent of the ground counts, and the mean bird densities derived from ground counts differed from aerial counts by less than 30 percent (Laursen *et al.* 2008, p. 165). Within the same survey type (ground or aerial), we attach greater confidence to apparent population trends when the survey team and methods have been consistent over time.

In addition, an international team of scientists have marked tens of thousands of shorebirds since the mid-1990s, including red knots (BandedBirds.org 2012). About 1,000 red knots per year are trapped for scientific study in Delaware Bay, and about 300 in South America (Niles *et al.* 2008, p. 100). Additional birds are trapped in some years in other parts of the range. The legs of the trapped birds are marked with bands and with individually numbered, color-coded flags (one color per country). By tracking where the marked birds are observed in future seasons and years, researchers can draw inferences about wintering and migration areas, migration routes and timing, life history, regional population sizes, and survival rates.

Through 2008, about 50 of the birds caught in Delaware Bay each year were the subject of radio-telemetry studies in which a radio transmitter was glued to the back of each bird (Niles *et al.* 2008, p. 100). Relative to resighting of marked birds, radio tracking allows for more direct observations of bird movements within the area. However, because the radio tags drop off after 1 to 2 months (Niles *et al.* 2008, p. 100), birds can be observed only during the same season that the transmitter was attached. Satellite transmitters that can remotely send locational data over long distances and time frames are currently too heavy for use on red knots (J. Cohen and B. Watts pers. comm. October 4, 2012), although technological advances could make them lighter in the future. Information on where migrant red knots spent the previous winter can also be discerned from chemical analysis that produces a stable isotope “signature” from the feathers, which reflects the geographic region in which the birds were feeding when those feathers were grown.

Since 2009, researchers have gained many new insights into red knot migration and life history using light-sensitive geolocators, a small device attached to the bird’s leg. These instruments record periodic, time-stamped, ambient light levels that can be used to determine geographic location. Geolocators are small enough to be used on the legs of medium-sized shorebirds such as red knots. However, the birds must be recaptured to access the data, so return rates are low. Although geolocators record data for only about a year, the data are still retrievable for up to 20 years if birds are recaptured (Niles *et al.* 2010a, pp. 123–124). Based on light-dark periods indicating day length, the latitude, longitude, and duration of stopovers can be estimated. Flight segments connecting the stopovers can be inferred based on the time between stops and general knowledge of shorebird migration (e.g., Cornell Lab of Ornithology 2007; Alerstam *et al.* 2001, entire). The precision of the flight segments is limited, as they must be inferred based on points where birds remained in one place for at least an entire day (Normandeau Associates, Inc. 2011, p. 77). Even at such stopping points, the amount of geospatial imprecision ranges from 31 to 186 miles (mi) (50 to 300 kilometers (km)) depending on the latitude (higher latitudes render more precise readings, except at very high latitudes where there are not 24-hour light-dark cycles) and other conditions (e.g., shading of the geolocator) (Normandeau Associates, Inc. 2011, p. 77). In addition to light levels, many geolocators record measures of conductance, indicating whether the bird is in contact with salt water (Burger *et al.* 2012a, p. 28).

An important caveat to use of geolocator data relates to the scope of inference. Since these units must be recovered to yield data (i.e., they do not "transmit" data), they are likely to be recovered only from birds that revisit, and spend a fair amount of time in, the site where they were originally deployed, or another site where a collaborating researcher may encounter them. While the geolocators recovered to date have yielded a great deal of information about the migratory strategies of birds using certain well-known stopover or wintering areas, we would have little or no data on birds that used different migration strategies (e.g., if they spent much of their time in poorly known nonbreeding habitats where they would not have been detected by researchers) (D. Newstead pers. comm. May 8, 2014).

TAXONOMY

Calidris canutus is classified in the Class Aves, Order Charadriiformes, Family Scolopacidae, Subfamily Scolopacinae (American Ornithologists Union (AOU) 2012a). Six subspecies are recognized, each with distinctive morphological traits (i.e., body size and plumage characteristics), migration routes, and annual cycles. Each subspecies is believed to occupy a distinct breeding area in various parts of the Arctic (Buehler and Baker 2005, pp. 498–499; Tomkovich 2001, pp. 259–262; Piersma and Baker 2000, p. 109; Piersma and Davidson 1992, p. 191; Tomkovich 1992, pp. 20–22), but some subspecies overlap in certain wintering and migration areas (Conservation of Arctic Flora and Fauna (CAFF) 2010, p. 33; see Subspecies Nonbreeding Distributions, below).

Birds in the Order Charadriiformes are commonly called shorebirds, and include the plovers (Family Charadriidae). Birds in the Family Scolopacidae are commonly known as sandpipers or snipes. Historically, Roosevelt (1866, pp. 91–93) reported considerable confusion around the names of commonly hunted shorebirds, which he referred to generally as bay-snipe. Many different scientific and common names have been used for the rufa red knot since the early 1800s. Scientific names for the rufa red knot appearing in the historical literature include *Tringa canutus, T. rufa, T. islandica, T. cinerea, Canutus canutus*, and *Calidris canutus rufus*. The following common names for the rufa red knot have been used by scientists, naturalists, and hunters, although many of these names have also been used to refer to a variety of other shorebird species: red-breasted sandpiper, ash-colored sandpiper, robin snipe, robin-breasted snipe, red-breasted snipe, gray snipe, white robin snipe, red-breasted plover, rosy plover, blue plover, silver plover, gray red-breasted plover, red-breast, buff-breast, gray-back, silverback, whiting, wahquoit, beach robin, knot, red knot, American knot, and Western Atlantic knot (AOU 2012b; Harrington *et al.* 2010a, p. 191; Lowery 1974, p. 308; Urner and Storer 1949, p. 185; Hellmayr and Conover 1948, p. 166; Stone 1937, p. 456; Bent 1927, pp. 131–132; Forbush 1925, p. 402; Ridgway 1919, p. 231; Forbush 1912, p. 262; Eaton 1910, p. 307; Shriner 1897, p. 94; Dixon 1895 in Barnes and Truitt 1997, pp. 113–114; Mackay 1893, p. 25; Stearns and Coues 1883, p. 229; Hallock 1877, p. 168; Coues 1868, p. 293; Roosevelt 1866, p. 151; Herbert 1853, p. 160; Audubon 1844, plate 328; Giraud 1844, p. 224; Wilson 1829, p. 140).

Four genetically distinct groups of *Calidris canutus* have been identified. Three of the groups correspond to recognized subspecies: *C. canutus canutus, C.c. piersma, C.c. rogersi*. The fourth is a North American group containing the other three recognized subspecies (*C.c. rufa, C.c. roselaari* and *C.c. islandica*), which are not fully distinct at the genetic level based on analyses conducted to date (discussed further below) (Buehler and Baker 2005, p. 502). Based on low overall genetic variability, *C. canutus* is thought to have recently survived a genetic bottleneck. On the scale of evolutionary time, the subspecies groups are estimated to have diverged very recently, within the past 20,000 years. The North American group is estimated to have diverged from a Siberian ancestor about 12,000 years ago, then split into the three recognized North American subspecies within the past 5,500 years (Buehler and Baker 2005, p. 505). Buehler *et al.* (2006, p. 485) estimated that populations of the three North American breeding subspecies may have diverged even more recently, within about the last 1,000 years.

Because of the evolutionarily recent divergence times and low overall genetic variability, genetic distances among subspecies are small and not fully distinct. However, it is important to consider morphological and ecological differences along with the genetic evidence, particularly when taxa have only recently diverged (Committee on the Status of Endangered Wildlife in Canada (COSEWIC) 2007, p. 9; Buehler and Baker 2005, pp. 507–508). In addition, the results of the genetic analysis conducted by Buehler and Baker (2005) may have been affected by these authors' attribution of birds in the Southeast United States to *Calidris canutus roselaari* (Buehler and Baker 2005, p. 498), as was widely believed at the time. In that study, all the birds sampled as the *roselaari* subspecies were from the Southeast (Buehler and Baker 2005, p. 511). As discussed further below, newer evidence now suggests that birds in the Southeast are *C.c. rufa* (U.S. Fish and Wildlife Service (USFWS or Service) 2011a, p. 305; Niles *et al.* 2008, p. 132). The Service and our partners are investigating *C. canutus* genetics to better assess the relationship between the *rufa* and *roselaari* subspecies and population structure within *rufa*; results are expected within the next few years.

Calidris canutus canutus, *C.c. piersma*, and *C.c. rogersi* do not occur in North America. The subspecies *C.c. islandica* breeds in the northeastern Canadian High Arctic and Greenland, migrates through Iceland and Norway, and winters in western Europe (COSEWIC 2007, p. 4). *Calidris c. rufa* breeds in the central Canadian Arctic (just south of the *C.c. islandica* breeding grounds) and winters along the Atlantic coast and the northern Gulf of Mexico coast (Gulf coast) of North America, in the Caribbean, and along the north and southeast coasts of South America including the island of Tierra del Fuego at the southern tip of Argentina and Chile (figures 1 and 3); detailed information on the distribution of rufa red knots is provided in the sections that follow. *Calidris c. islandica* breeds just north of *C.c. rufa*; the southern limit of the *C.c. islandica* breeding range and the northern limit of the *C.c. rufa* breeding range (and thus the potential for any overlap) are poorly known (CAFF 2010, p. 33; COSEWIC 2007, p. 12; Morrison and Harrington 1992, p. 73) (see Breeding Distribution, below). Resightings of three marked birds have documented infrequent movements between the *C.c. islandica* and *C.c. rufa* migratory flyways. It is unknown if any of the three birds permanently changed flyway or breeding area. However, these were clearly atypical movements, probably of vagrant individuals (Wilson *et al.* 2010, entire).

Subspecies *Calidris canutus roselaari* breeds in western Alaska and on Wrangel Island in eastern Russia (Carmona *et al.* 2013, p. 169; Buehler and Baker 2005, p. 498) and winters on the Pacific coast from northern Mexico through Panama and possibly farther south (D. Newstead pers. comm. February 13, 2014; Carmona *et al.* 2013, pp. 171, 175). As detailed below (Subspecies Nonbreeding Distributions), best available data are limited but suggest that the nonbreeding ranges of *C.c. roselaari* and *C.c. rufa* overlap, at least in Texas during spring and in Panama during winter (D. Newstead pers. comm. May 13, 2014; D. Newstead pers. comm.

February 13, 2014; D. Newstead pers. comm. February 11, 2014; D. Newstead pers. comm. August 20, 2012). However, geolocator data that have become available since 2010 confirm the existence of distinct breeding areas for the *rufa* and *roselaari* subspecies (Niles 2014; S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014; Bimbi *et al.* 2014, pp. 29–31; D. Newstead pers. comm. May 8, 2014; D. Newstead pers. comm. February 13, 2014; L. Niles pers. comm. January 4, 2013; Newstead *et al.* 2013, p. 56; Niles *et al.* 2012a, pp. 197–200; Niles 2012a; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, pp. 125–126).

LONGEVITY AND SURVIVAL

The oldest *Calidris canutus* of any subspecies recorded worldwide was estimated to be 25 years old when recaptured (Niles *et al.* 2008, p. 28). The oldest recorded rufa red knot (marked with the orange flag code B95) was at least 21 years old when last resighted in spring 2014 (Bauers 2014; Jordan 2014). Although these records indicate that the potential lifespan of a *C. canutus* is considerable, the average life span is thought to be much less. Based on estimated survival rates for a stable population, few red knots live for more than about 7 years (Niles *et al.* 2008, p. 28). No rigorous longevity analysis has been done since the 7-year estimate, but an update may be warranted based on a number of birds known (from resightings) to be considerably older than 7 years of age. Of 1,377 red knots marked in 2003, 83 (6 percent) were resighted between January 2013 and March 2014. Of 1,104 knots marked in 2004, 86 (8 percent) were resighted between January 2013 and March 2014 (J. Parvin pers. comm. March 14, 2014). As many of these birds would have already been adults (i.e., at least 2 years old) when marked, they were at least 12 or 13 years old when resighted during this recent window. It is unlikely all surviving birds from those 2003 and 2004 cohorts would have been spotted during the 2013-2014 window (which was used in a query of the BandedBirds.org database). Thus, the actual percent of surviving birds from these 2003 and 2004 cohorts is likely higher than 6 or 8 percent. As of March 2014, the BandedBirds.org database contained records of 332 red knots that lived (some still living) to be at least 10 years old, 121 at least 12 years old, 20 at least 14 years old, 5 at least 16 years old, and 2 at least 17 years old (J. Parvin pers. comm. March 14, 2014). In addition to B95, another orange-flagged knot (code YY1) was at least 18 when observed in 2014 (Jordan 2014).

Niles *et al.* (2008, p. 63) characterized red knots as being typical of shorebird species that exhibit low fecundity, delayed maturity, and high annual survival. Age of first breeding is uncertain but for most birds is probably at least 2 years (Harrington 2001, p. 21). Indeed, the first geolocator results from a juvenile bird show it attempted to breed during its second summer (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014). Although little is known about the survival rate of juvenile red knots or the rate of recruitment into the adult population, the Delaware Department of Natural Resources and Environmental Control (DDNREC 2013) has suggested that low juvenile survival and recruitment may be currently limiting population growth, based on the fact that annual adult survival (i.e., as estimated by McGowan *et al.* 2011a, p. 13) is high while the population (i.e., stopover counts at Delaware Bay) has been stable over the past few years (see Population Surveys and Estimates, below).

From resightings of marked birds, survival estimates have been calculated for several particular rufa red knot wintering and stopover areas. Based on resightings of marked birds, Harrington *et al.* (1988, pp. 442–443) reported survival rates of 35.8 percent for knots banded in

Massachusetts during fall migration, 34.0 percent for birds banded in Delaware Bay during spring migration, and 75.8 percent for birds banded in Florida during winter. However, these survival rates do not account for emigration (failure of surviving birds to return to the banding site in subsequent years); thus, differences among locations could imply differences in year-to-year site fidelity rather than (or in addition to) differences in survival (Harrington *et al.* 1988, p. 442).

For birds stopping in Delaware Bay in spring, Atkinson *et al.* (2002, p. 4) estimated survival at 80.5 percent for the period 1997 to 2001, although the 95 percent confidence interval was large (44.4 to 95.5 percent). Baker *et al*. (2004, pp. 878–897) estimated adult survival rates for Delaware Bay at 84.6 percent from 1994 to 1998, but only 56.4 percent from 1998 to 2001. With a longer data set, 1997 to 2008, McGowan *et al.* (2011a, p. 13) calculated a survival rate of about 92 percent for Delaware Bay.

For birds wintering in Florida, Schwarzer *et al.* (2012, p. 729) found an average annual adult survival rate of 89 percent, with the 95 percent confidence interval overlapping the 92 percent survival estimate from McGowan *et al.* (2011a). Given that similar survival rates have been calculated for Delaware Bay migrants, most of which winter in South America (P. Atkinson pers. comm. November 8, 2012), as for Florida-wintering birds, Schwarzer *et al.* (2012, p. 729) concluded that factors influencing adult survival likely do so where populations of red knots from different wintering regions co-occur, such as along the Atlantic migratory flyway or on the breeding grounds (i.e., important survival factors operating in the wintering areas would be expected to produce differential survival rates among different wintering regions).

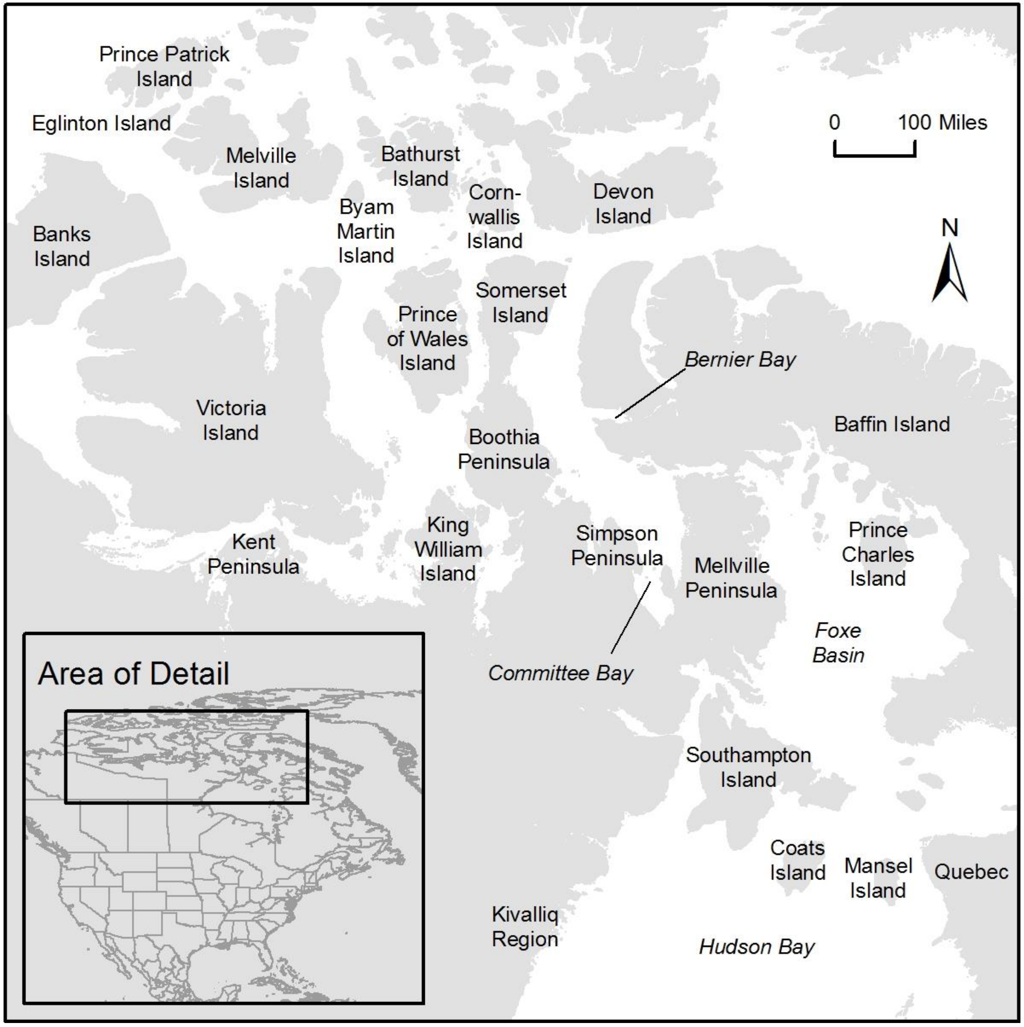
BREEDING

Breeding Distribution

The red knot breeds in the central Canadian Arctic (figure 1), primarily in Nunavut Territory, but with some potential breeding habitat extending into the Northwest Territories. Breeding red knots have been documented via telemetry on King William Island and Boothia Peninsula, and on the shorelines and islands of Committee Bay (Simpson Peninsula), Foxe Basin, and Hudson Bay, including Melville Peninsula and Baffin, Prince Charles, Southampton, Coats, and Mansel Islands (Niles *et al.* 2008, pp. 15–16). Niles *et al.* (2008, p. 16) and Morrison and Harrington (1992, p. 73) also include southern Victoria Island in the known breeding range. See Factor A—Breeding Habitat for information on the eco-regional classification of the red knot's breeding range.

Based on habitat modeling, the predicted breeding range of *Calidris canutus rufa* is larger than the documented range discussed above, extending farther east to southern Baffin Island, farther west to northern Victoria and Banks Islands, and farther north through Somerset, Prince of Wales, and western Baffin (around Bernier Bay) Islands up to the southern Queen Elizabeth Islands of Eglinton, Melville, Byam Martin, Bathurst, Cornwallis, and Devon (Niles *et al.* 2008, p. 16). Some of these potential *C.c. rufa* breeding areas are within the breeding range of *C.c. islandica*, as it was mapped by Morrison and Harrington (1992, p. 73). However, CAFF (2010, p. 33) presented a different range for *C.c. islandica* that shows only minimal overlap (e.g., on the northern tip of Melville Island) with the potential *C.c. rufa* breeding range predicted by Niles *et* *al.* (2008, p. 16). It is not known whether there is any overlap between the actual breeding ranges of these two subspecies (Morrison and Harrington 1992, p. 73), largely due to limitations on research methods (satellite transmitters are too heavy for red knots to carry; geolocators cannot be used to determine location at very high latitudes) and accessibility (field studies of breeding *C. canutus* in North America have been relatively few and localized because the birds are thinly distributed in remote areas across a huge region of the Arctic).

Figure 1. Known and potential breeding range of the red knot



Some shorebird surveys have been conducted within and adjacent to the predicted breeding range of the rufa red knot (as mapped by Niles *et al.* 2008, p. 16), in areas where breeding *Calidris canutus rufa* have not been confirmed to date. One pair of *C. canutus* (subspecies unknown) was observed on Somerset Island in summer 2001 (Bart and Johnston 2012, p. 146). Nine *C. canutus* were observed June 10 to 28, 2008, in the Kivalliq region on the west coast of Hudson Bay (Bart and Johnston 2012, pp. 152, 154). The Kivalliq region is south of the known *C.c. rufa* breeding range and only a few hundred miles north of a known rufa red knot migration stopover at the Nelson River delta; it is unknown if the birds were migrating or breeding in this area. No *C. canutus* were observed during small-scale shorebird surveys of the Kent Peninsula, northwest Quebec, or the southwestern Queen Elizabeth Islands (Prince Patrick, Eglinton, and Melville) in the 2000s (Bart and Johnston 2012, pp. 141–155). No *C. canutus* were observed in northern Victoria Island or Banks Island during extensive surveys in the 1950s (Morrison and Harrington 1992, p. 73).

New geolocator information suggests that red knots are indeed nesting within the known or predicted breeding range (Newstead *et al.* 2013, p. 56; Niles *et al.* 2012a, pp. 197–200; Niles 2012a; Niles *et al.* 2010a, pp. 125–126), although information about the breeding range that can be discerned from geolocators is very general in nature. Daylight is continuous during the middle of the arctic summer, so geolocation by monitoring light levels is not possible at very high latitudes (Burger *et al*. 2012a, p. 28). However, some breeding birds arrive or depart close enough to the Arctic Circle where there is a sufficient daily light variation to allow some estimation of breeding location (Burger *et al*. 2012a, p. 28). Despite these limitations, geolocator data are sufficient to suggest a general trajectory (i.e., to or from central Canada, as opposed to Alaska or Russia). Thus, geolocator results are a key piece of evidence indicating that the birds wintering along the north coast of Brazil, along the Gulf coasts of Texas and Florida, and in other parts of the Southeast are *Calidris canutus rufa*. For birds wintering in these areas, all geolocator results to date show migration flight trajectories to and from *C.c. rufa* breeding areas in central Canada, and none show trajectories to or from *C.c. roselaari* breeding areas in Alaska or Russia (Niles 2014; S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014; Bimbi *et al.* 2014, pp. 29–31; D. Newstead pers. comm. May 8, 2014; L. Niles pers. comm. January 4, 2013; Newstead *et al.* 2013, p. 56; Niles *et al.* 2012a, pp. 197–200; Niles 2012a; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, pp. 125–126).

The extent to which rufa red knots from different wintering areas mix on the breeding grounds, and therefore potentially interbreed, is poorly known (Harrington *et al.* 1988, p. 443). Limited evidence from one nesting area, Southampton Island, suggests that mixing does occur in this location. Stable isotope signatures indicate that some of the red knots nesting in this area winter in northern South America or the Southeast. Birds that had been marked in a different wintering region, southern Argentina, have also been seen on the same island (Niles *et al.* 2008, p. 54). Geolocator results also suggest that red knots from the Southeast and Argentina-Chile wintering regions both breed on Southampton Island (Niles 2011b) (an inference that may be possible because Southampton Island is just below the Arctic Circle). While limited stable isotope and geolocator data may point to potential interbreeding, further data suggest otherwise. Red knots from different wintering areas exhibit morphological, particularly size, differences. On average and controlling for factors like sex and molt status, birds from Tierra del Fuego are significantly smaller (bill length, mass) than those from Brazil or Florida (Niles *et al.* 2008, pp. 9–11). These size differences suggest that red knots from different wintering areas may have discrete breeding areas, but the available information is insufficient to draw any conclusions on this question. In an earlier study, Harrington *et al.* (1988, p. 441) found no statistically significant differences in wing or bill lengths of red knots caught in Florida versus those caught in Argentina.

Breeding Habitat and Food

Red knots generally nest in dry, slightly elevated tundra locations, often on windswept slopes with little vegetation. Breeding areas are located inland, but near arctic coasts. Nests may be scraped into patches of mountain avens (*Dryas octopetala*) plants, or in low spreading vegetation on hummocky (characterized by knolls or mounds) ground containing lichens, leaves, and moss. On Southampton Island, nests are located in exposed areas of glacial/shattered rocks and mudboils (bare patches that form on silt or clay soils), and most are located in proximity to suitable wetland foraging areas (Niles *et al.* 2008, p. 27; Harrington 2001, p. 8). After the eggs hatch, red knot chicks and adults quickly move away from high nesting terrain to lower, wetland habitats (Niles *et al.* 2008, p. 27).

In 1999, Niles *et al.* (in Baker 2001, p. 14) found eight previously radio-tagged red knots on Southampton Island. All tagged knots used patches of low-elevation barren tundra within 31 mi (50 km) of the ocean coast. In 2000, these authors found 11 nests. All nests were scraped within small patches of *Dryas* spp. (average patch size of 20 in (51 cm) across) in sparsely vegetated tundra (only 29 percent vegetated within 33 feet (ft) (10 meters (m)) of the nest). Nests were spaced approximately 0.6 mi (1 km) apart on sparsely vegetated, narrow ridges less than 33 ft (10 m) high. All nests occurred within 590 ft (180 m) of an isolated, freshwater wetland. All adults were followed via radio telemetry; birds stayed within 1.2 mi (2 km) of the nest area, appeared to defend an area 0.3-mi (0.5-km) around the nest, and shared wetlands outside that area (Niles *et al.* in Baker 2001, p. 14). Recent findings suggest that knots rarely enter salt water while in the Arctic (Burger *et al.* 2012a, p. 26; Niles *et al.* 2008, p. 61), although two of eight rufa red knots located via aerial radio tracking on Southampton Island in 1999 were found in coastal wetlands (Niles *et al.* 2008, p. 16).

On the breeding grounds, the red knot's diet consists mostly of terrestrial invertebrates such as insects and other arthropods. However, early in the breeding season, before insects and other macroinvertebrates are active and accessible, *Calidris canutus* will eat grass shoots, seeds, and other vegetable matter (Harrington 2001, p. 11). *Calidris canutus* is able to feed in shallow thawed sod in the Arctic at times when insects are rare and slow-moving due to low-temperature conditions (Gerasimov 2009, p. 227).

Breeding Chronology and Success

The breeding chronology of the rufa red knot is poorly known (Niles *et al.* 2008, p. 25). Other *Calidris canutus* subspecies may be paired or unpaired on arrival in breeding areas in late May or early June, and the start of breeding in *C. canutus* varies with snowmelt conditions. Although males tend to predominate among early arrivals, simultaneous arrival of male and female *C. canutus islandica* has been noted from May to early June, followed by movement into inland nesting habitats within a few days (Niles *et al.* 2008, pp. 25–26). Flocks of *C. canutus* sometimes arrive at breeding latitudes before snow-free habitat is available. Upon arrival, or as soon as favorable conditions exist, male and female *C. canutus* occupy breeding habitat, and territorial displays begin (Harrington 2001, p. 16). In rufa red knots, pair bonds form soon after arrival on breeding grounds and remain intact until shortly after the eggs hatch (Niles *et al.* 2008, p. 25). Female rufa red knots lay only one clutch (group of eggs) per season, and, as far as is known, do not lay a replacement clutch if the first is lost. The usual clutch size is four eggs, though three-egg clutches have been recorded. The incubation period lasts approximately 22 days from the last egg laid to the last egg hatched, and both sexes participate equally in egg incubation. Young are precocial, leaving the nest within 24 hours of hatching and foraging for themselves (Niles *et al.* 2008, p. 27). No information is available regarding chick survival rates (Niles *et al.* 2008, p. 28). Females are thought to leave the breeding grounds and start moving south soon after the chicks hatch in mid-July. Thereafter, parental care is provided solely by the males, but about 25 days later (around August 10) males also abandon the newly fledged juveniles and move south. Not long after, they are followed by the juveniles (Niles *et al.* 2008, p. 14). Niles *et al.* (2008, p. 26) reported that no published information was available for the rufa red knot on mate fidelity between years. In one study site on Southampton Island, Niles *et al.* (2008, p. 26) observed only limited evidence of site fidelity despite the fact that studies of other subspecies suggest breeding site fidelity is high, especially in males.

Data from 19 rufa red knots (adults of undetermined sex) fitted with geolocators show that the median arrival date in the Arctic was June 10, the median departure date was July 22, and the mean time in the Arctic was 44 ± 2.3 days (range 28 to 65 days) (Burger *et al.* 2012a, p. 26). Geolocator data have also been used to estimate incubation periods and infer hatching success. Because of 24-hour sunlight during the arctic summer, a prolonged duration of a mainly dark signal indicates that a geolocator was not exposed to the ambient light, and a period of nest incubation can be inferred because the bird would be sitting on the leg-mounted device. Of 20 geolocator records from the breeding grounds (2 from the same bird in successive years), 17 records (85 percent) indicated initiation of incubation. Of those 17 records, 11 birds (65 percent) incubated for 18 to 24 days, suggesting successful hatching of chicks. Three birds (18 percent) incubated for 9 or 10 days, suggesting nest failure due to abandonment or predators. One bird that incubated for 30 days probably laid infertile eggs, and the nesting success of the two remaining birds could not be inferred from the geolocator data (Burger *et al.* 2012a, pp. 34–35). After incubation ended, red knots remained in the Arctic for 1 to 21 days. The sex of the birds was not determined, so it was not possible to verify the presumed gender differences in departure timing (Burger *et al.* 2012a, pp. 31, 33). Geolocator data from one bird trapped as a juvenile show that it migrated to the Arctic during its second summer and incubated, but the incubation period was shopped short (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014), suggesting an unsuccessful breeding attempt.

Data from seven red knots (some with 2 years of data) that wintered in Texas showed that time spent in the Arctic prior to initiation of incubation was 18.8±5.6 days (range 11 to 26 days). Assuming pair monogamy, shared incubation between sexes, and an incubation time of about 20 to 22 days, five of the six birds in 2010 and all three birds in 2011 successfully incubated (at least 19 days incubation signal) with no interruption to suggest a failure and re-nest attempt. Birds remained in the Arctic for 5.9±1.5 days (range 3 to 8 days) after completion of incubation before departing for southbound migration. Total days spent in the Arctic for breeders presumed to have successfully incubated was 46.3±4.7 days (range 40 to 52 days), while the presumed failed breeder left after 32 days and did not show further incubation signals to indicate a re-nest attempt (Newstead *et al.* 2013, p. 56).

Niles *et al.* (2008, p. 63) characterized red knots as being typical of shorebird species that exhibit low fecundity (reproductive capacity) (e.g., small clutch size, high nest failure, only one brood per year), delayed maturity, and high annual survival. McGowan *et al.* (2011b, p. 129) estimated rufa red knot fecundity at about 0.25 female fledglings per breeding female, consistent with published estimates for other *Calidris canutus* subspecies. However, these authors note that a better understanding of red knot fecundity parameters is needed for demographic modeling (McGowan *et al.* 2011b, p. 145). Although there is much uncertainty around typical reproductive rates, certainty is high that the red knot's reproductive success varies widely among areas and years and is highly sensitive to predation and weather, as discussed below.

Breeding success of High Arctic shorebirds such as *Calidris canutus* varies dramatically among years in a somewhat cyclical manner (see Factor C—Predation). Two main factors seem to be responsible for this annual variation: abundance of arctic lemmings (*Dicrostonyx torquatus* and *Lemmus sibericus*) and weather. First, lemming abundance is often cyclical, although less so in North America than in Eurasia. Predators feed largely on lemmings, which are easily caught when their abundance is high. However, in years when lemming numbers are low, the predators turn to alternative prey, such as shorebird eggs, chicks, and adults. The variation in shorebird production closely follows variations in lemming abundance. Second, production of shorebird young is also very sensitive to adverse weather during the breeding season. Successful shorebird reproduction occurs almost exclusively during peak lemming years when snowmelt is early (see Factor E—Asynchronies) (Piersma and Lindström 2004, pp. 63–64; Blomqvist *et al*. 2002, p. 149; Summers and Underhill 1987, p. 169). Growth rate of *C. canutus* chicks is very high compared to similarly sized shorebirds nesting in more temperate climates and is strongly correlated with weather-induced and seasonal variation in availability of invertebrate prey (Schekkerman *et al*. 2003, p. 332). *Calidris canutus* chicks grow poorly during cold weather due to higher rates of energy expenditure, shorter foraging periods, and reduced prey availability (Piersma and Lindström 2004, p. 64; Schekkerman *et al*. 2003, p. 340).

Nonbreeding Birds

Unknown numbers of nonbreeding red knots remain south of breeding grounds during the breeding season, and many, but not all, of these knots are 1-year-old (i.e., immature) birds (Niles *et al.* 2008, p. 28). Little information is available about these nonbreeding birds. Nonbreeding knots, usually individuals or small groups, have been reported during June along the U.S. Atlantic and Gulf coasts, with smaller numbers around the Great Lakes and Northern Plains in both the United States and Canada (eBird.org 2014). All juveniles of the Tierra del Fuego wintering region are thought to remain in the Southern Hemisphere during their first year of life, possibly moving to northern South America, but their distribution is largely unknown (Niles *et al.* 2008, pp. 15, 26). *Calidris canutus* (of unknown subspecies composition) have been observed during summer months in Panama, with highest counts exceeding 200 birds (Carmona *et al.* 2013, p. 174). Johnson (2013, p. 5) notes that a small number of nonbreeding red knots are found along Louisiana’s coast throughout the summer, typically in a winter-like plumage; these may be subadults, nonbreeding adults, or both (Johnson 2013, p. 5). There are at least 7 Louisiana records from mid-June to early July, with high counts of 105 and 120 birds during this time period (Purrington 2012, p. 65). Lowery (1974. p. 310) reported over 175 knots on Chandeleur Island on June 12, 1971, and 261 on the Chandeleurs from June 19 to 21, 1973.

One juvenile fitted with a geolocator at Monomoy National Wildlife Refuge (NWR) in Massachusetts in fall spent its first winter on the Outer Banks of North Carolina. The next spring, it did not migrate to the Arctic (as would be expected, being too young to breed), but instead returned to Monomoy NWR for the summer. This bird spent its second winter in Cuba, then migrated to the Arctic and attempted to breed during its second summer (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014). A second juvenile fitted with a geolocator in Monomoy NWR in fall migrated directly to Cuba for its first winter, then started north in spring but did not travel far, spending its first summer in South Carolina and Georgia (S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8, 2014).

One juvenile fitted with a geolocator in Texas remained in Texas during the summer, and for the full year (October 2009 to October 2010) that data were recorded (Newstead *et al.* 2013, p. 57). This result suggests that hatch-year birds arrive on the Texas coast at just over 1 month old, and then spend at least 22 months in the Northwest Gulf of Mexico before undertaking their first northward migration for breeding. Recruitment for this population could therefore be entirely dependent on survival through all seasons in the Northwest Gulf of Mexico (Newstead *et al.* 2013, p. 58). From a radiotelemetry study at Laguna Madre, Texas, the estimated home range for oversummering birds was 225.1 square mi (mi^2) (582.9 square km (km^2). During April and May, knots were concentrated at just three sites and nearby Gulf beaches, but in June nonbreeding Texas birds dispersed southward and by July most were in the Lower Laguna Madre (Newstead 2014a, p. 8).

SUBSPECIES NONBREEDING DISTRIBUTIONS

Geolocator and resightings data show definitively that the *rufa* nonbreeding range includes the entire Atlantic and Caribbean coasts of South America and the Caribbean islands (Niles 2014; S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8 and 12, 2014; Niles *et al.* 2012a, entire; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, entire); Chiloé Island on the central Pacific coast of Chile (J. Parvin pers. comm. March 13, 2014); the Pacific coast of Panama (B. Paxton pers. comm. November 9, 2008; Buehler 2002, p. 42); the North American Gulf and Atlantic coasts from Tamaulipas, Mexico through Quebec, Canada (Niles 2014; S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8 and 12, 2014; Bimbi *et al.* 2014, pp. 29–31; Newstead 2014a, p. 19; Newstead *et al.* 2013, p. 55; Burger *et al.* 2012b, p. 107; Niles *et al.* 2012a, entire; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, entire; Niles *et al.* 2008, p. 19); the interior of South America (Niles 2011a; Niles *et al.* 2010a, p. 126); and the interior of the United States and Canada west at least as far as the Great Plains (D. Newstead pers. comm. May 8, 2014; Newstead *et al.* 2013, pp. 56–57; Niles 2012a). See figures 3 and 4. As discussed below, *Calidris canutus roselaari* also occurs in certain parts of this established *rufa* nonbreeding range. Geolocator and resightings data provide strong evidence that *Calidris cantus* on the Pacific coast from northeastern Russia to Las Garzas, Mexico are the *roselaari* subspecies, and we conclude from the best available data that the rufa red knot does not occur in this region of the Pacific (D. Newstead pers. comm. February 13, 2014; Carmona *et al.* 2013, entire; J. Buchanan pers. comm. January 9, 2013).

In some nonbreeding areas, the *rufa* and *roselaari* subspecies likely or possibly overlap. In other areas with reported nonbreeding occurrences of *Calidris cantus*, no data exist to indicate the subspecies composition (i.e., all *rufa*, all *roselaari*, or both subspecies occur at the same or different times of year). In particular, best available data are insufficient to permit a full understanding of the subspecies composition in parts of the western interior United States (e.g., Idaho, Montana, Wyoming, Nevada, Utah, Colorado, Arizona, and New Mexico), on the coast of Texas, in Central America (i.e., the Pacific and Caribbean coasts from central Mexico through Panama), in northwest South America (i.e., the Caribbean coasts of Colombia and Venezuela), and along the Pacific coast of South America (i.e., Colombia through central Chile). Despite a number of population-wide morphological differences (USFWS 2011a, p. 305), the *rufa* and *roselaari* subspecies cannot be distinguished in the field because physical variability among individuals results in overlaps in many physical parameters (e.g., wing and bill length) between the two subspecies (USFWS 2011a, p. 205; Harrington 2001, pp. 4–5; Harrington *et al.* 1988, p.

441). Because these two subspecies cannot be distinguished in the field, other methods (e.g., mark-resighting efforts, stable isotope analysis, genetics) are needed to delineate their distributions (D. Newstead pers. comm. September 14, 2012). Although geolocator and resightings data have yielded new insights into the nonbreeding distributions of these two subspecies, definitive information on subspecies composition is still lacking for the regions listed above. The best available data for these regions are presented below.

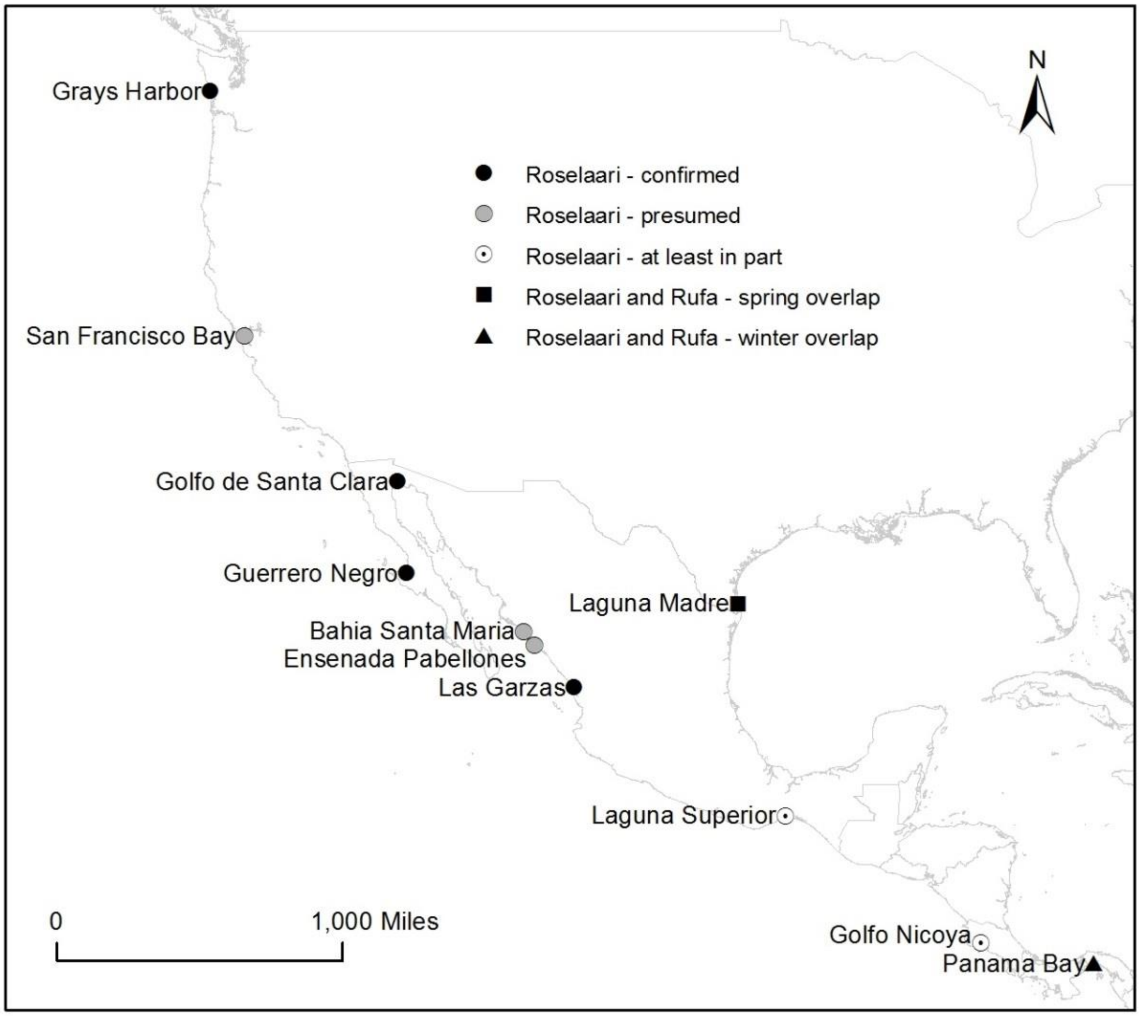
In the past, *Calidris canutus* wintering along the northern coast of Brazil, the Gulf coasts of Texas and Florida, and the southeast Atlantic coast of the United States have sometimes been attributed to the *roselaari* subspecies. However, based on population-wide morphological differences, resightings of marked birds of known subspecies, and results from geolocators, *C.c. roselaari* is now thought to be largely or wholly confined to the Pacific coast during the nonbreeding seasons (Carmona *et al.* 2013, entire; Buchanan *et al.* 2011, p. 97; USFWS 2011a, pp. 305–306; Buchanan *et al.* 2010, p. 41; Soto-Montoya *et al.* 2009, p. 1; Niles *et al.* 2008, pp. 131–133; Tomkovich and Dondua 2008, p. 1). However, the southern extent of the *roselaari* range is still uncertain, as discussed below.

While *Calidris canutus roselaari* is understood to occur mainly along the Pacific coast, a growing number of *C. canutus* movements are being documented between the Pacific coast and Texas during migration (J. Johnson pers. comm. May 15, 2014; D. Newstead pers. comm. May 13, 2014; D. Newstead pers. comm. May 13, 2014), providing evidence that both subspecies likely occur along the Texas coast in spring (D. Newstead pers. comm. February 11, 2014). In addition, evidence of overlapping winter ranges has emerged from Panama (D. Newstead pers. comm. May 13, 2014). Although marked birds of both subspecies have been observed in Texas, to date none of the 1,250 marked *C.c. roselaari* have been observed on the Atlantic coasts of North or South America, and none of the thousands of marked *C.c. rufa* have been observed on the Pacific coast north of Panama.

Known *roselaari* Wintering Range

Wintering areas for *Calidris canutus roselaari* are poorly known (Harrington 2001, p. 5). In some years, small numbers of *C.c. roselaari* winter at Golfo de Santa Clara (State of Sonora) in the northern Gulf of California (Soto-Montoya *et al.* 2009, p. 192). An important wintering aggregation of *C.c. roselaari* has been documented in western Mexico at Guerrero Negro (State of Baja California Sur on the Baja peninsula) (Carmona *et al.* 2013, entire; Carmona *et al.* 2008, p. 12). North of Guerrero Negro, *C. canutus* (presumed but not confirmed *roselaari*) have been reported between December and February in the Mexican State of Baja California and along the coast of California (especially around San Francisco Bay), with small numbers as far north as northern Washington State and southern British Columbia, Canada (eBird.org 2014). Resightings of marked birds in Oregon and California support our understanding that these birds are the *roselaari* subspecies (Carmona *et al.* 2013, p. 174).

Figure 2. Known nonbreeding range of *Calidris canutus roselaari* south of Canada



Until recently, Guerrero Negro was the southern-most confirmed wintering location for *Calidris canutus roselaari*, although several authors had hypothesized that the *roselaari* nonbreeding range probably extends farther south (USFWS 2011a, p. 306; Soto-Montoya *et al.* 2009, p. 1; Carmona *et al.* 2008, p. 13). Two new pieces of information have shown that at least some *C.c. roselaari* do occur south of Guerrero Negro. First, one *C.c. roselaari* marked in Washington in spring 2011 was observed at Las Garzas, Mexico (State of Nayarit), about 630 mi (1,000 km) south of Guerrero Negro (Carmona *et al.* 2013, p. 171). Second, preliminary geolocator data from one *C.c. roselaari* show this bird passed through Texas in spring, bred in Alaska, and wintered along the Central American Pacific coast from Mexico's Baja Peninsula through Panama before returning to Texas the following spring (D. Newstead pers. comm. February 13, 2014). This new geolocator track, the first available for *C.c. roselaari*, lends preliminary support to the hypotheses that an autumn peak at Guerrero Negro includes passage migrants that move on to winter farther south (Carmona *et al.* 2008, p. 13), and that at least some *C.c. roselaari* undertake an elliptical migration path, typical of many shorebirds (Cornell Lab of Ornithology 2007), that takes them through Texas in spring (Newstead 2014b, p. 4)

Based on these resighting and geolocator data, we conclude at least some of the *Calidris canutus* wintering between Guerrero Negro, Mexico and Panama are *C.c. roselaari*. We have no information to indicate whether or not *C.c. rufa* also winters from central Pacific Mexico through Costa Rica. However, based on several resightings of rufa red knots in Panama (see Pacific Panama, below), we conclude that *C.c. roselaari* and *C.c. rufa* both likely occur on the Pacific coast of Panama at least in winter, although we do not have information to determine if the two subspecies utilize the same habitats at the same time within Panama (D. Newstead pers. comm. May 13, 2014), nor can we determine the relative abundance of the two subspecies in Panama.

Northwest Gulf of Mexico

Best available data indicate that *Calidris canutus* in the Northwest Gulf of Mexico wintering area (which extends from Louisiana through Texas, particularly the Laguna Madre, to Mexican State of Tamaulipas) are wholly or predominantly the *rufa* subspecies, at least during winter, summer, and fall. These data include resightings of marked birds that show considerable movement between Texas and known *rufa* areas, including both the Southeast wintering area and the Delaware Bay stopover site (D. Newstead pers. comm. May 13, 2014; BandedBirds.org 2012; D. Newstead pers. comm. August 20, 2012; Niles *et al.* 2008, p. 74). In addition, geolocator data have shown several Texas-wintering birds to use the previously documented midcontinental (Central) flyway in spring (see Migration—Midcontinent), but then follow a fall migration route along the U.S. Atlantic coast (a known *rufa* area) before returning Texas via the Gulf coast (D. Newstead pers. comm. May 8, 2014; D. Newstead pers. comm. February 13, 2014). To date, no known geolocator tracks from Texas birds have shown use of the Atlantic coast during spring migration, but some resighting data suggest that this may also occur (D. Newstead pers. comm. May 8, 2014). Further, and significantly, all geolocator results of Texas-wintering birds to date (at least 45 tracks) show migration pathways to and from *C.c. rufa* (central Canada) rather than *C.c. roselaari* (Alaska and eastern Russia) breeding grounds (Newstead 2014b, p. 1; D. Newstead pers. comm. May 8, 2014; Newstead *et al.* 2013, pp. 56–57; Niles 2012a).

However, it is clear that some movements do occur of birds (presumed or possibly *Calidris canutus roselaari*) from the Pacific coasts to the Northwest Gulf of Mexico, at least in spring. As of July 2014, the following movements between the Pacific and Texas have been documented (see figure 2).

- Two *C.c. roselaari* marked on the Pacific coast (one in Guerrero Negro, Mexico, the other in Grays Harbor, Washington, a known *roselaari* stopover) were resighted in Texas in spring 2012. The bird from Washington was subsequently resighted back at its original banding site 10 days after it was observed in Texas (D. Newstead pers. comm. August 20, 2012).

- One *C. canutus* (presumed *roselaari*) marked in Texas in spring was resighted on the Pacific coast at Grays Harbor, Washington in May 2014 (D. Newstead pers. comm. June 4, 2014; J. Johnson pers. comm. May 15, 2014).

- Preliminary geolocator data from one *C.c. roselaari* show this bird passed through Texas in spring, bred in Alaska, and wintered along the Central American Pacific coast from Mexico's Baja Peninsula through Panama before returning to Texas the following spring

(D. Newstead pers. comm. February 13, 2014), thus following an elliptical migration route. See Known *roselaari* Wintering Range, above; Pacific Coast from Central Mexico to Costa Rica, Pacific Panama, below.

- In February 2012, three marked *C. canutus* were reported from Laguna Superior, part of a Pacific coast lagoon complex that straddles the Mexican States of Oaxaca and Chiapas (Newstead 2014b, p. 1; D. Newstead pers. comm. August 20, 2012). All three of these birds had been banded in Texas in April 2010 (Newstead 2014b, p. 1; D. Newstead pers. comm. August 20, 2012). In February 2013, two additional *C. canutus* that had been marked in Texas during spring (one in 2010, one in 2012) were observed in Laguna Superior (Newstead 2014b, pp. 2, 4). To date, no *C. canutus* marked in other flyways (e.g., Pacific, Atlantic) have been observed in this region, though it has not been frequented by researchers or birders (Newstead 2014b, p. 1). No data are available to indicate if these five *C. canutus* documented passing through Texas in spring and wintering in Oaxaca are *C.c. rufa* or *C.c roselaari* (Newstead 2014b, p. 4; D. Newstead pers. comm. May 13, 2014). However, we conclude that at least some of the birds in southern Mexico are the *roselaari* subspecies based on the one geolocator track available to date for this subspecies (D. Newstead pers. comm. February 13, 2014); see Pacific Coast from Central Mexico to Costa Rica, below.

- One *C. canutus* marked in Texas in spring 2010, and observed again in Texas in spring 2012, was observed in January 2014 in Pacora Este on the Pacific coast of Panama (J. Parvin pers. comm. February 11, 2014). No data are available to indicate if this bird is *C.c. rufa* or *C.c. roselaari* (D. Newstead pers. comm. May 13, 2014). However, we conclude that both subspecies occur on Pacific coast of Panama based on other data; see Pacific Panama, below.

- Two *C. canutus* marked in Texas in April 2012 were observed on Chiloé Island on the central Pacific coast of Chile, one in February 2013 and one in February 2014 (J. Parvin pers. comm. March 13, 2014). No data are available to indicate if these birds are *C.c. rufa* or *C.c. roselaari* (D. Newstead pers. comm. March 12, 2013). However, we conclude at least some of the birds on Chiloé are the *rufa* subspecies based on resighting data (J. Parvin pers. comm. March 13, 2014); see South American Pacific Coast, and figure 3, below.

Through 2012, 1,250 *Calidris canutus roselaari* had been marked at 6 different Pacific coast sites in Russia, Alaska, Washington, and Mexico; and 515 *C. canutus* had been marked on Padre Island, Texas (Carmona *et al.* 2013, p. 170). Despite coordinated resighting efforts, only 12 marked birds (listed above) have been documented moving between the Pacific and Texas. In contrast, researchers in Washington and northwestern Mexico have regularly seen *C.c. roselaari* marked as part of small-scale banding operations in Russia, on the Alaskan breeding grounds near Nome, and at the Yukon River delta (Carmona *et al.* 2013, p. 170; J. Buchanan pers. comm. January 9, 2013). These Pacific coast researchers have seen only one bird that was banded on the Texas coast, despite there being more birds banded in Texas than at these three northern (Alaska-Russia) *C.c. roselaari* areas combined (D. Newstead pers. comm. June 4, 2014; J. Johnson pers. comm. May 15, 2014; J. Buchanan pers. comm. January 9, 2013). These findings are consistent with our current understanding that birds in Texas are primarily *C.c. rufa*, and that C*.c. roselaari* generally occurs along the Pacific coast.

Notwithstanding this understanding of a primarily Pacific distribution for *Calidris canutus roselaari*, the four known or presumed *C.c. roselaari* documented passing through Texas (listed above) show that at least some *C.c. roselaari* undertake an elliptical migration path that takes them through Texas in spring (Newstead 2014b, p. 4). The prevalence of this elliptical route through Texas among *C.c. roselaari*, and the prevalence of *roselaari* among *C. canutus* in Texas in spring, are both still unknown. However, based on best available data, we conclude that both subspecies occur in Texas during spring (D. Newstead pers. comm. February 11, 2014). To date, no fall-captured Texas-marked birds have been resighted anywhere on the Pacific coast, supporting the hypothesis that those *C.c. roselaari* moving through Texas are utilizing the elliptical migration pathway documented by the one available geolocator track for this subspecies (D. Newstead pers. comm. February 11, 2014).

All documented movements between the Pacific and Texas (listed above) involved birds observed in Texas during spring. To date, there are no documented instances of the *roselaari* subspecies wintering in the Northwest Gulf region. Nevertheless, because the *rufa* and *roselaari* subspecies cannot be distinguished in the field due to physical variability among individuals (D. Newstead pers. comm. September 14, 2012), it is possible that some of the birds wintering in the Northwest Gulf of Mexico are *Calidris canutus roselaari* (D. Newstead pers. comm. September 14, 2012). However, based on geolocator and resighting data available to date, we consider all or nearly all of the *C. canutus* wintering the Northwest Gulf of Mexico to be rufa red knots.

Southern Gulf and Caribbean Coasts

There are scattered reports of *Calidris canutus* on the Gulf coast of Mexico, particularly on the Yucatán peninsula, with additional records along the Caribbean coasts of Belize, Honduras, Costa Rica, Panama, Colombia, and Venezuela (eBird.org 2014). Other sources confirm wintering *C. canutus* on the Caribbean coasts of Colombia and western Venezuela (Niles *et al.* 2012a, p. 200; Ruiz-Guerra 2011, entire; Morrison and Ross 1989, Vol. 2, p. 149), and small, isolated groups of wintering birds extend along most of the northern coast of South America (L. Niles pers. comm. January 8, 2013) (see figures 2, 3, and 5). Ruiz-Guerra (2011, p. 194) characterize *C. canutus* as “rather scarce” on Colombia’s Caribbean coast.

Calidris canutus rufa from the Northwest Gulf of Mexico wintering area are documented to extend only as far south as the Mexican State of Tamaulipas, adjacent to Texas (Newstead 2014a, p. 19; Newstead *et al.* 2013, p. 55). Thus, we have no information regarding the subspecies composition of *C. canutus* along the Gulf coast south of Tamaulipas through the Caribbean coast of Panama.

Although information is fragmentary, based on best available data we conclude that at least some of the *Calidris cantuus* on the Caribbean coasts of Colombia and western Venezuela are *C.c rufa.* A rufa red knot fitted with a geolocator in Massachusetts in September 2009 wintered that year near the border between Colombia and Venezuela (Niles *et al*. 2012a, p. 200), and another rufa red knot that wintered in Tierra del Fuego passed through Venezuela in both spring and fall migration (Niles 2011a). We have no data to indicate whether or not *C.c. roselaari* also occurs on Caribbean coasts of Colombia or Venezuela.

Pacific Coast from Central Mexico to Costa Rica

South of Guerrero Negro on Mexico's Pacific coast, *Calidris canutus* has been documented in considerable numbers at several sites in the Gulf of California including Ensenada Pabellones and Bahía Santa Maria (State of Sinaloa); Las Garzas (State of Nayarit); and Laguna Superior (State of Oaxaca) (eBird.org 2014; Newstead 2014b, p. 1; Carmona *et al.* 2013, pp. 172, 174). Smaller numbers of *C. canutus* occur farther south along the Pacific coast of Central America from Guatemala through Costa Rica (eBird.org 2014; Buehler 2002, p. 43).

As discussed above (Northwest Gulf of Mexico), wintering *Calidris canutus* in Laguna Superior on the southern Pacific coast of Mexico (State of Oaxaca) have been shown to include five birds that had been marked in Texas during spring (Newstead 2014b, pp. 1, 2, 4). No data are available to indicate if these five *C. canutus* documented passing through Texas in spring and wintering in Oaxaca are *C.c. rufa* or *C.c roselaari* (Newstead 2014b, p. 4–5; D. Newstead pers. comm. May 13, 2014).

As discussed above (Known *roselaari* Wintering Range), we conclude from preliminary resighting (Carmona *et al.* 2013, p. 171) and geolocator (D. Newstead pers. comm. February 13, 2014) data that at least some of the Pacific coast *Calidris canutus* from Central Mexico to Costa Rica are *C.c. roselaari*. We have no data to indicate whether or not *C.c. rufa* also occurs in this region. However, we have no evidence to suggest that *C. canutus* in the Mexican States of Sinaloa or Nayarit are the *rufa* subspecies, and thus presume Pacific birds in central Mexico are *C.c. roselaari*. Further south in this region, however, we do not rule out overlap of *rufa* and *roselaari*, based on the documented occurrences of *rufa* in Panama (discussed below), and the unknown subspecies composition of the five birds Texas birds that wintered at Laguna Superior. See figure 2.

Pacific Panama

Hundreds of wintering (Buehler 2002, p. 42) and thousands of migrant (Watts 1998, p. 11) *Calidris canutus* have been reported from the Pacific coast of Panama. Limited available data suggest that the *rufa* and *roselaari* subspecies both occur along Panama's Pacific coast.

To date, three resightings of marked birds on the Pacific coast of Panama indicate movement from known *Calidris canutus rufa* areas. Two birds observed in Panama in February and March 2002 had been marked in Argentina in previous years (Niles *et al.* 2008, p. 73; Buehler 2002, p. 42), and a bird seen in November 2008 had been marked in Delaware Bay (B. Watts pers. comm. August 28, 2012; B. Paxton pers. comm. November 9, 2008). The bird that had been marked in Delaware Bay had a stable isotope signature just outside the main cluster of northern-wintering birds from the eastern Caribbean (N. Clark pers. comm. November 10, 2008). Thus, we conclude at least some of the *C. canutus* wintering on the Pacific coast of Panama are rufa red knots.

As described above (Known *roselaari* Wintering Range; Northwest Gulf of Mexico), preliminary geolocator data from one *C.c. roselaari* show this bird passed through Texas in spring, bred in Alaska, and wintered along the Central American Pacific coast from Mexico's Baja Peninsula through Panama before returning to Texas the following spring (D. Newstead pers. comm. February 13, 2014). Thus, we conclude that *C.c. roselaari* and *C.c. rufa* both likely occur in Panama at least in winter, although we do not have information to determine if the two subspecies utilize the same habitats at the same time within Panama (D. Newstead pers. comm. May 13, 2014), nor can we determine the relative abundance of the two subspecies in Panama.

The subspecies of another marked *Calidris canutus* observed in Panama is unknown, as described above (Northwest Gulf of Mexico). This bird was marked in Texas in spring 2010, observed again in Texas in spring 2012, and observed in January 2014 in Pacora Este on the Pacific coast of Panama (J. Parvin pers. comm. February 11, 2014). No data are available to indicate if this bird is *C.c. rufa* or *C.c. roselaari* (D. Newstead pers. comm. May 13, 2014).

South American Pacific Coast

Small numbers of *Calidris canutus* have been reported along the Pacific coast of South America from Colombia through the Los Lagos Region of Chile (eBird.org 2014; Carmona *et al.* 2013, p. 175; Hughes 1979, p. 52). Ruiz-Guerra (2011, p. 194) reported that *C. canutus* winters and is a regular migrant on the Pacific coast of Colombia, reporting tens of birds from intertidal habitats in the Iscuandé River Delta over multiple years. Most of these birds in Colombia were observed foraging on intertidal mudflats or resting on sandy beaches, and most were in nonbreeding plumage and had yellowish legs so were identified as juveniles. Fewer than half the Colombian birds showed breeding plumage in April and May 2011 (Ruiz-Guerra (2011, p. 194). The subspecies composition of *C. canutus* on Colombia’s Pacific coast is unknown. We have no information regarding the subspecies composition of *C. canutus* on the South American Pacific coast in Colombia, Ecuador, Peru, and northern Chile (Carmona *et al.* 2013, p. 175; Ruiz-Guerra 2011, p. 194–195).

Several birds have been documented moving between *Calidris canutus rufa* areas and the central Pacific coast of Chile, primarily Chiloé Island in the Los Lagos Region (J. Parvin pers. comm. March 13, 2014; González *et al.* 2006, p. 110). As of March 2014, the BandedBirds.org database contained six records (listed below) of confirmed or possible rufa red knots wintering on Chiloé Island in the Los Lagos Region of Chile’s Pacific Coast (J. Parvin pers. comm. March 13, 2014), and a seventh record of *rufa* in this region was previously reported by González *et al.* (2006, p. 110).

- At least one rufa red knot marked in Argentina sometime after 1994 was subsequently observed on the central Pacific coast of Chile during southbound migration (González *et al.* 2006, p. 110).

- One rufa red knot banded and observed in Delaware Bay in spring from 2004 to 2009 was observed on Chiloé Island in January 2011 and 2012 (J. Parvin pers. comm. March 13, 2014).

- One rufa red knot banded on Chiloé Island in February 2007 was observed in on Chiloé in January and December 2008, in Virginia in May 2009, in Massachusetts in July and August 2009, on Chiloé in January 2011, and back in Massachusetts in July and August 2011 (J. Parvin pers. comm. March 13, 2014).

- One rufa red knot banded and observed on Chiloé Island in February 2007, December 2008, and January 2011 was seen in Georgia in August 2011, back on Chiloé in January 2012, and in Delaware Bay in May 2012 (J. Parvin pers. comm. March 13, 2014).

- One rufa red knot banded in South Carolina in May 2010 was observed on Chiloé Island in January 2011, and in Massachusetts in August 2011 (J. Parvin pers. comm. March 13, 2014).

- Two *Calidris canutus* banded on Padre Island, Texas in April 2012 were seen on Chiloé Island, one in February 2013 and one in February 2014 (J. Parvin pers. comm. March 13, 2014). Because some *C.c. roselaari* are known to move through Texas in spring (see Northwest Gulf of Mexico, above), we cannot determine if these two birds are *C.c. rufa* or *C.c. roselaari* (D. Newstead pers. comm. March 12, 2013).

Based on these observations of marked birds, we conclude that at least some of the *Calidris canutus* wintering on Chiloé Island, Chile are *C.c. rufa.* Based on best available data, we cannot determine if *C.c. roselaari* also use this area (i.e., if the two subspecies mix on Chiloé Island in winter or fall), nor can we ascertain the subspecies composition of *C. canutus* that occur farther north along the Pacific coasts of Chile, Peru, Ecuador, or Colombia.

Western Interior United States

There is considerable uncertainty regarding the subspecific identity of *C. canutus* in the western interior U.S. (particularly in Montana, Idaho, Wyoming, Colorado, Utah, Nevada, Arizona, and New Mexico), and it is possible that the two subspecies both occur in some of these States during migration. The decision of which of these western States to include in the rufa red knot range was based on best professional judgment of best available data, summarized below.

- Oklahoma, Kansas, Nebraska, South Dakota, and North Dakota were included in the rufa range based on Newstead *et al*. 2013, which reported geolocator results from Texas-wintering knots that were found to use a spring stopover in southern Saskatchewan and (one bird) northern North Dakota. These results were confirmed by subsequent geolocator data that are yet unpublished (D. Newstead pers. comm. May 8, 2014).

- Morrison and Harrington (1992, p. 77) reported, “At least some Knots wintering in Florida travel north through the interior as shown by records of a bird banded in Florida in November being found on northward migration in Manitoba in May, and the sighting of another Florida bird on northward migration in Alberta (B.A. Harrington unpubl. data).” The Manitoba record would not change the presumed rufa range (relative to the preceding bullet), but the Alberta sighting does push the presumed rufa range farther west than any other available data. Communications with B. Harrington (pers. comm. January 10, 2013) on the Florida-to-Alberta bird provided context but not the specific location within Alberta. Colorado, Wyoming, and Montana were included in the rufa range due to their location along a presumed flight path from the Gulf coast toward Alberta, which we examined using Google Earth©. From Alberta, it is reasonable for a bird to continue north to the western part of the rufa breeding range (*e.g.*, Victoria Island).

- New Mexico, Utah, and Idaho were excluded from the rufa range because they lie west of the presumed rufa flight path from the Gulf coast toward Alberta.

- Birds in Arizona and Nevada were presumed *roselaari* based on proximity to known *C.c. roselaari* wintering sites (Baja Peninsula, and northern Gulf of California, both in Mexico), and the location of these two States along a reasonably presumed flight path between the Gulf of California and a well-known *C.c. roselaari* spring stopover at Grays Harbor, Washington.

- Idaho, Nevada, Utah, Arizona, and New Mexico comprise the range of potential flight paths between Padre Island, Texas and Grays Harbor, Washington—a flight that has been documented by at least three birds (confirmed or presumed *C.c. roselaari*) since 2012.

- Birds along the west coast are all known (Washington) or confidently presumed (Oregon, California) *C.c. roselaari* based on Carmona *et al*. (2013).

While it is possible that rufa red knots range nearly all the way to the Pacific coast during migration, we do not have any evidence of *rufa* past the Rocky Mountains to date. In comparison, we do have three documented cases of confirmed or presumed *roselaari* moving from Texas to Washington. Thus, we conclude at least some of the inland birds west of the Rocky Mountains are *roselaari*. It is possible the two subspecies could overlap in these western interior areas, but we find that *rufa* would probably not veer past the Rocky Mountains given their presumed flight trajectory to and from their breeding grounds in the Canadian Arctic archipelago. We acknowledge considerable uncertainty around the subspecies composition in the western States but conclude, based on best available data, that the *rufa* range likely extends to the western limit of the Great Plains (as mapped by the Level I ecoregions (U.S. Environmental Protection Agency (USEPA) 2013a)).

WINTERING

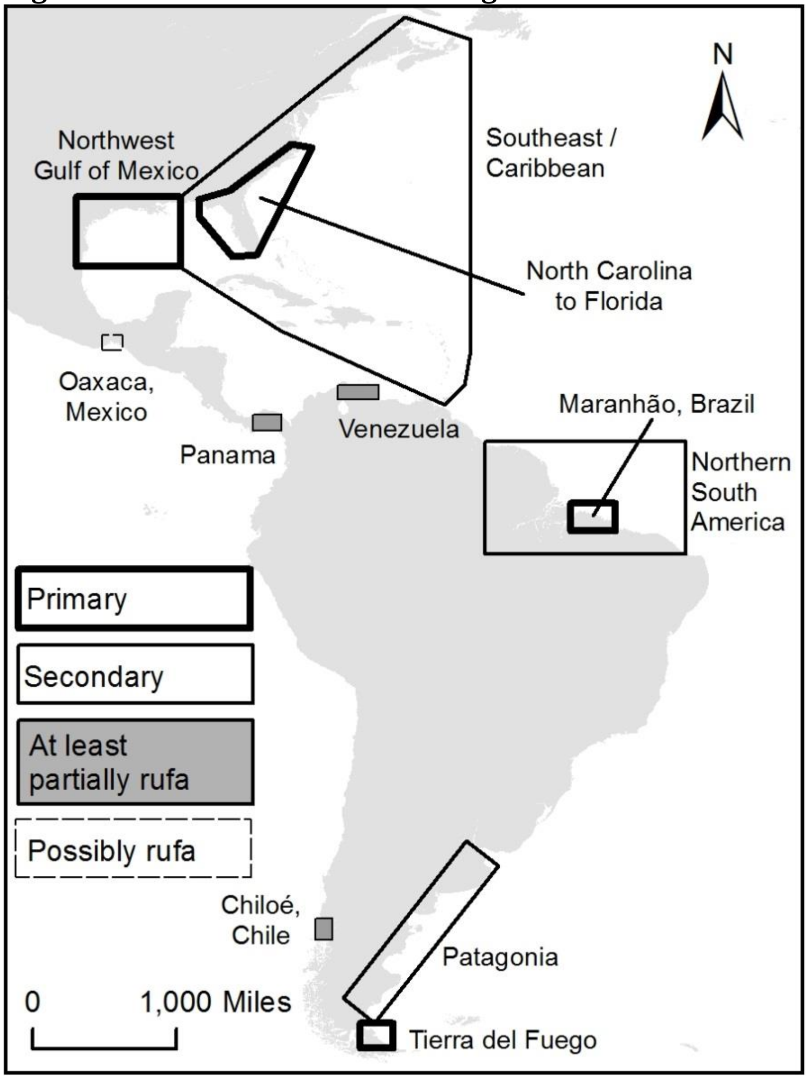
In this document, "winter" is used to refer to the nonbreeding period of the red knot life cycle when the birds are not undertaking migratory movements. Red knots occupy all known wintering areas from December to February, but may be present in some wintering areas as early as September or as late as May. In the Southern Hemisphere, these months correspond to the austral summer (i.e., summer in the Southern Hemisphere), but for consistency in this document the terms "winter" and "wintering area" are used throughout the subspecies' range.

Wintering Distribution and Range

Wintering areas for the rufa red knot (figure 3) include the Atlantic coasts of Argentina and Chile (particularly the island of Tierra del Fuego that spans both countries), the north coast of Brazil (particularly in the State of Maranhão), the Northwest Gulf of Mexico from the Mexican State of Tamaulipas through Texas (particularly at Laguna Madre) to Louisiana, and the Southeast United States (discussed further below) from Florida (particularly the central Gulf coast) to North Carolina (Newstead 2014a, p. 19; Newstead *et al.* 2013, p. 55; L. Patrick pers. comm. August 31, 2012; Niles *et al.* 2008, p 17). Smaller numbers of knots winter in the Caribbean, and along the central Gulf coast (Alabama, Mississippi), the mid-Atlantic, and the

Northeast United States. As discussed above (Subspecies Nonbreeding Distribution), *Calidris canutus* is also known to winter in Central America, northwest South America, and along the Pacific coast of South America, but it is not yet clear if all these birds are the *rufa* subspecies.

Figure 3. Known red knot wintering areas



In some years, more red knots have been counted during a coordinated spring migration survey than can be accounted for at known wintering sites (see Population Surveys and Estimates, below), suggesting there are unknown wintering areas. Indeed, geolocators have started revealing previously little-known wintering areas, particularly in the Caribbean (Niles *et al*. 2012a, pp. 197–200; L. Niles pers. comm. January 8, 2013).

Geolocator results from the Southeast (S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8, 2014; Niles *et al.* 2012a, pp. 198, 200, 202) and from the Northwest Gulf of Mexico (Newstead *et al.* 2013, p. 55) have shown that birds wintering in these regions may move considerable distances during the core winter months (also see Juveniles, below). Similarly, a radiotracking study at Laguna Madre, Texas found that some birds undertook regional movements during winter. In the winters of 2012 and 2013, the mean home range was estimated at 163.5 and 149.3 mi^2 (423.4 and 386.7 km^2), respectively. About 53 percent of birds were relocated in a fairly small geographic area, especially in the Upper Laguna Madre area and North Padre Island beach. For many of these birds, a relatively small (usually about 19mi (30 km)) stretch of Gulf beach and nearby flats in the Laguna Madre comprised all of the relocations. However, about 16 percent of birds were relocated over a wider geographic area (between 19 and 62 mi (30 and 100 km)), often comprising a longer stretch of beach and several bay systems or several areas of the Laguna Madre system. Further, about 32 percent of birds made use of Gulf beaches and the Upper Laguna Madre as well as Lower Laguna Madre area (movements of over 62 mi (100 km)). These longest-distance movements were not always unidirectional over time, with some birds relocating back and forth from Upper to Lower Laguna Madre and beaches multiple times. One bird was relocated three times on the upper Texas coast near San Luis Pass at the west end of Galveston Island. The majority of these large ranges of movement were oriented southward of the original catch area (Newstead 2014a, pp. 3, 6–8).

Southeast United States and the Caribbean

The core of the Southeast wintering area (i.e., that portion of this large region supporting the majority of birds) is thought to shift from year to year among Florida (particularly the central Gulf coast), Georgia, and South Carolina (Niles *et al.* 2008, p. 17). However, the geographic limits of this wintering region are poorly defined. Although only small numbers are known, wintering knots extend along the Atlantic coast as far north as Virginia (L. Patrick pers. comm. August 31, 2012; Niles *et al.* 2006, p. 89), Maryland (Burger *et al.* 2012b, p. 6), and New Jersey (BandedBirds.org 2012; H. Hanlon pers. comm. November 22, 2012; A. Dey pers. comm. November 19, 2012). Still smaller numbers of red knots have been reported between December and February from Long Island, New York, through Massachusetts and as far north as Nova Scotia, Canada (eBird.org 2014).

Further blurring the geographic limits of the Southeast wintering region are small numbers of red knots that winter along the central Gulf coast (Florida Panhandle, Alabama, Mississippi, and eastern Louisiana) (L. Patrick pers. comm. August 31, 2012; Morrison and Harrington 1992, p. 77). It is unclear if red knots that winter in the central Gulf should be considered part of the Southeast wintering group or the Northwest Gulf of Mexico group. Alternatively, the two areas may constitute a single, large wintering region. Further investigation of intra- and inter-annual red knot movements within and between these two wintering regions would be needed to clarify their geographic limits and degree of connectivity.

Although the numbers are poorly known, red knots also winter in the Caribbean islands (Morrison and Harrington 1992, p. 77), which are typically included in the Southeast wintering region. New geolocator results suggest the Caribbean may play a more important role in winter than previously known, with three of eight tracked “Southeast” birds wintering in Cuba or the Bahamas (Niles *et al*. 2012a, pp. 197–200). Other geolocator results have also documented red knots wintering in Haiti and Jamaica (L. Niles pers. comm. January 8, 2013; Burger *et al.* 2012c, p. 374). Previously, red knots had been known from the Caribbean islands from every month of the year, but were generally considered rare (defined by this author as occurring less than twice per year) (Raffaele *et al.* 1998, p. 277). This characterization of only incidental red knot occurrence in the Caribbean is being reconsidered in light of the geolocator results (Niles *et al.* 2012a, p. 200). Stable isotope signatures from the Southeast overlap with those from Brazil, suggesting even northern South America may be affiliated with this large wintering region (Atkinson *et al.* 2006a, p. 536).

Northern Versus Southern Wintering Regions

Researchers often distinguish between those rufa red knots that winter the farthest south (in Argentina and Chile) and therefore undertake the longest-distance migrations (referred to as "southern-wintering" in this document), from those that winter farther north in northern Brazil and the Southeast (referred to as "northern-wintering" in this document). As shown in figure 3, the southern-most wintering region (Patagonia and Tierra del Fuego) is separated from other wintering areas by a considerable distance, at least 2,300 mi (3,700 km). Southern-wintering red knots are concentrated in Tierra del Fuego, located roughly 3,700 mi (5,955 km) south of the northern Brazil wintering area. Winter area fidelity appears to be high, with minimal movement of birds between southern and northern wintering regions (see Wintering Area Fidelity). Differences observed between northern- versus southern-wintering red knots are discussed throughout this document, and are summarized below.

- Some data show statistically significant morphological differences between northern- versus southern-wintering red knots. On average and controlling for factors like sex and molt status, birds from Tierra del Fuego are statistically significantly smaller (bill length, mass) than those from Brazil or Florida (Niles *et al.* 2008, pp. 9–11). However, in an earlier study, Harrington *et al.* (1988, p. 441) found no statistically significant differences in wing or bill lengths of red knots caught in Florida versus those caught in Argentina (see Breeding Distribution).

- Resighting, geolocator, stable isotope, and radio tracking data all suggest that southern-wintering red knots may be more reliant on the Delaware Bay spring stopover area than are birds that winter in the Southeast (Bimbi *et al.* 2014, p. 31; Niles *et al*. 2012a, pp. 197–200; P. Atkinson pers. comm. November 8, 2012; Niles 2011a; Niles *et al.* 2010a, entire; Cohen *et al.* 2010a, p. 660; Atkinson *et al*. 2006a, p. 536; Atkinson *et al.* 2005, p. 738; Harrington 2005a, p. 1; Harrington *et al.* 1988, pp. 440–441) (see Migration and Migration and Wintering Food—Possible Differential Reliance on Horseshoe Crab Eggs).

- Although at least some southern-wintering knots do eat prey other than horseshoe crab (*Limulus polyphemus*) eggs at mid-Atlantic spring stopovers, there is considerable evidence that southern-wintering knots may be more reliant on horseshoe crab eggs than are northern-wintering knots (see Migration and Wintering Food—Possible Differential Reliance on Horseshoe Crab Eggs; table 2).

- The late arrivals in Delaware Bay that were a key synergistic factor (acting in conjunction with reduced availability of horseshoe crab eggs) accounting for declines in survival rates in the 2000s primarily affected southern-wintering red knots (see Factor E—Asynchronies—Delaware Bay—Late Arrivals).

- At the Delaware Bay spring stopover area, differences have been found in arrival and departure times, residence times, and return rates between northern- and southern-wintering red knots (Atkinson *et al.* in International Wader Study Group (IWSG) 2005, p. 15; Baker *et al.* 2004, p. 878; Harrington *et al.* 1988, p. 442) (see Migration—Differences in Migration Strategy by Wintering Region).

- Although there is some data to the contrary (Atkinson *et al.* 2005, pp. 738–746), southern-wintering knots are generally believed to delay molting their flight feathers until they reach the wintering grounds, while northern-wintering birds are generally thought to molt their flight feathers during fall or early winter, north of their final wintering destination (A. Dey pers. comm. November 19, 2013; Harrington *et al*. 2010b, p. 362; Niles *et al*. 2008, p. 15; Harrington *et al.* 2007, p. 39–41; Morrison and Harrington 1992, pp. 79–80) (see Migration—Atlantic Coast—Fall Timing and Distribution).

- Studying fall migration stopover areas in Massachusetts, Harrington *et al.* (2010b, p. 357) found that red knots from northern versus southern wintering areas had different migration chronologies, plumage characteristics, flight feather molt, foods, foraging habitats, stopover durations, and uses of foraging and roosting habitats. Knots from the two groups have different strategic uses of the Cape Cod stopover location, with the southern-wintering knots using it as a migration staging point and the remaining knots using it as a molting area. (See Migration—Differences in Migration Strategy by Wintering Region.)

- The southern-wintering population has declined sharply and contracted geographically relative to the early 1980s. No declines are apparent in red knot populations in Brazil or the Southeast, although survey data from these northern wintering areas are associated with lower confidence. (See Population Surveys and Estimates.)

Notwithstanding these differences, considerable data show that northern- and southern-wintering red knots do mix during both spring and fall migration (see Migration), and fragmentary evidence suggests they may also mix on the breeding grounds (see Breeding Distribution).

Red knots wintering in the Caribbean, the Northwest Gulf of Mexico, Venezuela, Colombia, Central America, and on the Pacific coast of South America were discussed in the preceding section. We have no information regarding how birds wintering in these areas may fit into the categorization of northern- versus southern-wintering red knots that has been reported by the authors cited above.

Wintering Period

Red knots occupy the southernmost wintering areas, in Tierra del Fuego, from late October to February, with some birds arriving as early as late September (Niles *et al.* 2008, p. 15; Morrison and Harrington 1992, p. 76). Birds wintering in the Caribbean or the United States typically stay later, through March or even May (S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8 and 12, 2014; Newstead *et al.* 2013, p. 56; Niles *et al*. 2012a, pp. 197–200; Niles 2012a; Niles 2009, p. 10). Birds wintering in the Southeast and the Caribbean seem to arrive in November (Niles *et al.* 2012a, pp. 197–200). However, two young birds arrived in Cuba somewhat earlier, from mid-August to mid-September, for their second and third winters, with one second-year bird then moving back north until November (see Juveniles, below) (S.

Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8 and 12, 2014). Compared to most Southeast-wintering birds, knots wintering in Texas seem to arrive much earlier, in late July or August (Newstead *et al.* 2013, p. 57; Niles 2012a). Geolocator results from seven adults showed that knots wintering in the Northwest Gulf of Mexico spent nearly the entire nonbreeding phase of their annual cycle (286 days, or 78.4 percent of the calendar year) on the Texas coast (Newstead *et al.* 2013, p. 55).

Wintering Area Fidelity

Red knots are generally thought to return to the same wintering region each year. Between January and April 1981, 263 red knots were marked in Florida and 181 were marked in Argentina; no Argentina birds were found in Florida or vice versa (Harrington *et al.* 1988, p. 441). Resightings of marked birds show few or no inter-annual movements of red knots between the Brazil and Tierra del Fuego wintering areas, or between the Southeast and Tierra del Fuego wintering areas (Baker *et al.* 2005, pp. 13–14; Harrington 2005a, p. 1). Extensive searches of Brazil in the winters of 2004 and 2005 failed to find any red knots marked from Tierra del Fuego or from the Southeast, but observation rates are too low to draw a firm conclusion because few knots have been marked in Brazil (Niles *et al.* 2008, pp. 9, 55; Baker *et al.* 2005, pp. 13–14). Using modeling and resighting rates, Schwarzer *et al.* (2012, p. 729) found that the estimated probability of surviving birds that had been banded as juveniles in Florida returning to Florida as adults during the following winter was 0.70 ± 0.08, and the estimated probability of winter-to-winter fidelity for surviving adults banded in Florida was 0.81 ± 0.05. It is unknown if the surviving birds that failed to return subsequently wintered elsewhere in the Southeast or in another region, but these results suggest that winter area fidelity might be higher in adults than in juveniles.

Examples of red knots changing wintering region do exist, but are few. Records of band resightings include about six birds that were observed in Florida during the core winter months (December to February) of one year, and in Chile between December and February of another year (BandedBirds.org 2012). Band resight and isotope data from Georgia indicated that a small percentage of red knots switch wintering locations between years (Georgia Department of Natural Resources (GDNR) 2013), with 7 birds out of 814 (0.9 percent) using a northern wintering area in one year and Tierra del Fuego in another year (T. Keyes pers. comm. July 30, 2014). One marked knot seen on February 20, 2002, in Panama had been banded in Tierra del Fuego in February 1995, and another seen in Panama on March 15 and 28, 1995, had been banded in San Antonio Oeste, Argentina, in March 1998 (Niles *et al.* 2008, p. 73; Buehler 2002, p. 42). These observations from Panama suggest other possible instances of red knots switching wintering regions.

Juveniles

There is little information on where juvenile red knots spend the winter months (USFWS and Conserve Wildlife Foundation of New Jersey (CWFNJ) 2012, p. 1), and there may be at least partial segregation of juvenile and adult red knots on the wintering grounds. Juvenile birds are typically under-represented in catches of red knots made in major wintering areas such as Chile and Florida (A. Dey pers. comm. April 7, 2008). In winter 2008–2009, Niles (2009, pp. 9–10) found the percentage of juvenile red knots peaked in January (13 to 15 percent) compared to catches made in November (3 to 6 percent) and February (0 percent). The overall low frequency suggests that juveniles may winter separately from adults or may occupy unknown habitats not used by adults (Niles 2009, pp. 9–10). Because we lack specific information on juvenile red knots, we use the best available data from adult red knots to draw conclusions about juvenile foraging and habitat use.

Many of the *Calidris canutus* seen in Panama in January and February 2002 were identified as juveniles based on leg color (Niles *et al*. 2008, p. 73). In French Guiana, juveniles birds are seen together with adults in the same roosting flocks (Conseil Scientifique Régional du Patrimoine Naturel (Regional Scientific Council for Natural Heritage, or CSRPN 2013)). One juvenile fitted with a geolocator at Monomoy NWR in Massachusetts in fall spent its first winter on the Outer Banks of North Carolina, while another spent its first winter in Cuba. This second juvenile showed a previously unknown behavior during its second winter, arriving in Cuba in early September, then moving slowly back north through the Bahamas to the Carolinas before returning to Cuba in mid-November. It is possible these were exploratory movements by this young bird (S. Koch, L. Niles, R. Porter, and F. Sanders pers. comm. August 8 and 12, 2014); see Migration—Stopover Usage regarding an exploratory flight documented during spring migration.

MIGRATION

Migration Biology

Each year some red knots make one of the longest distance migrations known in the animal kingdom, traveling up to 19,000 mi (30,000 km) annually. Red knots undertake long flights that may span thousands of miles without stopping. As *Calidris canutus* prepare to depart on long migratory flights, they undergo several physiological changes. Before takeoff, the birds accumulate and store large amounts of fat to fuel migration and undergo substantial changes in metabolic rates. In addition, the leg muscles, gizzard (a muscular organ used for grinding food), stomach, intestines, and liver all decrease in size, while the pectoral (chest) muscles and heart increase in size. Due to these physiological changes, *C. canutus* arriving from lengthy migrations are not able to feed maximally until their digestive systems regenerate, a process that may take several days. Because stopovers are time-constrained, *C. canutus* requires stopovers rich in easily digested food to achieve adequate weight gain (Niles *et al.* 2008, pp. 28–29; van Gils *et al.* 2005a, p. 2609; van Gils *et al.* 2005b, pp. 126–127; Piersma *et al*. 1999, pp. 405; 412) that fuels the next migratory flight and, upon arrival in the Arctic, also fuels a body transformation to breeding condition (Morrison 2006, pp. 610–612) (see Migration and Wintering Habitat). However, some researchers have suggested that these digestive changes are more pronounced, or have a more pronounced effect on stopover time and energy budgets, in southern-wintering (Argentina and Chile) than in northern-wintering (Southeast United States) rufa red knots (Niles *et al.* 2008, p. 36; Atkinson *et al.* 2006b, p. 41); see Migration and Wintering Food—Horseshoe Crab Eggs—Possible Differential Reliance on Horseshoe Crab Eggs. At stopover sites, body mass varies greatly from very light birds that have just arrived to very heavy birds that are just about to depart. In Delaware Bay, for example, some red knots arrive as light as 3.1 oz (89 g) (32 percent below fat-free weight), whereas near departure some birds may exceed 8.5 oz (240 g) (85 percent above fat-free weight) (Niles *et al.* 2008, p. 28).

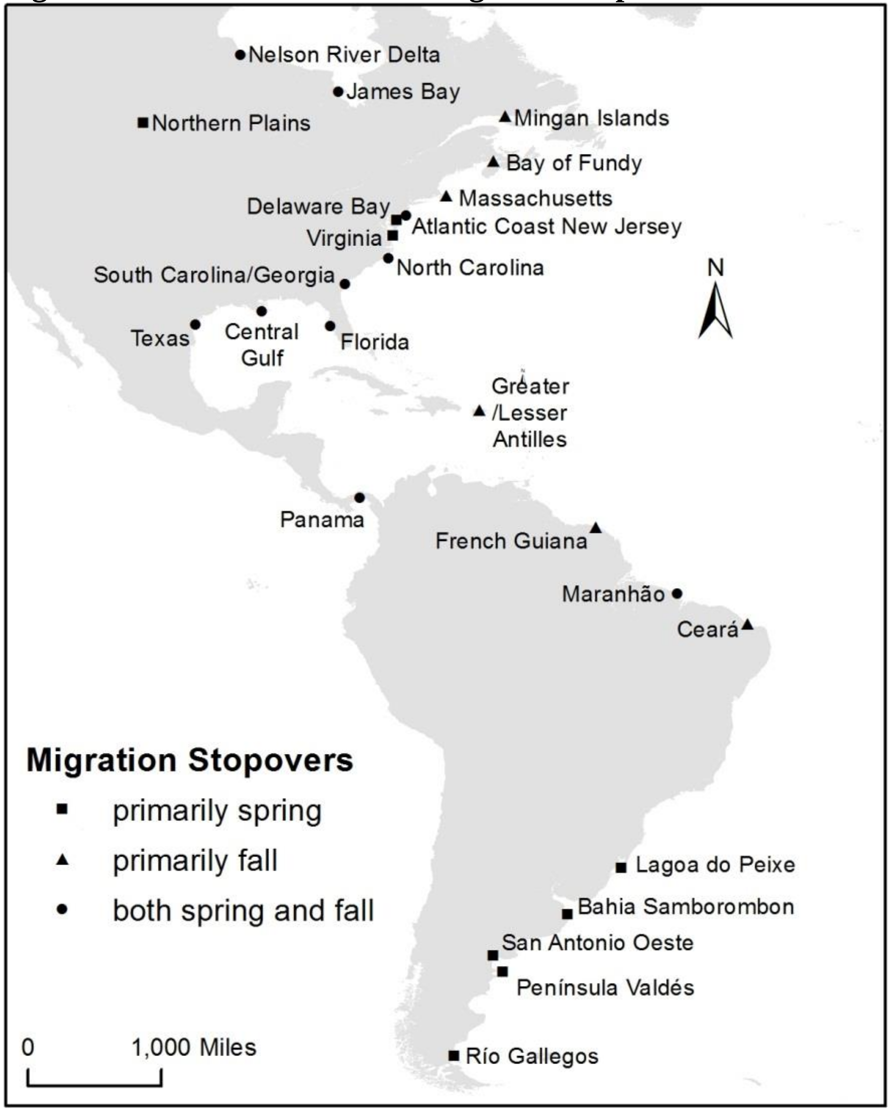
Red knots tend to migrate in single-species flocks with departures typically occurring in the few hours before twilight on sunny days. Size of the departing flocks tends to be large (greater than 50 birds) (Niles *et al.* 2008, p. 28). Likewise, based on observations of other *Calidris canutus* subspecies departing from Iceland towards Nearctic breeding grounds in spring 1986 to 1988, Alerstam *et al.* (1990, p. 201) found mean flock sizes of 100 to 200 individuals. These *C. canutus* leaving Iceland in spring departed in flight formations during the afternoon or evening, and during rising or high tide; their departures had significant differences in daily timing between seasons that was associated with between-year differences in the tidal cycle. Within the season, departures took place earlier in relation to high tide as the season progressed (Alerstam *et al.* 1990, p. 201). Consistent with the afternoon and evening departures of *C. canutus* from Iceland, rufa red knots are inferred to migrate during both night and day based on the duration and distance of migratory flight segments estimated from geolocator results (Normandeau Associates, Inc. 2011, p. 203).

Stopover Areas

Places where migrant birds stop to rest, drink, and eat at are often described as either stopover or staging sites (Warnock 2010, p. 621). Attempts have been made to differentiate between these two terms but they are frequently used interchangeably. Some authors have equated staging sites with sites that attract large concentrations (many thousands) of birds, a definition that others have expanded to include long stopover durations and significant rates of refueling on predictable, abundant prey. It has also been suggested that birds using staging sites are those that employ a "jump" migration strategy (Warnock 2010, p. 621) (i.e., longer flights, versus a "hop" strategy of shorter fights and more frequent stops). Several authors have defined stopover sites as any areas where birds rest and feed during migration (Warnock 2010, p. 621; Pompei and Cuthbert 2004, p. 3). However, Warnock (2010, p. 621) argued that further classification of stopover sites is of ecological and conservation value and proposed that staging areas should be defined as those stopover sites with abundant, predictable food resources where birds prepare for an energetic challenge (usually a long flight over a barrier such as an ocean or a desert) requiring substantial fuel stores and physiological changes without which significant fitness costs are incurred (Warnock 2010, p. 621). Using this more narrow definition, there are probably relatively few true staging sites in the world (e.g., Delaware Bay), and there can be profound individual and population-level consequences for birds when these sites are lost or degraded (Warnock 2010, p. 624).

Shorebirds migrate along traditional routes characterized by a chain of key staging areas that are essential to successful migration (Myers 1983, p. 23). Key stopover and staging areas serve as stepping stones between wintering and breeding areas (IWSG 2003, p. 10). Shorebirds reach staging areas with depleted fat reserves after many hours of nonstop flight (see Migration Biology). Without access to the energy available from prey at staging sites, the birds would be unable to continue their migrations. The birds' ability to shift to alternative staging areas is usually limited, as those few sites uniquely able to support large numbers of shorebirds stopping to "refuel" may be widely separated by hundreds of miles (Myers 1983, p. 23). Long-distance migrant shorebirds in particular are highly dependent on the continued existence of quality habitat at a few key staging areas. Conditions or factors influencing shorebird populations on staging areas control much of the remainder of the birds' annual cycle and survival (Skagen 2006, p. 316; IWSG 2003, p. 10). In essence, these staging areas are geographic bottlenecks, and the populations within entire migration corridors can be affected by their environmental health (Myers 1983).

Figure 4. Well-known red knot migration stopover areas



At some stages of migration, very high proportions of entire shorebird populations may use a single migration staging site to prepare for long flights. High fractions of the red knot's rangewide population can occur together at a small number of nonbreeding locations, leaving populations vulnerable to loss of key resources (Harrington 2001, p 22). For example, Delaware Bay provides the final Atlantic coast stopover for a significant majority of the red knot population making its way to the arctic breeding grounds each spring (Clark *et al*. 2009, p. 90; Niles *et al.* 2008, p. 30; Harrington 2001, p. 22). Based on earlier rangewide population estimates, Brown *et al*. (2001, p. 10) estimated that Delaware Bay supports 50 to 80 percent of all red knots during spring. Although no current, reliable, rangewide population estimate is available, reliable population data are available on a regional basis (see Population Surveys and Estimates). We have analyzed the most recent estimates of red knot numbers from each wintering region, Delaware Bay peak counts from the past 10 years, and Delaware Bay total passage population estimates from the past 3 years. Based on this analysis, we conclude that Delaware Bay continues to support the majority of red knots during spring.

Red knots show moderate fidelity to particular migration staging areas between years (CSRPN 2013; Duerr *et al.* 2011, p. 16; Harrington 2001, pp. 8–9, 21). Figure 4 shows well-known red knot stopover areas for reference; however, large and small groups of red knots, sometimes numbering in the thousands, occur in suitable habitats all along the Atlantic, Caribbean, and Gulf coasts. In figure 4 and throughout this document, "spring" is used to refer to the nonbreeding period of the red knot life cycle when the birds are undertaking northbound migratory movements, which may occur from February through early June in different parts of the range. In the Southern Hemisphere, these months correspond to the austral fall (i.e., fall in the Southern Hemisphere), but for consistency in this document the terms "spring" and "spring stopover area" are used throughout the subspecies' range. Likewise, "fall" is used to refer to the nonbreeding period of the red knot life cycle when the birds are undertaking southbound migratory movements, which may occur from July through December in different parts of the range. In the Southern Hemisphere, these months correspond to the austral spring (i.e., spring in the Southern Hemisphere), but for consistency in this document the terms "fall" and "fall stopover area" are used throughout the subspecies' range.

Atlantic Coast

Spring Timing and Distribution

Well-known spring stopover areas along the Atlantic coast include Río Gallegos, Península Valdés, and San Antonio Oeste (Patagonia, Argentina); Lagoa do Peixe (eastern Brazil, State of Rio Grande do Sul); Maranhão (northern Brazil); the Southeast United States (e.g., Georgia and the Carolinas); the Virginia barrier islands (United States); and Delaware Bay (Delaware and New Jersey, United States) (see figure 4) (A. Dey pers. comm. April 21, 2014; Wallover *et al.* 2014, p. 6; South Carolina Department of Natural Resources (SCDNR) 2013, p. 36; Cohen *et al.* 2009, p. 939; Niles *et al.* 2008, p. 19; González 2005, p. 14). However, large and small groups of red knots, sometimes numbering in the thousands, may occur in suitable habitats all along the Atlantic and Gulf coasts from Argentina to Massachusetts (Niles *et al.* 2008, p. 29).

<u>South America</u>

Based on resightings, one marked red knot was documented to fly about 5,000 mi (8,000 km) from San Antonio Bay, Argentina to Florida in 9 days (Western Hemisphere Shorebird Reserve Network (WHSRN) 2014a). From geolocators, examples of spring migratory tracks are available for four red knots that wintered in South America. One flew about 4,000 mi (6,400 km) over water from northeast Brazil in 6 days, and another flew about 5,000 mi (8,000 km) from the southern Atlantic coast of Brazil (near Uruguay) over land (the Amazon) and water (the eastern Caribbean) in 6 days. Both of these birds touched down in North Carolina, and then used Delaware Bay as the final stopover before departing for the arctic breeding grounds (Niles *et al.* 2010a, p. 126). A third red knot, which had wintered on the north coast of Brazil, flew over water to Delaware Bay, passing near the Lesser Antilles but not stopping along the southeastern U.S. coast (Niles 2014).

Geolocator data show that a fourth red knot, which had wintered in Tierra del Fuego, followed an overland route through the interior of South America, departing near the Venezuela-Colombia border. This bird then flew over the Caribbean to Florida, and finally to Delaware Bay (Niles 2011a). This geolocator result of an inland route is consistent with observations of 25 red knots in interior Argentina (Laguna Mar Chiquita) in March 1988, and 67 knots in May 1991 (eBird.org 2014). The existence of an overland South American flyway is also supported by observations of northbound red knots stopping over in French Guiana in April and early May, with one report of 350 birds on in mid-May 2002 at Battures de Malmanoury (CSRPN 2013). A strictly coast route from Argentina to French Guiana seems unlikely. A few knots may pass through French Guiana as late as early June (CSRPN 2013).

Notwithstanding the one geolocator result and limited sightings data showing an overland route, most knots that winter in Tierra del Fuego are thought to work their way up the southern Atlantic coast, using stopover sites in Argentina and Uruguay before departing from Brazil (Niles *et al*. 2008, p. 15). For example, based on banding studies from 1984 to 1991, Antas and Nascimento (1996, p. 63) found that red knots stopping at Lagoa do Peixe in the spring had originated from southern Argentina and Chile; these birds migrated north to Punta Rasa (at Bahía Samborombón in northern Argentina) and the coast of Rio Grande do Sul (which includes Lagoa do Peixe in southeastern Brazil), then to Maranhão (northern Brazil), and then to Delaware Bay.

Although a few birds may depart before the end of January, the main red knot movement north from Tierra del Fuego occurs in February. The northward migration through South America is typically rapid, with only brief stopovers (Niles *et al*. 2008, p. 15). However, northbound migrants stopping at Península Valdés, Argentina, stay approximately 17 to 22 days, occurring from March to May but peaking in April (Musmeci *et al.* 2012, pp. 359–360). Birds moving north from Argentina typically arrive in Brazil in April (Niles *et al.* 2008, p. 29). Numbers of northbound red knots on the Atlantic coast of Brazil, just north of Lagoa do Peixe, peaked in April of 2008 and 2009 (Scherer and Petry 2012, p. 46). Departure from Brazil tends to occur in the first half of May (Niles *et al.* 2010a, p. 126; Niles *et al*. 2008, pp. 15, 29). Many knots marked in Argentina and Chile are seen on the Atlantic coasts of Florida, Georgia, South Carolina, and North Carolina during, but not before, May (B. Harrington pers. comm. November 14, 2013; SCDNR 2013, p. 31).

Southeast and Caribbean

Available data indicate that red knots wintering in the Southeast (including the Caribbean) use at least two different spring migration routes—coastal and inland. Geolocator results for eight knots from this northern wintering region showed most birds moving up the U.S. coast, using stopovers between the Carolinas and Massachusetts before departing for the Arctic (Bimbi *et al.* 2014, pp. 29–30; Niles *et al.* 2012a, pp. 197–200). These findings are supported by stable isotope analysis and resightings of marked birds, showing that at least a portion of the Southeast-Caribbean wintering group uses mid-Atlantic coastal stopovers including Virginia and Delaware Bay (BandedBirds.org 2012; SCDNR 2013, p. 38; Smith *et al.* 2008, p. 16; Atkinson *et al*. 2006a, p. 536; Atkinson *et al.* 2005, p. 738). Likewise, geolocator data from one juvenile bird show it used a coastal migration route during its second spring, en route to its first summer in the Arctic. Moving north from its wintering area in Cuba, this young bird stopped in South

Carolina and Delaware Bay before heading to Canada (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014).

In contrast to this coastal route, some red knots wintering in the Southeast-Caribbean region use an inland route in spring. The geolocators on two Southeast-wintering birds showed an inland route, flying to central Canada directly from the wintering area in South Carolina, passing over the Great Lakes. One of these birds headed northwest over Kentucky to stopover at the Nelson River delta, while the other headed north over western Pennsylvania to stopover at James Bay (Bimbi *et al.* 2014, p. 31; Niles *et al.* 2012a, p. 197). Use of an interior route through the United States and Canada has also been documented by resightings in Manitoba and Alberta, Canada of birds that had been marked in Florida (Morrison and Harrington 1992, p. 77). Existence of an inland route is further supported by a lack of resightings in Delaware Bay (despite extensive searches) of birds marked in South Carolina or Georgia (Harrington 2005a, p. 1). For example, between 2011 and 2012, 285 red knots out of 513 birds either banded or resighted in South Carolina were not seen in Delaware Bay (SCDNR 2013, p. 38). Of course, the detection rate of marked birds within Delaware Bay is not 100 percent, and some northbound coastal migrants use other mid-Atlantic stopovers (e.g., Virginia). Nonetheless, the fact that about 55 percent of South Carolina birds were not seen in Delaware Bay is further support for existence of an inland route, suggesting at least some of these birds headed straight for Canada from the Southeast.

Geolocator results to date show birds departing the Caribbean in March, while departures from the Southeast were from late April through May (Niles *et al.* 2012a, pp. 197–200). From these same geolocator data (for six birds using Southeast-Caribbean wintering areas), the total time spent along the U.S. Atlantic coast (including spring, fall, and for some birds winter) averaged 218 days (range 121 to 269 days) (Burger *et al.* 2012b, p. 1), or about 60 percent of the calendar year.

Stopover Usage

Some red knots from the Southeast-Caribbean wintering region, and most birds from South American wintering areas, utilize spring stopovers along the coasts of the Southeast United States, from Florida to North Carolina. The length of stopover at these locations is generally believed to be brief (Niles *et al.* 2008, p. 29), but geolocator data show two birds stopped in South Carolina for longer periods (4 to 7 weeks) from late March through early May (Niles *et al.* 2012a, pp. 197–200). Spring red knot numbers tend to peak in the Southeast earlier than in the mid-Atlantic (Virginia to Massachusetts) (M. Bimbi pers. comm. June 27, 2013). Some birds migrate north in hops along coast from the Southeast to a final stopover location in the mid-Atlantic, while other birds use an inland route from the Southeast directly to central Canada, as discussed above.

Both southern- and northern-wintering red knots typically use mid-Atlantic stopovers from late April through late May or early June (Cohen *et al.* 2009, p. 941; Niles *et al.* 2008, p. 15). The stopover time in Delaware Bay for individual birds is about 10 to 14 days (Niles *et al.* 2008, p. 15). From Delaware Bay and other mid-Atlantic stopovers, birds tend to fly overland directly northwest to the central Canadian breeding grounds, with many stopping briefly along the shores of James and Hudson Bays (Bimbi *et al.* 2014, pp. 29–31; Niles *et al.* 197–200; Niles *et al.* 2010a, pp. 125–126; Cohen *et al.* 2009, p. 943; Niles *et al.* 2008, pp. 20, 24; Morrison and

Harrington 1992, p. 79). Large numbers of knots have been observed in James Bay, Canada, on the day following a mass departure from Delaware Bay. Red knots pass rapidly through southern James Bay, sometimes stopping less than one day. In some years, concentrations of red knots (400 to 1,500 birds, but more typically fewer than 100) have been observed at locations around the Great Lakes, and may represent weather-induced stops (eBird.org 2014; Morrison and Harrington 1992, p. 79). Geolocator data from one young bird at the Hudson Bay stopover revealed different behaviors than those typical of adults. During its second spring, en route to its first summer in the Arctic, this bird wandered back and forth along the southern coastline line from James Bay to the Nelson River delta. Even in the far north breeding area, this young bird wandered, moving north, then south, then north again (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014). It is possible these were exploratory movements by this young bird; see Wintering—Juveniles regarding an exploratory flight documented during that period.

One study of Delaware Bay and the Virginia barrier islands found high stopover site fidelity within a particular spring stopover season (Cohen *et al.* 2009, p. 942). Fidelity among years is apparently much lower. For example, geolocator data from a knot that passed through South Carolina in spring show an inland route to Canada, while resightings data indicate this same bird passed through Delaware Bay the previous year (Bimbi *et al.* 2014, p. 31). Further, Duerr *et al.* (2011, p. 16) found 60 percent of knots using the Virginia stopover did not use that site in the following year, although as many as 48 percent returned to Virginia in subsequent years. Similarly, between 2010 and 2012, 275 of the 493 birds banded in South Carolina (about 56 percent) were resighted in the State (SCDNR 2013, p. 30), consistent with other findings of moderate intra-year fidelity to a spring stopover area. Likewise, Watts (2009a) reported that 49 percent of marked red knots observed in Virginia had also been observed in Delaware Bay between 2005 and 2009. This included 326 movements between these spring stopover sites from year to year, and 88 movements that occurred within a given year. Preliminary analysis of these observations suggested that, in some years, movements of red knots from Delaware Bay to Virginia (southward) within a given spring migration season may equal the number of birds that move north from Virginia to Delaware Bay (Watts 2009a), a phenomenon known as reverse migration. Thus, there is some flexibility in red knots' use of the final Atlantic spring stopover site.

Both northbound and southbound intra-year movements have also been reported between two spring stopovers about 119 mi (192 km) apart on Argentina's coast—San Antonio Oeste (to the north) and Península Valdés (to the south) (see figure 4). Monitoring of marked birds was carried out between February and May from 2006 to 2010 at both sites. In this 5-year period, 75 knots were detected moving south from San Antonio Oeste to Península Valdés (i.e., reverse migration). Approximately 13 percent of the detected individuals made this southward movement in more than 1 year of the study, and some southward movements occurred in each year of the study. One red knot was observed in San Antonio Oeste and Península Valdés on the same day in 2009 (D'Amico *et al.* 2011, entire).

These observations from both Argentina and the mid-Atlantic suggest that red knots may use clusters of suitable habitats as a regional stopover complex. As the near and far ends of the Delaware Bay and Virginia stopovers are separated by about 35 and 155 mi (56 and 250 km), respectively, observed movements of marked birds from the mid-Atlantic and from Argentina show that separation distances of about 100 mi (160 km) between suitable habitats are not a barrier to intra-regional movements during a single spring migration season, even in a southward direction.

Over a smaller geographic scale, red knots stopping in spring at Península Valdés, have been observed using the two different embayments (San José Gulf and Nuevo Gulf) as a single trophic unit. It is thought that red knots take advantage of the inverted tidal cycles between these two gulfs (i.e., high tide in one gulf corresponds with low tide in the other) to avoid interruptions in foraging during high tide (Musmeci *et al.* 2012, pp. 357, 360–361). (Knots in Texas take similar advantage of opposite tidal cycles in the Upper and Lower Laguna Madre (Newstead 2014a, p. 17); see Migration and Wintering Habitat—Coastal.) Birds moving between these two gulfs at Península Valdés, sometimes whole flocks, travel 14.6 ± 5.2 mi (23.5 ± 8.4 km), with marked birds observed in both locations on the same day (Musmeci *et al.* 2012, pp. 357, 360–361). Similarly, some red knots that feed primarily in Delaware Bay during the spring stopover regularly fly 10 to 50 mi (16 to 81 km) across the Cape May Peninsula to use the extensive sandy beach, mud flats, and salt marshes in the vicinity of Stone Harbor, New Jersey for both foraging and roosting (Clark *et al.* 2009, pp. 87, 89; Niles *et al.* 2008, p. 44; Sitters 2005, p. 6; Sitters 2001, p. 2; Harrington 1996, p. 76). These findings support the idea that red knots often rely on clusters of suitable habitats during migration. Such clusters may also be important for the resiliency of red knots. For example, Musmeci *et al.* (2011, entire) reported that knots began feeding in Nuevo Gulf after a storm reduced invertebrate populations at a site in San José Gulf.

Fall Timing and Distribution

Departure from the breeding grounds begins in mid-July and continues through August. Females are thought to leave first, followed by males and then juveniles (Niles *et al.* 2008, pp. 14–15; Harrington 2001, p. 6). Adult *Calidris canutus* pass through stopover sites along the migratory route earlier in years with low reproductive success than in years with high reproductive success (Blomqvist *et al*. 2002, p. 149).

At each stopover, the adults gradually replace their red breeding plumage with white and gray, but southern-wintering birds generally they do not molt their flight or tail feathers until they reach their wintering areas (Harrington *et al.* 2010b, p. 361; Niles *et al.* 2008, p. 15; Morrison and Harrington 1992, pp. 79–80). In contrast, northern-wintering red knots (i.e., bound for the Southeast or northern Brazil) are thought to routinely molt their flight feathers during fall or early winter, north of the final wintering destination (A. Dey pers. comm. November 19, 2013; Harrington *et al.* 2010b, p. 362; Harrington *et al.* 2007, p. 39–41); see Differences in Migration Strategy by Wintering Area. However, Atkinson *et al.* (2005, pp. 738–746), found that some southern-wintering birds start molting flight feathers in northern areas, suspend this, and then finish their molt in the wintering areas, although these authors could not discern if the birds that had started to molt farther north were adults or immature birds.

Well-known fall stopover sites include southwest Hudson Bay (including the Nelson River delta), James Bay, the north shore of the St. Lawrence River, the Mingan Archipelago, and the Bay of Fundy in Canada; the coasts of Massachusetts and New Jersey and the mouth of the Altamaha River in Georgia, United States; the Caribbean (especially Puerto Rico and the Lesser Antilles); and the northern coast of South America from Brazil to Guyana (see figure 4) (eBird.org 2014; Autoridad de Energía Eléctrica (Electric Energy Authority, or AEE) 2013; Newstead *et al.* 2013, p. 57; Niles 2012a; D. Mizrahi pers. comm. October 16, 2011; Niles *et al.*

2010a, pp. 125–136; Schneider and Winn 2010, p. 3; Niles *et al.* 2008, pp. 30, 75, 94; B. Harrington pers. comm. March 31, 2006; Antas and Nascimento 1996, p. 66; Morrison and Harrington 1992, p. 74; Spaans 1978, p. 72). However, birds can occur all along the coasts in suitable habitat.

Based on surveys from 1975 to 1978, Morrison and Harrington (1992, p. 75) reported that 60 to 90 percent of southbound migrating red knots on the U.S. Atlantic coast occurred in 2 states, Massachusetts and New Jersey, and the relative importance of sites within these States varied from year to year. Islands at the mouth of the Altamaha River, Georgia, support a large late-summer and fall staging site (Schneider and Winn 2010, p. 2); see Population Surveys and Estimates—Fall Stopover Areas. Based on modeling using resighting data from fall 2011, the average stopover duration on the Altamaha is estimated at 30.2 days (95 percent confidence interval: 28.1 to 32.2 days) (GDNR 2013). Based on resighting and stable isotope data, between 83 and 96 percent of the red knots using the Altamaha in fall were part of the northern-wintering group (GDNR 2013).

In the mid-Atlantic, southbound red knots start arriving in July (Harrington 2001, p. 2). Numbers of adults peak in mid-August and most depart by late September, although geolocators have shown some birds (especially northern-wintering knots) stay through November (Niles *et al.* 2012a, pp. 197–200). Migrant juveniles begin to appear along the U.S. Atlantic coast in mid-August, occurring in much lower numbers and scattered over a much wider area than adults (Harrington 2001, p. 2; Morrison and Harrington 1992, p. 75). On Cape Cod, Massachusetts from 2005 through 2008, Harrington *et al.* (2010b, p. 360) found juvenile knots first arrive in mid-August and reach their highest numbers in mid-September, comprising about 3 percent of all the knots seen after August 15. In Cape Romain NWR in South Carolina, fall knot numbers peaked in August from 2008 to 2010 (Wallover *et al.* 2014, p. 6). In French Guiana, the first red knots arrive by mid-July, but biggest flocks are encountered from the third week of August to mid-September (CSRPN 2013). Based on banding studies from 1984 to 1991, Antas and Nascimento (1996, p. 63) found that southbound migrants arriving in Brazil originated from South Carolina and Massachusetts in July and August, and from Guyana in September. Numbers of southbound red knots on the Atlantic coast of Brazil, just north of Lagoa do Peixe, peaked in September of 2008 and 2009 (Scherer and Petry 2012, p. 46).

The Caribbean islands may be an important refuge for shorebirds migrating during storms (Nebel 2011, p. 217). Red knots are usually seen in Puerto Rico during southbound migration between October and February (AEE 2013). However, several studies suggest that adult red knots fly directly to South America from the eastern seaboard of the United States, arriving in northern South America in August (Niles *et al.* 2008, p. 29). Geolocator data from one bird that wintered in Tierra del Fuego showed a fall migration from the Arctic through Massachusetts, then over open ocean to the boundary area between Venezuela and Guyana; this bird then flew overland through the interior of South America to the coast of Argentina, finally working its way down the Patagonian coast to Tierra del Fuego (Niles 2011a). This geolocator result of an inland route over Amazonia is consistent with observations of two birds at Bahía de Asunción in Paraguay, one in October 1987 and one in November 1988 (Niles *et al.* 2008, p. 29; Hayes *et al.* 1990, p. 953), as well as two birds in interior Argentina (Laguna Mar Chiquita) in November 1976 (eBird.org 2014).

As with the red knot bound for Tierra del Fuego, geolocators on four birds wintering in Brazil all showed an over-water route from the eastern seaboard of the United States to northern South America. Two of these birds stopped in the Lesser Antilles for a few days in September, and three of these birds added hundreds of flight miles to avoid storms (Niles 2014; Niles *et al*. 2010a, pp. 125–126). To avoid a storm, one of these birds flew 1,000 mi (1,600 km) out into the ocean, more than halfway to Africa (Niles 2014).

Geolocators on eight birds heading south from Massachusetts to Southeast-Caribbean wintering areas all followed a direct route over water, though much closer to the coast than the birds heading to Brazil. These birds tended to stay on the U.S. coast later into the fall, typically departing in early November. Only two of the eight knots departing from Massachusetts stopped en route—one in Virginia and one in South Carolina (Niles *et al.* 2012a, pp. 197–200). Geolocator data on two birds that wintered on the Gulf coastal of Florida both showed an overland route over the Great Lakes to stopovers on the Southeast coast (Bimbi *et al.* 2014, pp. 29–30).

Geolocator data are available from two juvenile birds, fitted with geolocators in Massachusetts during their first fall. From Massachusetts, one bird went straight to North Carolina, where it spent its first winter. After spending its first (nonbreeding) summer back in Massachusetts, this bird's second fall migration was straight to Cuba, where it spent its second winter. After breeding in the Arctic its second summer, this young bird's third fall migration included stops in Hudson Bay, as well as a stop in Jamaica Bay, New York, which is unusual. This third fall migration bypassed Massachusetts, heading straight to Cuba from New York in August (S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014), earlier than many northern-wintering birds that typically stop in Massachusetts for one or two months. We do not know if this bird will resume use of the Massachusetts fall stopover as it matures. The second juvenile migrated from Massachusetts straight to Cuba, where it spent its first winter, and again straight to Cuba from its first-year (nonbreeding) summering area in Georgia. This bird arrived early in Cuba its second winter, in early September, then headed north again before returning to Cuba in November (see Wintering—Juveniles) (S. Koch, L. Niles, R. Porter and F. Sanders pers. comm. August 8 and 12, 2014).

Midcontinent

Spring

Geolocator results from seven red knots (one with 2 years of data) wintering in Texas showed that all these birds used a central, overland flyway across the midcontinental United States (Newstead *et al.* 2013, p. 56). Birds flew 1,600 to 2,000 mi (2,600 to 3,300 km) to the first stopover (Niles 2012a). A Northern Great Plains stopover (Saskatchewan, Canada, and, for one bird, North Dakota, United States) was used by five of six birds in 2010, while southern Hudson Bay in Manitoba, Canada (the Nelson River delta and James Bay), was used by one bird in 2010 and all three birds in 2011(Newstead *et al.* 2013, p. 56). These findings support earlier reports of large numbers (1,000 to 2,500) of red knots in Saskatchewan and Alberta, Canada, between January and June (Skagen *et al.* 1999). Following publication of the Newstead *et al.* (2013) study results, additional geolocator data became available showing six additional red knots stopping on the U.S. side of the Northern Plains—three in North Dakota, two in Montana, and one possibly in Nebraska (D. Newstead pers. comm. May 16, 2014).

All birds in this geolocator study departed Texas between May 16 and 21 (Newstead *et al.* 2013, p. 58). The northbound migration generally consisted of either a 2-day direct flight to a stopover site in the Northern Great Plains, or a 3-day flight to stopover at the southern edge of Hudson Bay in Manitoba or Ontario (Nelson River Delta and James Bay). Migratory flight time was 51.5±16.9 hours with a minimum migration speed of 36.4±8.0 mi per hour (58.5±12.8 km per hour) (Newstead *et al.* 2013, p. 56). These birds spent an average of 18.3 ± 3.2 days (range of 13 to 22 days) at the northbound stopover, departing for the breeding grounds between June 1 and 13 (Newstead *et al.* 2013, p. 56). Breeding-age birds observed on Texas beaches prior to departure in early May are typically in full breeding plumage, distinct from juveniles and possibly other nonbreeding birds, indicating that the molt to breeding plumage takes place in Texas (Newstead *et al.* 2013, p. 58).

Newstead *et al.* (2013, p. 58) suggest that the use of the Nelson River delta and the southwest shore of Hudson Bay may have gone undetected in previous years due to lack of survey effort directed at these remote areas. Geolocator data from other recent studies indicate this is also an important spring stopover area for rufa knots that migrate along the Atlantic coast (Niles *et al.* 2012a, pp. 197–200; Niles *et al.* 2010a, pp. 125–128). Harrington *et al.* ((2010b, entire) found that birds with different wintering destinations (Southeast United States and Argentina-Chile) used the same areas during fall migration in Massachusetts, but showed very little temporal overlap (see Northern Versus Southern Wintering Regions, above, and Differences in Migration Strategy by Wintering Region, below). In contrast, the collective findings of Bimbi *et al.* (2014, pp. 29–31), Niles (2014), Newstead *et al.* (2013, entire), Niles *et al.* (2012a, pp. 197–200), Niles 2011b, and Niles *et al.* (2010a, p. 126) show that birds wintering in the Northwest Gulf of Mexico as well Atlantic coast migrants (that winter in the Southeast and South America) probably co-occur both spatially and temporally along southwestern Hudson Bay during spring migration, at least in some years (Newstead *et al.* 2013, p. 58).

Fall

Geolocators on seven birds wintering in Texas showed they all used a similar and direct interior flyway across the midcontinental United States during the southbound migration. All of these birds arrived at a southbound stopover site on the south shore of Hudson Bay (Nelson River Delta to James Bay) between July 9 and 31. Mean stopover duration was 11.2±3.9 days (range 6 to 16 days) before departure on a 2- or 3-day direct flight back to the coast of the Gulf of Mexico. All Gulf arrivals were in Texas except one, which arrived first near New Orleans, Louisiana, spending 3 days there before heading west to Texas. Migratory flight time was 62.0±7.0 hours with a minimum migration speed of 33.9±3.6 mi/hour (54.6±5.8 km/hour) (Newstead *et al.* 2013, p. 56; Niles 2012a). For knots wintering in the Northwest Gulf of Mexico, the primary molt in fall is thought to occur on the wintering grounds. Adult birds captured in Texas in October were in the latter half of primary molt, which is likely to have been initiated after completion of southbound migration (Newstead *et al.* 2013. p. 58).

Flyway Fidelity

Geolocator results (Bimbi *et al.* 2014, pp. 29–31; Niles 2014; Newstead *et al.* 2013; Niles *et al.* 2012a, p. 197-200; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, pp. 125–128) have suggested that rufa red knots exhibit strong flyway fidelity (Newstead *et al.* 2013, p. 58) (i.e., not switching between Atlantic coast and midcontinental routes). However, newer geolocator data, as yet unpublished, do show some switching between these two flyways. David Newstead (pers. comm. May 8, 2014) reported that several Texas-wintering birds have been shown to use the “typical” midcontinental flyway in spring, but then follow a fall migration route along the U.S. Atlantic coast before returning Texas via the Gulf coast. To date, no known geolocator tracks from Texas birds have shown use of the Atlantic coast during spring migration, but some resighting data suggest that this may also occur (D. Newstead pers. comm. May 8, 2014).

Supporting these geolocator results, resightings of marked birds show considerable movement of birds between Texas and known *rufa* areas, including both the Southeast wintering area and the Delaware Bay stopover site (D. Newstead pers. comm. May 13, 2014; BandedBirds.org 2012; D. Newstead pers. comm. August 20, 2012; Niles *et al.* 2008, p. 74). For example, we evaluated data from BandedBirds.org (2012) regarding 242 marked red knots that were banded or resighted in Texas, including 139 that had only 1 record in the database (*e.g.*, they had been banded in Texas and not yet resighted anywhere). Thus, 103 birds had more than 1 record (i.e., a resighting history). Thirty-three of these marked birds known to occur in Texas were also observed on the Gulf coast of Florida or along the Atlantic coast (including some in Delaware Bay). These 33 birds represent about 13.6 percent of the total 242 marked birds, and about 32 percent of the 103 marked birds with a resighting history. Red knots marked in Texas and resighted along the Atlantic coast are often seen together with knots marked on the Atlantic (D. Newstead pers. comm. May 8, 2014). In addition, at least one bird marked in Argentina passed through Texas during spring migration (Niles *et al.* 2008, p. 74; González *et al.* 2006, p. 110). Together, these data indicate that at least some Texas-wintering knots do mix with Atlantic coast birds during migration (also see discussion of mixing at Hudson Bay under Spring, above), and suggest a complex pattern of movements of rufa red knots between Texas and the Atlantic coast that is not yet fully understood.

Three full years of geolocator data are available for one Texas-wintering red knot that demonstrates considerable variability in migratory strategy among years. Although certain stopover sites were used in multiple years, the actual routes and number of stopovers varied considerably from year to year (D. Newstead pers. comm. May 8, 2014). All of this bird’s northbound tracks were west of its southbound tracks (D. Newstead pers. comm. May 8, 2014), typical of an elliptical migration path common in shorebirds (Cornell Lab of Ornithology 2007). In one year, this bird followed northbound and southbound tracks spaced relatively close to one another, both passing over Minnesota. In a second year, the northbound and southbound tracks were widely separated, passing over North Dakota in spring and over the Great Lakes in fall. In the third year, this bird again passed over North Dakota in spring, but followed the Atlantic and Gulf coasts in fall (D. Newstead pers. comm. May 8, 2014).

As discussed above (Migration, Atlantic Coast), most data show that knots wintering along the central Gulf coast of Florida mainly use a coastal migration route, moving north to points between the Carolinas and Massachusetts before flying overland to Canada (Bimbi *et al.* 2014, pp. 29–30; Niles *et al.* 2012a, pp. 197–200; BandedBirds.org 2012; Atkinson *et al.* 2006a, p. 536; Atkinson *et al.* 2005, p. 738). However, Morrison and Harrington (1992, p. 77) reported the sighting of one marked Florida bird on northward migration in Alberta, Canada, hinting that some Florida birds may migrate northwest overland directly from Florida’s Gulf coast, potentially crossing paths with Texas-wintering birds. Further, we lack information regarding the migration routes of those birds that winter from central Texas through the Florida panhandle. Thus, there are also no data to indicate where along the Gulf coast the change from predominantly coastal (Florida) to predominantly midcontinental (south Texas) migration routes occurs, if the change is gradual or abrupt, or how frequently birds may switch between the two routes from year to year.

Stopovers

Other than the Northern Plains of southern Saskatchewan (likely extending into the U.S. portion of the Northern Plains), we are not currently aware of any consistently used stopovers in the Central Flyway. However, there are clusters of sightings records in both the midcontinent and farther east through the Mississippi Valley and along the Great Lakes. These cluster areas may warrant further study to more fully evaluate their usage as red knot stopovers. For example, looking at only eBird records 25 miles from an ocean coast within the rufa red knot range (across seasons and years), 134 U.S. counties had more than 1 record of this species, and 94 counties had more than 2 records, as of February 2014. These 134 counties show considerable clustering along the Great Lakes, the Mississippi River and its tributaries, and other major rivers and water bodies. Looking only the Central Flyway States of Montana, Wyoming, Colorado, North Dakota, South Dakota, Nebraska, Kansas, Oklahoma, and Texas (greater than 25 miles from the coast), we found 26 counties with more than 1 red knot record. These 26 counties contain 207 records with an average of 4.73 birds per record. Of these 207 records, 79 are located in just 1 county, where the maximum bird count was 160 knots. The remaining 25 counties had a mean of more than 5 records each. Many of these 26 counties are clustered along the South Platte and Arkansas Rivers (see figure 6). We note that some of these eBird records likely represent multiple reports of the same bird or flock, and that these birds were reported over multiple years and during different seasons. On the other hand, many other sightings data are not available through eBird. For example, along a portion of the Missouri River in North Dakota with only one eBird record, red knots have been "regularly" observed, but not recorded, incidental to other shorebird research since 2005 (C. Dovichin pers. comm. May 6, 2014). See also Population Surveys and Estimates—Inland Areas.

Subspecies Overlap

As discussed under Subspecies Nonbreeding Ranges, some movements of Pacific coast birds (confirmed or likely *Calidris canutus roselaari*) through Texas have been documented during spring migration. Best available data indicate that *C. canutus* in the Northwest Gulf of Mexico wintering area are wholly or predominantly the *rufa* subspecies during winter, summer, and fall. However, we conclude that both subspecies occur in Texas during spring (D. Newstead pers. comm. February 11, 2014). The prevalence of *roselaari* among *C. canutus* in Texas in spring is still unknown. Although marked birds of both subspecies have been observed in Texas, to date none of the 1,250 marked *C.c. roselaari* have been observed on the Atlantic coasts of North or South America (or on the Gulf coast of Florida), and none of the thousands of marked *C.c. rufa* have been observed on the Pacific coast north of Panama. Although considerable uncertainty remains, we conclude from best available data that the Texas coast is likely the only U.S. area of appreciable overlap between the *rufa* and *roselaari* subspecies during spring migration, with no documented overlaps in the United States during fall or winter.

Moreover, all available geolocator data to date confirm the existence of distinct breeding areas for the *rufa* and *roselaari* subspecies (Bimbi *et al.* 2014, pp. 29–31; Niles 2014; S. Koch, L. Niles, and R. Porter pers. comm. August 12, 2014; D. Newstead pers. comm. February 13, 2014; L. Niles pers. comm. January 4, 2013; Newstead *et al.* 2013, p. 56; Niles *et al.* 2012a, pp. 197–200; Niles 2012a; Niles 2011a; Niles 2011b; Niles *et al.* 2010a, pp. 125–126). Although it is not possible to pinpoint breeding locations above the Arctic circle from geolocator data, results to date do not suggest any separation in breeding range between those rufa red knots using midcontinental versus coastal migration routes (D. Newstead pers. comm. May 8, 2014).

Differences in Migration Strategy by Wintering Region

Red knots from different wintering areas appear to employ different migration strategies, including differences in molt, timing, routes, and stopover areas. Use of a predominantly midcontinental flyway by birds wintering in Texas, for both spring and fall migration, was described under Migration, as was use of an inland route by some knots wintering in the Southeast.

In addition, differences have been observed between those northern-wintering (Southeastern) knots following a coastal spring migration route compared to southern-wintering birds from Argentina and Chile that also typically migrate along the U.S. coasts. Early evidence of at least partial segregation of northern- versus southern-wintering knots during migration came from Harrington *et al.* (1988, pp. 440–441), who marked 444 red knots in Argentina and Florida between January and April 1981. Relative to their band-class sizes, Argentina knots were resighted 4.5 times more often in Delaware Bay (during spring migration) and Massachusetts (during fall migration) than were Florida-banded birds (Harrington *et al.* 1988, pp. 440–441). Southern-wintering knots continue to comprise a majority of Delaware Bay birds in spring, while in Massachusetts there has been a documented shift over recent decades from predominantly southern-wintering to predominantly northern-wintering birds in fall (Harrington *et al.* 2010a, entire). As discussed above in Migration, and below in Migration and Wintering Food, resighting, geolocator, stable isotope, and radio tracking data all suggest that southern-wintering red knots may be more reliant on the Delaware Bay spring stopover area than are birds that winter in the Southeast (Bimbi *et al.* 2014, p. 31; SCDNR 2013, p. 38; Niles *et al.* 2012a, pp. 197–200; P. Atkinson pers. comm. November 8, 2012; Niles 2011a; Niles *et al.* 2010a, entire; Cohen *et al.* 2010a, p. 660; Atkinson *et al.* 2006a, p. 536; Atkinson *et al.* 2005, p. 738; Harrington 2005a, p. 1).

Although northern- and southern-wintering knots do mix in Delaware Bay during spring (table 2), differences have been observed in their migration strategies. Baker *et al.* (2004, p. 878) reported that arrival time of the southernmost-wintering birds in relation to the more northerly birds has differed over time, but in most years southern birds appear to arrive in Delaware Bay later than northern birds. In 2000 and 2001, the highest proportion of southern-wintering birds occurred at or after the peak aerial count (Baker *et al.* 2004, p. 878) (see Factor E—Asynchronies—Delaware Bay—Late Arrivals). In spring 2005, Atkinson *et al.* (in IWSG 2005, p. 15) used observations of individually marked red knots to estimate return rates and residence time of birds in Delaware Bay. Based on observations of about 500 birds for which wintering areas were known from stable isotope analysis, both return rates and residence times within the bay varied by wintering region. There were also slight differences in arrival and departure patterns. The longest distance migrants from Argentina and Chile showed lower return rates and greater heterogeneity in arrival, such that the total passage period of the group was longer, though the stopover length of individuals did not significantly exceed that of other groups (Atkinson *et al.* in IWSG 2005, p. 15). Lower return rates for southern-wintering knots reported by Atkinson *et al.* (in IWSG 2005, p. 15) appears consistent with Harrington *et al.* (1988, p. 442), who reported return rates (survival rates not adjusted for emigration) in Florida wintering areas nearly double those in Massachusetts or Delaware Bay stopovers, although we know that these stopovers are not currently as dominated by southern-wintering birds as was believed in past decades. As in Delaware Bay, resighting data indicate spring timing differences in South Carolina, as South American birds appear to move through in May and June, later than red knots that winter in the Southeast (Bimbi *et al.* 2014, p. 27).

Northern- versus southern-wintering knots also have different strategies in southward migration (B. Harrington pers. comm. November 14, 2013; Harrington *et al.* 2007, p. 39–41). In general, the southern group passes through Atlantic regions of North America before September (Niles 2011a; Niles *et al.* 2010a, pp. 125–126), and strongly depends upon being able to accumulate fat and protein prior to departing on over-ocean flights between North and South America (A. Dey pers. comm. February 24, 2014; B. Harrington pers. comm. November 14, 2013; Harrington *et al.* 2007, p. 39–41). In contrast, northern-wintering birds linger on the North American coast (e.g., Massachusetts, New Jersey, Georgia coasts) often through early November (Niles *et al.* 2012a, pp. 197–200), typically using fall stopover locations as molting areas (A. Dey pers. comm. November 19, 2013; Harrington *et al.* 2010b, p. 362; Harrington *et al.* 2007, p. 39–41). Unlike southern-wintering birds, the southbound migration of the northern-wintering knots is typically finished following completion of their body and flight feather molt (Harrington *et al.* 2010b, p. 361; Harrington *et al.* 2007, p. 39–41). The resource requirements of these two groups during fall migration are quite different (Harrington *et al.* 2010a; Harrington *et al.* 2007, p. 39–41) and, at least in some areas, northern- and southern-wintering knots use different food and habitat resources (B. Harrington pers. comm. November 14, 2013; Harrington *et al.* 2010b, p. 362).

From research conducted over more than two decades, Harrington *et al.* (2007, p. 37) concluded that most knots transiting through Massachusetts during southward migration had South American destinations, while most of those transiting through Georgia were en route to northern wintering areas. Although newer studies now show a considerable proportion of migrants in Massachusetts are headed to northern wintering areas, this has changed over time and was not the case in past decades (Harrington *et al.* 2010a, entire). Thus, the results of Harrington *et al.* (2007) showed a substantial degree of separation between northern- and southern-wintering knots during fall migration. Likewise, although South Carolina supports some areas with substantial numbers of fall migrants (Bimbi *et al.* 2014, p. 27; Wallover *et al.* 2014, p. 6), no southern-wintering knots have been documented moving through that State in fall.

Although northern- and southern-wintering birds do now mix in Massachusetts stopover areas in fall, Harrington *et al.* (2010b, p. 357) found that red knots bound for different wintering areas (Argentina-Chile versus the Southeast) had different migration chronologies, plumage characteristics, flight feather molt, foods, foraging habitats, stopover durations, and uses of foraging and roosting habitats. Passage of the southern group through Cape Cod is earlier and more rapid than passage of the knots traveling to the Southeast. The passage of adult knots from the southern group is effectively complete before September, whereas knots from the northern group remain on Cape Cod into late October. Knots from the two groups have different strategic uses of the Cape Cod stopover location, with the southern-wintering knots using it as a migration staging point and the remaining knots using it as a molting area. These authors did not see flight feather molt in any knot that had been banded in South America (Harrington *et al.* 2010b, p. 361). Knots from the two wintering groups also used different habitats and foods in Cape Cod in fall, with southern-wintering birds selecting blue mussels (*Mytilus edulis*) and northern-wintering birds selecting thicker-shelled gem clams (*Gemma gemma*) (Harrington *et al.* 2010b, p. 361). Relative to northern-wintering birds, greater reliance of southern-wintering knots on thinner-shelled (i.e., higher-quality) prey is consistent with a shorter stopover duration on Cape Cod (i.e., less time to gain weight), and with the potentially higher energetic demands and more dramatic physiological transformations that these longest-distance migrants are thought to undergo (see Migration and Wintering Food).

Notwithstanding the differences described above, there is not full segregation of migration strategies, routes, or stopover areas among red knots from different wintering areas. For example, knots from northern and southern wintering areas are known to mix at mid-Atlantic spring stopovers (see table 2). Although fall migrants in Georgia have been dominated by northern-wintering birds, small numbers of Georgia-marked birds have been resighted in Argentina-Chile (Harrington *et al.* 2007, p. 38), showing that some degree of mixing does occur in Georgia during fall. In addition, resightings data show red knot movements between Texas and the Atlantic coast (D. Newstead pers. comm. May 13, 2014; BandedBirds.org 2012; D. Newstead pers. comm. August 20, 2012; Niles *et al.* 2008, p. 74); see Migration—Midcontinent—Flyway Fidelity.

MIGRATION AND WINTERING HABITAT

Coastal

Coastal habitats used by red knots in migration and wintering areas are similar in character (Harrington 2001, p. 9), generally coastal marine and estuarine (partially enclosed tidal area where fresh and salt water mixes) habitats with large areas of exposed intertidal sediments. Migration and wintering habitats include both high-energy ocean- or bay-front areas, as well as tidal flats in more sheltered bays and lagoons (Harrington 2001, p. 9). Preferred wintering and migration microhabitats are muddy or sandy coastal areas, specifically, the mouths of bays and estuaries, unimproved tidal inlets and tidal flats (Niles *et al.* 2008, p. 30; Harrington 2001, p. 8). In many wintering and stopover areas, quality high-tide roosting habitat (i.e., close to feeding areas, protected from predators, with sufficient space during the highest tides, free from excessive human disturbance) is limited (CSRPN 2013; K. Kalasz pers. comm. November 26, 2012; L. Niles pers. comm. November 19 and 20, 2012; Kalasz 2008, p. 9). In nonbreeding habitats, *Calidris canutus* require sparse vegetation to avoid predation (Niles *et al.* 2008, p. 44; Piersma *et al.* 1993, pp. 338–339, 349).

North America

In North America, red knots are commonly found along sandy, gravel, or cobble beaches, tidal mudflats, salt marshes, shallow coastal impoundments and lagoons, and peat banks (Cohen *et al*. 2010b, pp. 355, 358–359; Cohen *et al*. 2009, p. 940; Niles *et al*. 2008, pp. 30, 47;

Harrington 2001, pp. 8–9; Truitt *et al*. 2001, p. 12). In the Mingan Islands Archipelago in Canada (Province of Quebec), migrant red knots forage on limestone flats that are similar in habitat structure to red knot foraging areas in Argentina (USFWS 2011b, p. 8). In Newfoundland, Canada, red knots occur mostly on shorelines, sandflats and salt marshes, especially hummocky salt marsh. Open, sandy estuaries have been identified as prime habitat for this species in Newfoundland and Labrador, with rotting kelp deposits ranked as the second best habitat type (Garland and Thomas 2009, p. 4). In Massachusetts, red knots use sandy beaches and tidal mudflats during fall migration. In New York and the Atlantic coast of New Jersey, knots use sandy beaches during spring and fall migration (Niles *et al.* 2008, p. 30).

In Delaware Bay, red knots are found primarily on beaches of sand or peat at the mouths of tidal creeks, along the edge of tidal marshes dominated by salt marsh cordgrass (*Spartina alterniflora*) and saltmeadow cordgrass (*S. patens*), and in salt pannes (shallow, high salinity, mud-bottomed depressions on the marsh surface) and shallow coastal ponds or embayments (K. Clark pers. comm. April 11, 2012; Cohen *et al.* 2009, p. 940; Niles *et al.* 2008, pp. 33–34; Karpanty *et al.* 2006, p. 1706; Meyer 1999, p. ii; Burger *et al.* 1997, p. 288). Radio tracking showed that most of the time red knots roosted along the shoreline or in sandy washovers above the high tide line, but knots also roosted in bare, shallow-water openings 0.5 to 1.3 mi (850 to 2,050 m) inland in the adjacent salt marsh (Zimmerman 2010, pp. 9, 13, 26). The preference for inland roost sites was greater at night and during spring tides, and Delaware Bay is the only area in which rufa red knots have been observed roosting inland (Zimmerman 2010, pp. 1, 13). Some red knots that feed primarily in Delaware Bay regularly move to the Atlantic coast of New Jersey to feed and roost in inlet habitats (Clark *et al.* 2009, pp. 87, 89; Niles *et al.* 2008, p. 44; Sitters 2005, p. 6; Sitters 2001, p. 2; Harrington 1996, p. 76). In Delaware, salt pannes are an important roosting habitat (K. Kalasz pers. comm. November 17, 2011).

In the southeastern U.S., red knots forage along sandy beaches during spring and fall migration from Maryland through Florida. During migration, knots also use the tidal mudflats in Maryland and along North Carolina's barrier islands. Red knots also forage along peat banks for mussel spat in Virginia and along small pockets of peat banks where the beach is eroding in Georgia (Niles *et al.* 2008, p. 47). In North Carolina, where red knots occur in fall, winter, and spring, the birds primarily use ocean-front beaches and inlets between barrier islands, as well as sandy shoals. North Carolina roost sites are above the mean high tide line and on sandy shoals (North Carolina Wildlife Resource Commission (NCWRC) 2013). In Florida, red knots use salt marshes, brackish lagoons, tidal mudflats, and mangrove areas (Niles *et al*. 2008, p. 47). In Lee County on Florida's Gulf coast, Lott *et al.* (2009, pp. 18–19) found that red knots during fall migration used intertidal substrates on ocean beaches, almost exclusively at inlets. Red knots are rarely seen away from the sandy Gulf coast beaches in Louisiana (Purrington 2012, p. 65; Lowery 1974, p. 308), but will sometimes be found on mudflats near the beach at the height of migration (Purrington 2012, p. 65), even occurring occasionally in rice fields and coastal marshes as far inland as the Intracoastal Waterway and Lake Calcasieau (Russell 2014, p. 4).

Along the Texas coast, red knots forage on beaches, oyster reefs, and exposed bay bottoms, and roost on high sandflats, reefs, and other sites protected from high tides (USFWS 2011b, p. 5). In addition to using the Gulf beaches, red knots in Texas use extensive tidal flats on the bay sides of barrier islands (Newstead *et al.* 2013, pp. 53, 58). A radiotelemetry study found that the water level in the Laguna Madre, Texas is a key driver of whether red knots were found in bay or beach habitats (Newstead 2014a, pp. 13–14). With lowest bay water levels, almost all relocations were in bay habitats, and the converse—during high bay water levels, most relocations were on the beach—was also true. Beach water levels were not as strong an influence (i.e., when beach water levels were low, this did not necessarily result in a higher proportion of relocations on the beach, unless bay water levels were high). When both bay and beach water levels were high, most birds were on the beach. These data illustrate that, in general, red knots prefer bayside habitats when available, at least during the period from fall through spring, but become dependent on beach habitats when bay water levels are high. This conclusion is also supported by the commonly observed phenomenon that red knots are present on the beach in higher numbers in the fall and spring, coinciding with high seasonal tides that typically push water levels much higher in bays to the point where wind-forcing events are less likely to expose preferred tidal flats (Newstead 2014a, pp. 13–14). While these findings illustrate that there is a preference for bay habitats when they are available in the Upper Laguna Madre, it is also clear that the beach is providing alternative habitat for substantial periods of the fall and spring. Red knots appear to be far more sensitive than piping plovers (*Charadrius melodus*) to high water levels in the bay. Although their habitat preferences are not mutually exclusive, in general knots are strongly associated with lower sandflat habitats, while plovers will opportunistically use lower sandflats when available but otherwise are most strongly associated with algal flats (Newstead 2014a, p. 14). Red knots make regional movements to take advantage of periods of inundated and exposed flats that are typically opposite between the Upper and Lower Laguna Madre. In addition to foraging in Laguna Madre, red knots commonly roost in its shallow waters. On the Gulf beaches, knots were sometimes encountered roosting in tight groups either just above the swash zone, or during spring atop mounds of *Sargassum* spp. algae (discussed below). However, knots on Texas' Gulf beaches were usually able to forage on the sand in the swash zone, seaward of the *Sargassum* (Newstead 2014a, p. 17).

Caribbean and South America

In the southern (Argentina-Chile) wintering areas, habitats include extensive mudflats, sandy beaches, and "restinga" formations (an intertidal shelf of densely-packed dirt blown by strong, offshore winds) (Niles *et al.* 2008, pp. 50, 52). Wintering and stopover habitats along Argentina's Patagonian coast include extensive sandy beaches, mudflats, and restingas, with gravel beaches in some areas (Niles *et al.* 2008, p. 50). One such Patagonian stopover site, Fracasso Beach, consists of a fine-sediment (fine sand and mud) tidal flat (Bala *et al.* 2002, p. 27). At the Lagoa do Peixe stopover site on Brazil's Atlantic coast, red knots use both the ocean beach and the lagoon for foraging; within the lagoon, red knots usually forage in areas covered by shallow water or, less often, on recently exposed flats covered by algae (Harrington *et al.* 1986, p. 49). In the wintering and migration stopover area at Maranhão, Brazil, red knots forage along sandy beaches, tidal mudflats, and mangroves, while red knots wintering in Panama use soft silty mudflats (Niles *et al.* 2008, pp. 48, 52). Migrants in Panama may also forage in mangroves and sandy beaches (Niles *et al.* 2008, p. 48).

In Puerto Rico, red knots may be spotted in several locations, but frequent the rocky coast of Piñones, the mouth of the Camuy River, and the mouth of the Cibuco River in Vega Baja near the Maritime Police Unit (AEE 2013). Red knots have been observed using rice fields in Trinidad (eBird.org 2014) and French Guiana (Niles 2012b). In French Guiana, red knots also use tidal mudflats and small rocks emerging at low tides (CSRPN 2013). They may gather on sand beaches or rice fields (situated just behind beaches) as high tide roosts. Most of French

Guiana's coastline consists of large mudflats and mangrove (*Avicennia* spp.) forests. The mudflats and mangrove forests undergo a 10 to 15-year cycle of sedimentation and erosion. Sand beaches are restricted to the Cayenne area of French Guiana (at the eastern part of the Space Center at Kourou) and the extreme northwest part of the coast (within the Natural Reserve of l'Amana) where they may be used by red knots as high tide roosting habitat. Rocky islets used by red knots in the Kourou-Malmanoury section (included in the Space Center area) provide suitable, though small, habitats for feeding, but are mostly used as high tides roosts (especially the "Battures de Malmanoury" and the "Pointe des Roches" on the left bank of the Kourou estuary). The largest rocky islands (Devil Islands off Kourou and the "Ilets de Rémire" off Cayenne, or the Grand Connétable off the Approuague estuary) are used by other shorebirds but not by red knots. The places where most red knots are observed in French Guiana have been known for decades and seem stable (Réserve de l'Amana, Battures de Malmanoury, and Pointe des Roches at Kourou). A number of knots marked in Argentina or Delaware Bay have been seen several times, suggesting that particular birds are faithful to French Guiana migration stopover sites. This fidelity can be explained by the rarity of these tidal roosts (CSRPN 2013).

In Suriname, red knots use sandy beaches and firm clay banks (Ottema and Spaans 2008, p. 341). In the early 1970s, small numbers of red knots were observed on firm and tough clay banks emerging from eroding coastline and in shallow lagoons, but knots were never found on soft tidal flats (Spaans 1978, p. 72). These observations suggest a deviation from the red knot's typical nonbreeding habitats. However, there are sandy beach habitats in Suriname's Marowijne District, less than 31 mi (50 km) from the site where approximately 1,700 red knots were observed in French Guiana. In addition, several hundred knots were observed using coastal mud flat habitat in the Commewijne district of Suriname during spring migration in April 2012 (New Jersey Audubon Society (NJAS, also known as New Jersey Audubon) *et al.* 2013). These reports suggest that red knots may also use more typical habitats in Suriname.

On the Pacific coast of Costa Rica, *Calidris canutus* has been reported using tidally exposed mud flats during winter (Smith and Stiles in Pitelka 1979, pp. 41–43). On the Pacific coast of Panama, *C. canutus* has been reported during winter and early spring using two sites of soft, silty mudflats, and a third site of hard mud and sand beach flanked by mangrove and marsh. All three Panamanian sites experience dry-season upwellings of nutrient-rich water that increase invertebrate abundance (Buehler 2002, p. 41). In Colombia's Iscuandé River Delta on the Pacific coast, *C. canutus* have been reported foraging on intertidal mudflats or resting on sandy beaches (Ruiz-Guerra (2011, p. 194). On Chiloé Island on the Pacific coast of Chile, *C. canutus* use intertidal flats and beaches in estuaries in the northern and eastern part of the island, favoring more sandy sites and not observed in the muddiest, freshwater sites (B. Andres pers. comm. July 21, 2014). We lack information regarding the subspecies composition of *C. canutus* in these Pacific coast habitats (see Subspecies Nonbreeding Distributions).

Wrack, Inlets, and Artificial Habitats

In Delaware Bay, red knots preferentially feed in microhabitats where horseshoe crab (*Limulus polyphemus*) eggs are concentrated, such as at horseshoe crab nests (Fraser *et al.* 2010, p. 99), at shoreline discontinuities (e.g., creek mouths) (Botton *et al.* 1994, p. 614), and in the wrack line (Nordstrom *et al.* 2006a, p. 438; Karpanty *et al.* 2011, pp. 990, 992). (The wrack line is the beach zone just above the high tide line where seaweed and other organic debris are deposited by the tides.) Wrack may also be a significant foraging microhabitat outside of

Delaware Bay, for example where mussel spat (i.e., juvenile stages) are attached to deposits of tide-cast material. Garland and Thomas (2009, p. 4) identified rotting kelp deposits as the second best red knot habitat type in Newfoundland and Labrador, Canada. Wrack material also concentrates certain invertebrates such as amphipods, insects, and marine worms (Kluft and Ginsberg 2009, p. vi), which are secondary prey species for red knots (see Migration and Wintering Food, below). However, most beach use by foraging red knots is seaward of tidal wrack lines, and is on infaunal (buried in the sediment) prey in the swash zone. Juvenile knots will sometimes forage in wrack, but this appears to be incidental use versus being a key habitat (B. Harrington pers. comm. November 14, 2013).

On the Gulf beaches of south Texas during spring, knots are sometimes encountered roosting in tight groups atop mounds of *Sargassum*, a planktonic brown algae that washes into Texas shores in spring and summer, sometimes forming “drifts” above the high tide line up to 1.6 ft (0.5 m) in height and covering about 33 ft (10 m) width of beach. On several occasions knots have been observed foraging among the *Sargassum*, but it is unclear what prey items they were capturing. This is not their typical foraging area. In fact, when tides reach them, the tall “walls” of *Sargassum* can prevent red knots from foraging in their preferred swash zone habitat for extended periods (Newstead 2014a, p. 17). In a study by Williams *et al.* (2008, pp. 6–7), red knots were not among the bird species documented feeding in or near *Sargassum* on Galveston Island, Texas.

For many shorebirds, the supra-tidal (above the high tide) sandy habitats of inlets provide important areas for roosting, especially at higher tides when intertidal habitats are inundated (Harrington 2008, pp. 4–5). For red knots, unimproved tidal inlets are a preferred nonbreeding habitat (NCWRC 2013; Lott *et al.* 2009, pp. 18–19; Niles *et al.* 2008, p. 30; Sitters 2005, entire; Harrington 2001, p. 8). Along the Atlantic coast, dynamic and ephemeral (lasting only briefly) features are important red knot habitats, including sand spits, islets, shoals, and sandbars, features often associated with inlets (Harrington 2008, p. 2; Harrington in Guilfoyle *et al.* 2007, pp. 18–19; Winn and Harrington in Guilfoyle *et al.* 2006, pp. 8–10). From South Carolina to Florida, red knots are found in significantly higher numbers at inlets than at other coastal sites (Harrington 2008, pp. 4–5).

In some localized areas, red knots will use artificial habitats that mimic natural conditions, such as nourished beaches, dredged spoil sites, elevated road causeways, or impoundments. For example, in Long Island and Delaware Bay, some nourished beaches and managed impoundments support important concentrations of red knots (Niles *et al.* 2008, pp. 30, 42). In Delaware Bay, Botton *et al.* (1994, p. 614) found that artificial obstructions such as jetties can, in the same manner as natural shoreline discontinuities, act to concentrate drifting horseshoe crab eggs and thereby attract shorebirds. In Florida, A. Schwarzer (pers. comm. March 25, 2013) has observed multiple instances of red knots using artificial structures such as docks, piers, jetties, causeways, and construction barriers, but we have limited information regarding the frequency, regularity, timing, or significance of red knots’ use of use of these artificial habitats. Also in Florida, red knots are known to use impoundments at Merritt Island NWR (L. Patrick pers. comm. March 27, 2013).

Inland

Red knots are restricted to ocean coasts during winter, and occur primarily along the coasts during migration. However, small to moderate numbers of rufa red knots are reported annually across the interior United States (i.e., greater than 25 miles from the Gulf or Atlantic Coasts) during spring and fall migration. These reported sightings are concentrated along the Great Lakes, but multiple reports have been made from every interior State (Central Flyway Council 2013; eBird.org 2014); see Migration—Midcontinent, above. Information is scarce on the specific noncoastal stopover habitats used by red knots.

Available information suggests that red knots use inland saline lakes as stopover habitat in the Northern Great Plains. For example, Skagen *et al.* (1999) reported peak counts of over 1,000 red knots between January and June over the period 1980 to 1996 at several saline lakes in the northern Plains of southern Canada. In May 2005, 25 red knots were observed at North Dakota's Long Lake NWR (North Dakota Game and Fish Department (NDGFD) 2013). Although we lack data on the specific habitat used during this incident, Long Lake is a natural saline lake (WHSRN 2012), suggesting that this may be a habitat type used by inland-migrating knots. Geolocator data show red knots stopping in the Northern Great Plains in spring ((Newstead *et al.* 2013, p. 57). Although the Northern Plains encompass a large area, most of the concentrations of red knots recorded in recent years have been at Chaplin, Reed, and Quill Lakes in Saskatchewan, Canada (Newstead *et al.* (2013, p. 57), all saline lakes. Further, geolocator data from one bird that stopped in North Dakota indicate this bird utilized a salty environment (e.g., natural alkaline lake or wetland) (D. Newstead pers. comm. May 16, 2014). Recognizing the importance of these Northern Plains saline lakes to shorebirds, including red knots, several have been designated Western Hemisphere Shorebird Reserve Network (WHSRN) sites (WHSRN 2012).

Athalassic (inland) saline lakes are defined as those which have had no connection to the sea in geologically recent times, or have been evaporated to dryness after having been flooded by marine waters and subsequently reflooded. Therefore, the fauna and flora in these waters are not directly derived from marine biota (Hammer 1986, p. 15). Most biologists use the following classification scheme for lakes: fresh water (less than 1 part per thousand (ppt)), subsaline (1 to 3 ppt), hyposaline (3 to 20 ppt), mesosaline (20 to 50 ppt), and hypersaline (greater than 50 ppt) (Last and Ginn 2009). The Northern Great Plains of southwestern Canada contain an estimated 3.5 million lakes and another 6 to 8 million "sloughs," most of which are saline or hypersaline. From the standpoint of salt lake geochemistry, the two most important physical features of this region are the high evaporation to precipitation ratios, and the presence of large areas that do not drain to the ocean. Although most of the lakes in Canada's Great Plains have similar origins, the waters show consider chemical diversity. Not only is there a complete spectrum of salinities from relatively dilute (0.1 ppt) to brines more than an order of magnitude greater than sea water, but also virtually every water chemistry type is represented in lakes of this region. Although generalizations should be viewed carefully, "average" lake water in Canada's Plains is about 30 ppt. Sulfate and carbonate-rich lakes clearly dominate Canada's Great Plains, comprising over 95 percent of the total lakes. It is generally well accepted that groundwater plays an important role in both the hydrology and chemistry of saline lakes. However, with a few notable exceptions, groundwater interaction processes with individual salt lake basins in Canada's Northern Great Plains are still poorly understood. A major complicating factor in characterizing the chemistry of the salt lakes of the Northern Great Plains is that many of the lakes exhibit playa characteristics, filling with water during the spring and early summer and drying completely by late summer or fall. About 85 percent of the salt lakes in this region are influenced by this type of seasonal hydrologic cycle (Last and Ginn 2009). Saline lakes are also distributed throughout the U.S. side of the Northern Great Plains (Skagen and Thompson 2013, pp. 15, 29; Kennedy 1994, p. 70; Last and Schweyen 1983, p. 246; McCarraher 1964, p. 1; Robinove *et al.* 1958, p. 1).

The biota differ significantly between fresh and saline lakes. At low salinities the species composition of salt lakes is comparable to that of their fresh water counterparts. As salinity increases, the diversity of species declines, and as salinities reach extremely high values, species diversity becomes very low. Saline and hypersaline lakes have some of the highest measured rates of organic productivity in the world. At moderate to high salinities (30 to 100 ppt), the main contributors to this biomass are green algae and cyanobacteria. At more elevated salinities, halophilic (salt-loving) bacteria dominate the ecosystem (Last and Ginn 2009).

Newstead *et al.* (2013, p. 57) reported on 2 years of geolocator data for red knots that wintered in Texas and migrated through the midcontinental United States and Canada. Though sample sizes from this study were small, it is notable that five of the six northbound migration tracks in 2010 showed stopovers in the Northern Great Plains, yet none of the 2011 tracks did (instead all three used the Nelson River Delta in Canada). Water levels in Saskatchewan's lakes are known to affect the abundance and distribution of migrant shorebirds, with high water levels drastically reducing the availability of habitat for shorebirds. Water levels were at or near normal in southern Saskatchewan during migration in 2010 but extremely high in 2011, while levels at Quill Lakes (central Saskatchewan) were extremely high in both years. Lack of suitable habitat due to high water levels could explain why birds in this study did not stop in the Northern Great Plains in 2011. It is unclear whether most inland-migrating knots also made the same decisions as those in this study (suggesting plasticity in migration strategies), or if a significant proportion of birds used Northern Great Plains stopovers in 2011 which may have had consequences for migratory fitness, survival, or both. For instance, in June 2011, 10 red knots (including one banded as part of this study) were killed on a road by a vehicle at Reed Lake in southern Saskatchewan. It has been suggested that high water levels resulted in many birds being forced to roost on roads during this period (Newstead *et al.* 2013, pp. 57–58).

We have little information to indicate whether or not red knots may also utilize inland freshwater habitats during migration, but some of the clusters of sightings along the Mississippi River and its tributaries discussed above (Migration—Midcontinent) suggest that certain freshwater areas may warrant further study as potential stopover habitats. In addition, along a portion of the Missouri River in North Dakota with only one reported eBird record, red knots have been "regularly" observed, but not recorded, incidental to other shorebird research since 2005 (C. Dovichin pers. comm. May 6, 2014) (see figure 6). In the Mississippi Flyway, red knots occasionally use wetlands and riverine sandbars (Russell 2014, entire).

Best available data indicate that small numbers of red knots sometimes use manmade freshwater habitats. Red knots are known to use Kitsam Reservoir in Canada (eBird.org 2014). Most of the sightings in North Dakota have been in sewage lagoons throughout the State. These usually involve only one or two birds at a time, but 13 knots used the West Fargo lagoons from July 31 to August 3, 2009 (NDGFD 2013; A. Simnor pers. comm. October 15, 2012). It is likely that the usage of these North Dakota lagoons is over represented in the available data, reflecting their proximity to human population centers (A. Simnor pers. comm. October 15, 2012). In Colorado, *Calidris canutus* have been recorded primarily at scattered reservoirs in the South Platte and Arkansas River drainages as individuals or small flocks, though no reservoirs are locations of consistent observations (B. Andres pers. comm. May 8, 2014; Central Flyway Council 2013) (see figure 6). All but 1 of the 40 Oklahoma red knot records reported by Oklahoma Department of Wildlife Conservation (ODWC 2013) from 1941 to 2012 occurred on man-made impoundments: 31 observations have been made on reservoirs, 1 on a sewage lagoon, 2 on fish hatchery ponds, 1 on a pond in a river flood plain, and 3 on managed wetland complexes. The only exception in Oklahoma is a single bird observed on the Arkansas River. There are no data to indicate that red knots use shallow wetlands, ponds, or streams on those rare occasions when they have been documented making landfall in Oklahoma (ODWC 2013). In the Mississippi Flyway, red knots are known to use falling reservoirs during fall migration, including in Iowa, Illinois, and Arkansas (Russell 2014, entire). Red knots occasionally use sewage treatment plants in southern Minnesota (Russell 2014, p. 4).

MIGRATION AND WINTERING FOOD

Across all subspecies, *Calidris canutus* is a specialized molluscivore, eating hard-shelled mollusks, sometimes supplemented with easily accessed softer invertebrate prey, such as shrimp- and crab-like organisms, marine worms, and horseshoe crab eggs (Piersma and van Gils 2011, p. 9; Harrington 2001, pp. 9–11). *Calidris canutus* do not necessarily prefer hard-shelled mollusks (in fact they do not, when given the choice), but they are specialized in finding and processing such prey. Due to this specialization, *C. canutus* have less ability to find the actively crawling soft-bodied worms and small crustaceans on which other sandpiper species specialize. One of this species' sensory capacities, the ability to use pressure gradients around hard objects in soft, wet sediments, has not been described in any other animal. This remote detection of buried hard-shelled prey is probably enabled by a bill-tip organ, a dense group of pressure sensors clustered in sensory pits of the mandibles. The mollusk prey is swallowed whole and crushed in the gizzard, which in *C. canutus* is the largest (relative to body size) among any shorebird species evaluated (Piersma and van Gils 2011, pp. 9–11). Large gizzards are another specialization to a mollusk diet, allowing *C. canutus* to grind the hard shells of its prey.

Calidris canutus prefer thin-shelled to thick-shelled prey species because they are easier to digest and provide a more favorable meat to mass ratio (higher prey quality) (Harrington 2001, p. 11; Zwarts and Blomert 1992, p. 113). From studies of other subspecies, Zwarts and Blomert (1992, p. 113) concluded that *C. canutus* cannot ingest prey with a circumference greater than 1.2 in (30 millimeters (mm)). Rufa red knots in San Antonio Oeste, Argentina were found to select mussels between 0.20 and 0.79 in (5 and 20 mm) long out of an available range of 0.04 to 1.10 in (1 to 28 mm) (Cohen *et al.* 2010b, p. 360; González *et al.* 1996, p. 575). In Virginia, the length of *Donax* clams averaged 0.16 in (4.12 mm) in red knot foraging habitat during the peak of spring migration, and red knot flock sizes were positively correlated with mean *Donax* length (Cohen *et al.* 2010b, pp. 359–360). Foraging activity is largely dictated by tidal conditions, as *C. canutus* rarely wade in water more than 0.8 to 1.2 in (2 to 3 cm) deep (Harrington 2001, p. 10). Due to bill morphology, *C. canutus* is limited to foraging on only shallow-buried prey, within the top 0.8 to 1.2 in (2 to 3 cm) of sediment (Gerasimov 2009, p. 227; Zwarts and Blomert 1992, p. 113).

Table 1 gives prey items that have been reported for rufa red knots in wintering and migration areas. Though eaten by *C. canutus* at one location in Cape Cod, Massachusetts (*rufa*) and one site in California (*roselaari*), the thick, hard-shelled gem clam is rarely eaten despite its abundance in other red knot nonbreeding areas, including western Cape Cod Bay (fall migration) and Delaware Bay (spring migration) (Harrington *et al.* 2010b, pp. 361–362; Harrington 2001, p. 11). During fall migration in Cape Cod, Massachusetts, Harrington *et al.* (2010b, p. 361) found prey differences between northern-wintering (gem clams) and southern-wintering (blue mussels, *Mytilus edulis*) red knots; see Migration—Differences in Migration Strategy by Wintering Area.

Red knots and other shorebirds that are long-distance migrants must take advantage of seasonally abundant food resources at migration stopovers to build up fat reserves for the next nonstop, long-distance flight (Clark *et al.* 1993, p. 694). During the migration period, although foraging red knots can be found widely distributed in small numbers within suitable habitats, birds tend to concentrate in those areas where abundant food resources are consistently available from year to year. The spatial distribution of red knots in Argentina, Georgia, South Carolina, Virginia, the Atlantic coast of New Jersey, and Delaware Bay stopover areas has been correlated with the distribution of the primary prey species (GDNR 2013; Thibault and Levisen 2013, p. 6; SCDNR 2013, p. 37; Musmeci *et al.* 2011; Fraser *et al.* 2010, p. 97; Cohen *et al.* 2010b, p. 355; Cohen *et al.* 2010a, pp. 659, 660–661; Niles *et al.* 2008, pp. 17, 19; Smith *et al.* 2008, p. 15; Karpanty *et al.* 2006, p. 1706; Botton *et al*. 1994, p. 605).

Table 1. Reported red knot prey items in wintering and stopover areas

Prey	Location	Source
small periwinkles (*Littorina* spp.), tiny blue mussels (*Mytilus edulis*)	Mingan Islands, Canada	USFWS 2011b, p. 8
blue mussel (*Mytilus edulis*) spat (preferred), gem clams (*Gemma gemma*) (not preferred), amphipods, naticid snails	Massachusetts	Harrington 2001, p. 11
mussel spat (northern birds), gem clams (southern birds)	Cape Cod, Massachusetts	Harrington *et al.* 2010b, p. 359
horseshoe crab eggs (preferred, spring)	Moriches Bay, Long Island, New York	McKown *et al.* 2014, p. 1
mussel spat, clams	Stone Harbor, Atlantic coast of New Jersey	Sitters 2001, p. 4
polycheate worms, insect larvae, crustaceans, mussel spat, clams	Atlantic coast beaches and marshes, New Jersey	Cohen *et al.* 2010a, p. 659
horseshoe crab eggs (almost exclusively)	Delaware Bay, New Jersey and Delaware (spring)	Harrington 2001, p. 11; see section that follows
coquina clams (*Donax variabilis*), crustaceans	Virginia barrier islands	Cohen *et al.* 2010b, p. 355
blue mussel (*Mytilus edulis*) spat	Virginia barrier islands (especially on peat banks)	Karpanty *et al.* 2012, p. 1; Truitt *et al*. 2001, p. 12

Table 1. Reported red knot prey items in wintering and stopover areas

Prey	Location	Source
small bivalves such as coquina clams (*Donax* spp.); sand fleas (haustoriids); mole crabs (*Emerita talpoida*)	North Carolina	NCWRC 2013
coquina clams (*Donax variabilis*) (especially on Kiawah Island, probably preferred), horseshoe crab eggs and larvae (possibly preferred), amphipod crustaceans (mainly Haustoriids), polycheates, other mollusks	South Carolina (spring)	Thibault and Levisen 2013, pp. 3, 6; Harrington and Winn in Baker 2001, p. 12
small clams including coquina (*Donax* spp.) and dwarf surf (*Mulinia lateralis*) in winter; primarily *M. lateralis* in fall; also horseshoe crab eggs in spring	Georgia	GDNR 2013; Schneider and Winn 2010, p. 2
amphipod crustaceans (*Emerita* spp.), bivalves (*Donax* spp.)	Gulf coast of Florida, Southern Brazil, Gulf of Venezuela	Harrington 2001, p. 10
clams (*Donax* spp. (preferred) and *Crassatellidae* spp.), snails (*Cypraeidae* spp.)	Tampa Bay Region, Gulf coast of Florida	Schwarzer 2011, pp. 74–75
horseshoe crab eggs (preferred in spring, but present in low densities)	Cedar Key, Gulf coast of Florida	Schwarzer 2011, pp. 81–83
coquina clams (*Donax variabilis, D. texasianae*)	Gulf coast of Texas – Gulf beaches	Newstead 2014a, p. 17; D. Newstead pers. comm. March 5, 2013; Niles *et al.* 2009, p. 1
probably dwarf surf clams (*Mulinia lateralis*)	Gulf coast of Texas – Laguna Madre	Newstead 2014a, p. 17
unknown	French Guiana mudflats and rocky islets	CSRPN 2013
mainly bivalves (*Tellina, Macoma, Donax, Gemmula, Iphigenia, Tivella,* and *Arca* spp.), also gastropods	State of Ceará, northeast Brazil	Carlos *et al.* 2010, pp. 16, 17
Donax spp. (especially *D. hanleyanus*), *Emerita* spp.	State of Rio Grande do Sul, southeast Brazil	Vooren and Chiaradia 1990, p. 20
snails (*Littoridina australis*; *Heleobia* sp.) in the lagoon; small clams (*Donax* spp.) (primary) and *Emerita* spp. on the oceanside beaches	Lagoa do Peixe, Brazil (State of Rio Grande do Sul)	Niles *et al.* 2008, p. 49; Harrington *et al.* 1986, p. 50
mud snail (*Littoridina australis*) (preferred); Scarabaeidae beetle (*Dyscinetus* spp.)	Punta Rasa, Argentina (Bahía Samborombón)	Ieno *et al.* 2004, p. 493; Alemany *et al.* 2001, p. 17

Table 1. Reported red knot prey items in wintering and stopover areas

Prey	**Location**	**Source**
a clam (*Darina solenoides*) in sandflats, a small mussel (*Brachidontes rodriguezi*) on restingas	San Antonio Oeste, Argentina	Barzola *et al.* 2006, p. 60; González *et al.* 1996, p. 575
bivalves (*Darina solenoides* and *Tellina petitiana*)	Península Valdés, Argentina	Hernández 2009, p. 208
clam (*Darina solenoides*)	Fracasso Beach (north embayment of Península Valdés), Argentina	Musmeci *et al.* 2011, entire; Bala *et al.* 2002, p. 28
clam (*Darina solenoides*), polychaete (*Travisia olens*)	Colombo Beach (south embayment of Península Valdés), Argentina	Musmeci *et al.* 2011, entire
clam (*Darina solenoides*)	Bahía Lomas, Tierra del Fuego, Chile	Niles *et al.* 2008, p. 50; Espoz *et al.* in Lanctot 2007, p. 10
clam (*Darina solenoides*) in soft sediments, mussels (mainly *Mytilus edulis*, *Aulacomya ater*, and *Perumytilus purpuratus*) on restingas	Río Grande, Tierra del Fuego, Argentina	Escudero *et al.* 2012, p. 356; Baker *et al.* 1996, p. 103

Horseshoe Crab Eggs

A prominent departure from typical prey items occurs each spring when red knots feed on the eggs of horseshoe crabs, particularly during the key migration stopover within the Delaware Bay of New Jersey and Delaware. Delaware Bay serves as the principal spring migration staging area for the red knot because of the abundance and availability of horseshoe crab eggs (Clark *et al.* 2009, p. 85; Harrington 2001, pp. 2, 7; Harrington 1996, pp. 76–77; Morrison and Harrington 1992, pp. 76–77). Horseshoe crab eggs are a superabundant source of easily digestible food.

Horseshoe crabs occur along the Atlantic coast from Maine to Florida, along Florida's Gulf coast, and along Mexico's Yucatan Peninsula (Brockman in Shuster *et al.* 2003, p. 33; Botton and Ropes 1987, p. 807). Within this geographic range, horseshoe crabs are most abundant between Virginia and New Jersey (Botton and Ropes 1987, p. 807), with the largest population occurring in Delaware Bay (Gerhart 2007, p. 2; Walls *et al.* 2002, pp. 43–44). Each spring, adult horseshoe crabs migrate from deep bay waters and the Atlantic continental shelf to spawn on intertidal sandy beaches. Beaches within estuaries are preferred spawning areas because they are low energy environments and are protected from the surf (Atlantic States Marine Fisheries Commission (ASMFC) 2004, pp. 7–8). Horseshoe crab spawning generally occurs from March through July, with the peak spawning activity occurring around the evening new and full moon high tides in May and June (Smith and Michels 2006, p. 487; Shuster and Botton 1985, p 368).

Over repeated spawning events, a female horseshoe crab deposits most of her 80,000 eggs in clumps of around 4,000 eggs, at depths of about 4 to 8 in (10 to 20 cm) beneath the surface of the sand, a distance beyond the reach of most shorebirds including red knots. However, wave action and burrowing by subsequent spawning horseshoe crabs move eggs toward the surface. Thus, a high density of spawning horseshoe crabs, such as occurs in Delaware Bay, is needed for the eggs to become available to shorebirds (Smith 2007, p. 287; Pooler *et al.* 2003, p. 698; USFWS 2003, p. 5; Berkson and Shuster 1999, p. 7; Clark in Farrell and Martin 1997, p. 24; Botton *et al.* 1994, p. 614; Shuster and Botton 1985, pp. 367–368).

Horseshoe crabs and surface egg availability are not found in similar densities in other areas on the Atlantic coast, which may explain why shorebirds concentrate in the Delaware Bay (Berkson and Shuster 1999, p. 7). Besides supporting red knots, Delaware Bay supports large numbers of other shorebirds, including semipalmated sandpipers (*Calidris pusilla*), ruddy turnstones (*Arenaria interpres*), and sanderlings (*C. alba*), and the bay ranks among the 10 largest shorebird migration staging sites in the Western Hemisphere (Clark *et al*. 2009, pp. 85, 90). Despite significant shorebird predation on horseshoe crab eggs, such activity probably has little impact on the horseshoe crab population, since the surface eggs consumed by birds typically do not survive anyway due to desiccation (ASMFC 2004, pp. 8–9; Botton *et al*. 1994, pp. 613–614).

Outside of Delaware Bay, horseshoe crab eggs are eaten opportunistically when available in nonbreeding habitats but are not considered a primary food resource for red knots in these areas (Thibault and Levisen 2013, p. 6; Schwarzer and Brush 2011, p. 3; Cohen *et al.* 2010a, pp. 659–660; Cohen *et al*. 2010b, pp. 355, 358; Schneider and Winn 2010, p. 2; González *et al.* 2006, p. 111; Harrington 2001, p. 11; Truitt *et al*. 2001, p. 12). However, in several areas along the Atlantic coast horseshoe crab eggs are a preferred food resource (see table 1) and may be an important component of the spring diet.

Delaware Bay provides the final Atlantic coast stopover for a significant majority (50 to 80 percent) of the red knot population making its way to the arctic breeding grounds each spring (Clark *et al*. 2009, p. 90; Brown *et al.* 2001, p. 10). Red knots stopping in Delaware Bay depend on horseshoe crab eggs to achieve remarkable rates of weight gain. Weight gains recorded on the Delaware Bay are the most rapid of any recorded for all known stopover populations of *Calidris canutus* in the world (Cohen *et al.* 2010b, p. 356; Piersma *et al*. 2005, p. 268) and are among the highest rates observed in the animal kingdom (Atkinson *et al*. 2007, p. 890). Although a single horseshoe crab egg contains a very small amount of energy, eggs are present in such superabundance that birds can eat enough in 2 weeks to nearly double their weights. No single stopover area is more important for the red knot than the Delaware Bay because the nutritive yield of the bay is so high (Harrington 1996, p. 73).

Research indicates this weight gain is important to sustain the birds on their final flights from Delaware Bay to the arctic breeding grounds. Stored fat and protein are used also for initial survival on the breeding grounds (McGowan *et al.* 2011a, p. 9; Piersma *et al.* 2005, p. 270; Baker *et al.* 2004, p. 876), particularly when conditions are adverse upon arrival (e.g., snow cover, lack of insect prey) (Buehler and Piersma 2008, p. 254). Such body stores may also be used by *Calidris canutus* for physical transformations necessary for breeding (Morrison *et al.* 2005, p. 449; Morrison 2006, pp. 610–612). For example, a female red knot radio-tagged in May 1999 in Delaware Bay and recaptured incubating 4 eggs 6 weeks later in Canada had used up 2.1 oz (60 g) of reserves (about one-third of the optimal migration weight), demonstrating the importance of weight gain during migration stopovers (Baker *et al*. 2001, pp. 5–6). Morrison and Hobson (2004, p. 341) found that *C.c. islandica* rely on stored fat to sustain them for a week or more upon arriving at the breeding grounds if there is still snowpack (as cited in Karpanty *et al.* 2011, p. 984). Using data on energetic flight costs by Kvist *et al.* (2001, p. 731), Baker *et al.* (2004, p. 876) calculated that red knots in the Delaware Bay need to achieve a departure mass of at least 6.3 to 7.1 oz (180 to 200 g) to cover the energetic demands of the flight to the breeding grounds and to survive an initial few days of snow cover.

Evidence for Reliance on Horseshoe Crab Eggs

The fact that red knots in Delaware Bay feed almost exclusively on horseshoe crab eggs is strongly supported by several lines of evidence, including physiological analyses, experiments, temporal and spatial correlations, direct field observation of foraging birds, and stable isotope analysis. For example, in 1996 and 1997, Tsipoura and Burger (1999, p. 640) found that horseshoe crab eggs accounted for greater than 60 percent to greater than 80 percent of food items found in the guts of Delaware Bay red knots. Using experimental feeding trays in the field, Gillings *et al.* (2007, p. 507) found that mixed flocks of red knots and other shorebirds consumed 80 percent of surface eggs in 5 minutes.

The timing of the arrival of red knots and other shorebirds in Delaware Bay typically coincides with the annual peak of the horseshoe crab spawning period (Clark *et al.* 2009, p. 91; Clark in Farrell and Martin 1997, p. 23; Morrison and Harrington 1992, p. 77). All indicators show that feeding conditions for red knots were poor in those years when the timing of the horseshoe crab spawn was out of sync with the birds' spring stopover period, such as occurred in 2003, 2005, and 2008 (Atkinson *et al*. 2007, p. 892; Dey *et al.* 2011a, p. 7).

As early as 1982, Wander and Dunne (1982, pp. 61–62) observed that red knots in Delaware Bay eat horseshoe crab eggs almost exclusively and found temporal correlations between shorebird abundance and horseshoe crab spawning activity, as well as spatial correlations of shorebird distribution relative to horseshoe crab spawning activity. Shuster and Botton (1985, p. 366) also found a spatial correlation between horseshoe crab spawning intensity and density of foraging shorebirds. In 1990 and 1991, Botton *et al*. (1994, pp. 605, 614) confirmed a spatial correlation between densities of horseshoe crab eggs and shorebirds in Delaware Bay, and found that shorebirds aggregated near shoreline discontinuities (e.g., salt-marsh creeks, jetties) that concentrated passively drifting eggs. Horseshoe crab eggs were the most abundant food item on these beaches, and there were few other available macroinvertebrates (Botton *et al.* 1994, pp. 605, 614).

In 2003 and 2004, Hernandez (2005, p. 29) found that red knot foraging responses were best explained by models that use horseshoe crab egg parameters only, mainly surface egg density and patchiness. In 2004, Karpanty *et al.* (2006, p. 1706) found that the crab egg biomass was more than 3.5 times larger at points used by red knots than at random points. The number of horseshoe crab eggs was the most important variable in their models predicting habitat usage, although incorporating secondary prey types (*Donax* clams and mussel spat) improved the models' predictions. Karpanty *et al.* (2006, p. 1706) also found red knots shifting from other habitats toward horseshoe crab spawning beaches as spawning activity increased. In 2005, Fraser *et al.* (2010, p. 97) found that red knot feeding activity was concentrated at horseshoe crab nests relative to random beach points. Several studies have found the wrack line is a preferred microhabitat for foraging because horseshoe crab eggs become concentrated there (Nordstrom *et al.* 2006a, p. 438; Karpanty *et al.* 2011, pp. 990, 992).

Using stable isotope diet tracking and captive feeding trials, Haramis *et al.* (2007, entire) demonstrated that red knots in Delaware Bay rely almost entirely on horseshoe crab eggs to support their very high rates of weight gain (as cited in Cohen *et al.* 2010b, p. 356) and established horseshoe crab eggs as a unique and likely critical resource to migratory shorebirds along the east coast of the United States (Haramis *et al*. 2007, p. 373). Haramis *et al*. (2007, p 373) attributed the value of horseshoe crab eggs to red knots and other shorebirds to the eggs' predictable abundance in Delaware Bay, rapid digestion and assimilation, demonstrated importance to rapid fattening during the migration stopover, and selected preference in the diet by the birds themselves. Further, Haramis *et al.* (2007, p. 373) were unable to identify alternative food sources in Delaware Bay with comparable quality (i.e., the combined abundance and digestibility of horseshoe crab eggs).

Possible Differential Reliance on Horseshoe Crab Eggs

Research supports the hypothesis that *Calidris canutus* of several subspecies select migration stopover sites on the basis of prey quality (i.e., flesh-to-shell ratios) to maximize daily net energy income (i.e., gain weight quickly to fuel migration) (van Gils *et al.* 2005a, p. 2611). For shellfish-eating *C. canutus* (including *rufa*), prey qualities at stopover sites average twice those at wintering sites. Consistent with ecological theory, field research shows that poor-quality stopovers are skipped (van Gils *et al.* 2005a, pp. 2610, 2614). Work has begun to extrapolate the "prey quality" concept to horseshoe crab eggs, which differ fundamentally from mollusks because they lack a hard shell.

As indicated above, research has provided strong evidence that a majority of red knots stop at the Delaware Bay during the spring migration, and that these birds are highly reliant on a superabundance of horseshoe crab eggs to gain weight during their stopover period. Newer research is examining possible differences in the use of, and reliance on, the Delaware Bay horseshoe crab resources among knots from different wintering regions. Some researchers have postulated that red knots from southern wintering areas (Argentina and Chile) are more reliant on horseshoe crab eggs than are birds from northern wintering areas (the Southeast) because southern birds cannot digest hard-shelled prey with the reduced digestive organs knots typically exhibit during long migration flights (Niles *et al.* 2008, p. 36; Atkinson *et al.* 2006b, p. 41). It is hypothesized that the northern birds are either able to arrive in the mid-Atlantic earlier than southern birds and re-grow their digestive apparatus to deal with hard-shelled prey, or do not undergo the major physiological changes of gut size reduction that the southern migrants undergo. This postulated difference in the stopover diet is attributed to differences in migration strategy (shorter flights or "hops" for northern birds, versus longer flights or "jumps" for southern birds), as well as absolute distance travelled. According to this hypothesis, knots wintering in Tierra del Fuego are particularly dependent on horseshoe crab eggs for successful fattening and are more vulnerable to a decline in the availability of those eggs than are northern-wintering birds (Niles *et al.* 2008, p. 36; Atkinson *et al.* 2006b, p. 41).

Support for this hypothesis comes from stable isotope work showing that catches of red knots in 2005 from New Jersey's Atlantic coast (where horseshoe crab eggs are not an important part of the diet) contained a much higher proportion of migrants from the Southeast than did catches from Delaware Bay (Atkinson *et al*. 2006a, p. 536), as shown in table 2. In addition, radio tracking suggests at least partial segregation between birds that use Delaware Bay and birds that use the Atlantic coast during the spring stopover (Cohen *et al.* 2010a, p. 660). Further support comes from geolocator results and resightings of marked birds indicating that birds wintering in the Caribbean and the Southeast commonly (but not always) bypass Delaware Bay (Niles *et al*. 2012a, pp. 197–200; Harrington 2005a, p. 1). In addition, red knots feeding on hard shelled prey in Virginia and Massachusetts may have to feed both day and night, while birds stopping in Delaware Bay can apparently meet their energy needs foraging (on superabundant, high-quality horseshoe crab eggs) only by day (Cohen *et al.* 2011, p. 352; Harrington 1996, p. 76).

Another difference between knots that utilize the Atlantic coast of New Jersey (which tend to be from the Southeast) relates to the use of horseshoe crab egg concentration areas when they do forage in Delaware Bay. In Delaware Bay, sites where marked birds foraged in 2004 contained a greater abundance and diversity of prey items than random points, with horseshoe crab eggs making up 91 percent of the prey biomass. In contrast, birds marked on New Jersey's Atlantic coast in 2006 did not appear to be in egg concentration areas when foraging on Delaware Bay. This difference could have arisen if eggs were less patchily distributed in 2006, but could also be related to foraging differences between the two groups (e.g., when foraging in Delaware Bay, Atlantic-coast knots were not selecting for egg concentration areas, or were excluded from the concentration areas through competition) (Cohen *et al*. 2010a, p. 661).

Table 2. Composition of red knots at mid-Atlantic spring stopover areas by wintering area

Location	**Year**	**Primary Prey**	**Percent from Argentina or Chile**	**Percent from Brazil**	**Percent from Florida / Southeast**	**Percent from Unknown Wintering Areas**	**Method and Source**
Delaware Bay	2003	Horseshoe crab eggs	64		30	7	Stable isotope, Atkinson *et al.* 2005, p. 738
Delaware Bay	2004	Horseshoe crab eggs	56.1	22.0	12.6	9.3	Stable isotope, Atkinson *et al.* 2006a, p. 536
Delaware Bay	2005	Horseshoe crab eggs	47.4	20.2	22.5	9.9	Stable isotope, Atkinson *et al*. 2006a, p. 536
Atlantic Coast of New Jersey	2005	Bivalves	< 25		> 50	< 25	Stable isotope, P. Atkinson pers. comm. November 8, 2012
Virginia	2006-2007	Bivalves	43	2		55*	Marked birds, Smith *et al.* 2008, p. 16

*Includes birds banded in the United States, Arctic Canada, and unknown locations. Some may have been banded in U.S. wintering areas, but this cannot be determined from the data presented by Smith *et al.* 2008, p. 16. Percentages may not add to 100 due to rounding.

A final piece of evidence that southern-wintering birds may be particularly dependent on horseshoe crab eggs comes from the temporal correlation of red knot population declines in

Patagonia and Tierra del Fuego with the sharp increase in the horseshoe crab harvest (see Factor E—Food Availability—Horseshoe Crab Harvest). No similar declines are apparent in red knot populations in Brazil or the Southeast, although survey data from these northern wintering areas are associated with lower confidence (see Summary—Population Surveys and Estimates).

However, there is not a strict correlation between wintering area and stopover diet because there is considerable mixing of birds from various wintering regions at the mid-Atlantic stopover sites. Resighting, geolocator, and stable isotope data show that substantial numbers, though not all, of the birds that winter in both Brazil and the Southeast pass through Delaware Bay during spring migration along with the southern-wintering birds (Niles *et al.* 2008, p. 54). For example, Niles *et al.* (2008, p. 9) noted that 15 of 46 red knots marked in Maranhão, Brazil in 2004 and 2005 were resighted in Delaware Bay in spring 2005, showing that at least some of these northern-wintering birds likely feed on horseshoe crab eggs during this stopover. As shown by this information and the data in table 2, it is clear that at least some southern (Argentina and Chile) red knots do eat prey other than horseshoe crab eggs, and some northern (Southeast and Brazil) birds do feed on horseshoe crab eggs, during their final spring stopovers along the mid-Atlantic.

Physiologically, the gizzards of red knots feeding on horseshoe crab eggs are about seven times larger than would be predicted from extrapolations of mollusk prey quality (i.e., with no shells to grind, gizzards during the Delaware Bay stopover should be very small, but they are not) (van Gils *et al.* 2005a, p. 2616; van Gils *et al.* 2003, p. 3369). Further, only Delaware Bay red knots have small stones in their gizzards, presumably to grind the leathery surface of the eggs. These findings suggest that the grinding of horseshoe crab eggs is a fundamentally different process than crushing the outer shells of mollusks (van Gils *et al.* 2005a, p. 2616). So although horseshoe crab eggs are clearly a "super high quality" food (van Gils *et al.* 2005a, p. 2616), there are still energetic costs associated with the digestive processing of the eggs.

The time and energy required to process horseshoe crab eggs, compared to high-quality mollusk prey, have not been investigated. (See Piersma *et al.* 2003a, p. 3365 and van Gils *et al.* 2003, p. 3374 for time and energy costs, respectively, of digesting mollusk prey of various qualities.) Without this information and comparative digestive anatomy of northern versus southern birds upon arrival in the mid-Atlantic, it is not yet possible to conclude if there are significant differences in stopover diet based on wintering region. Although the relative importance of horseshoe crab eggs to birds from various wintering areas is still being refined, it is clear that this food resource is crucial to the rufa red knot.

Historical accounts of red knot distribution, along with re-creations of pre-historic conditions in Delaware Bay, suggest that the exploitation of horseshoe crab eggs by migrating shorebirds may be a relatively recent phenomenon (Cohen *et al.* 2008, pp. 8–27; Fraser 2008, entire; Botton *et al*. in Shuster *et al.* 2003, pp. 24–26) (see Historical Distribution and Abundance, below). If true, this would reflect some flexibility in the red knot's migration strategies. Changes in birds' migratory routes in response to environmental change have been known to occur (Iwamura *et al.* 2013, p. 6). It cannot be presumed, however, that alternative stopover habitats could necessarily fulfill the birds' energy needs as well as Delaware Bay does today. The fact that stopover sites must function in precise sequence both in time and in space means that functional alternatives to current staging areas are unlikely, and suggests little room to adapt to major changes in flyway condition (Iwamura *et al.* 2013, p. 6; Botton *et al*. in Shuster *et al.* 2003, pp. 24–26; Myers *et al.* 1987, p. 22).

Inland

While *Calidris canutus* is somewhat unique among shorebirds as being a specialized molluscivore during much of its annual cycle, consumption of prey aside from mollusks in nonbreeding areas is well-documented, especially during prolonged migratory stopovers. As discussed above, most Atlantic coast knots time their spring migration to coincide with the availability of an abundance of horseshoe crab eggs. Similarly, there is recent evidence that many *C.c. roselaari* staging in the Gulf of California fuel the next leg of their spring migration by feeding on the eggs of Pacific [California] grunion (*Leuresthes tenuis*) (D. Newstead pers. comm. May 8, 2014). A preference for relatively easily digested or soft-bodied prey during these times is likely associated with extreme physiological changes experienced by long-distance migrant shorebirds prior to major migratory movements (see Possible Differential Reliance on Horseshoe Crab Eggs). While it may be that some sites used by knots in the Great Plains (Central Flyway) during their spring passage do not support robust molluscan communities, D. Newstead (pers. comm. May 8, 2014) has suggested that documented stopovers in the Northern Plains may be analogous to the horseshoe crab egg and grunion egg phenomena of the other flyways. The seasonal emergence of insect populations in the Central Flyway, various invertebrates on riverine sandbars, and brine shrimp in the saline lakes of Saskatchewan may be an ecological correlate to horseshoe crab eggs in the Atlantic flyway (D. Newstead pers. comm. May 8, 2014). We find this idea plausible and worthy of further investigation but note it currently lacks supporting documentation.

In addition to numerous insect hatches, there are small freshwater mussels and clams on the upper Garrison reach of the Missouri River in North Dakota, in an area where red knots are "regularly" seen (see figure 6); these bivalves are observed washed up on sandbars. We have no information regarding their abundance or species richness, or if red knots are targeting these mollusks (C. Dovichin pers. comm. May 6, 2014).

HISTORICAL DISTRIBUTION AND ABUNDANCE

The current geographic distribution of the red knot has not changed relative to that recorded in historical writings, with the notable exception of Delaware Bay (discussed further below). Several early writers reported that red knots breed in the Arctic and winter along the U.S. Gulf coast and in South America including Brazil and Tierra del Fuego (Lowery 1974, p. 309; Hellmayr and Conover 1948, p. 167; Bent 1927, p. 143; Forbush 1925, p. 403; Ridgway 1919, p. 233; Forbush 1912, p. 262; Eaton 1910, p. 307; Shriner 1897, p. 94; Mackay 1893, p. 25; Audubon 1844, plate 328). Bent (1927, p. 143) included Jamaica and Barbados as part of the possible wintering range of red knots, and described knots as "rarely" wintering in parts of Louisiana and Florida. Hellmayr and Conover (1948, p. 167) noted the use of the West Indies (Jamaica, Barbados, and Trinidad) during migration. Several writers described the red knot as occurring primarily along the coasts with relatively few sightings inland, but interior migration routes through the central United States were also known (Lowery 1974, p. 309; Hellmayr and Conover 1948, p. 167; Bent 1927, pp. 133, 145; Forbush 1925, pp. 403–404; Ridgway 1919, p. 233; Forbush 1912, pp. 262–263; Eaton 1910, pp. 307–308; Audubon 1844, plate 328). As with the geographic distribution, a number of historical accounts suggest that the timing of the red knot's spring and fall migrations along the Atlantic coast was generally the same in the past as it is today (Myers and Myers 1979, p. 192; Urner and Storer 1949, p. 189; Stone 1937, p. 459; Bent 1927, entire; Forbush 1925, pp. 403–404; Forbush 1912, pp. 262–263: Shriner 1897, p. 94; Dixon 1895 in Barnes and Truitt 1997, p. 114; Mackay 1893, p. 26; Stearns and Coues 1883, p. 229; Roosevelt 1866, p. 151; Giraud 1844, p. 225; Wilson 1829, pp. 140–141).

Although the large-scale geographic distribution of migration stopover habitats does not seem to have changed, some authors have noted regional changes in the patterns of red knot stopover habitat usage along the U.S. Atlantic coast. For example, based on a review of early literature, Cohen *et al.* (2008) and Karpanty *et al.* (2014, p. 2) suggested that red knots had a more extensive spring stopover range a century ago than now, with thousands of birds noted in spring in Massachusetts, New York, New Jersey, and Virginia. Harrington *et al.* (2010a, pp. 188, 190) found changing regional patterns of stopover habitat use in Massachusetts, as well as a shift in the wintering destination of birds stopping in Massachusetts in fall (see Population Surveys and Estimates—Fall Stopover Areas, below).

In Delaware Bay and Tierra del Fuego, the era of modern surveys for the red knot and other shorebird species began in the early 1980s. Systematic red knot surveys of other areas began later, and for many portions of the knot's range, available survey data are patchy (see Population Surveys and Estimates, below). Prior to the 1980s, numerous natural history accounts are available, but provide mainly qualitative or localized population estimates. Nonetheless, a consistent narrative emerges across many historical accounts that red knots were extremely abundant in the early 1800s, decreased sharply starting in the mid-1800s, and may have begun to recover by the mid-1900s. Most writers agree the cause of this historical decline was intensive sport and market hunting (see Factor B).

Audubon (1844, plate 328) observed red knots in spring and fall along the U.S. and Canadian coasts from Texas to the entrance of the Bay of Fundy in southeastern Canada. He found knots in winter in East Florida, concluding that "some of the species do not proceed beyond our southern limits at that season." Red knots were reported as "quite abundant in South Carolina," during both spring and fall migrations (Audubon 1844, plate 328). While in Galveston Bay, Texas, in April 1837, Audubon (1844, plate 328) observed groups of knots arriving daily and proceeding east along the shores of the Gulf of Mexico. Audubon (1844, plate 328) did not observe any interior migration through the United States, but did conclude that red knots migrate overland in Canada for the last part of their northbound migration "when they have reached the entrance to the Bay of Fundy." Regarding the breeding range, "it is certain that they reach a very high latitude, and that some stop to breed about Hudson's Bay" (Audubon 1844, plate 328). Providing later documentation of the migration route through Canada, Hope and Short (1944, p. 574) reported "huge flocks" of red knots at a site called Big Piskwanish on James Bay, Canada in July 1942; some of the flocks contained from 200 to 500 birds.

Herbert (1853, p. 160) reported the red knot was "very abundant" among the small islets of Long Island, New York. Roosevelt (1866, p. 150) described red knots as "far from plentiful," but locally "exceedingly numerous" at Egg Harbor, New Jersey (Roosevelt 1866, p. 97). Coues (1868, p. 293) described the red knot as "abundant" along the New England coast in spring and autumn migration. Hallock (1877, p. 168) described the red knot as "an abundant species found along the shores of Long Island and New Jersey in numbers." Stearns and Coues (1883, p. 229) reported the red knot "is a common spring and autumn migrant through New England, where it is found chiefly in small flocks along the seashore."

Mackay (1893, p. 26) described an earlier era when thousands of knots occurred on Cape Cod for about a week during spring migration. Prior to 1850, red knots were more numerous in parts of Cape Cod "than in all the rest of New England combined, and being very gregarious they would collect in those places in exceedingly large numbers, estimates of which were useless…immense numbers of these birds could be seen, as they rose up in clouds…" (Mackay 1893, p. 29). Mackay (1893, pp. 25, 30) described a considerable population decline caused by intensive hunting, "It is not my intention to convey the impression that the Knots are nearly exterminated, but they are much reduced in numbers, and are in great danger of extinction, and comparatively few can now be seen in Massachusetts, where formerly there were twenty to twenty-five thousand a year, which I consider a reasonable estimate of its former abundance." Mackay (1893, p. 31) noted that red knots were still found "in greater or less numbers along the Atlantic coast south of Chesapeake Bay." From Virginia as late as 1895, Dixon (1895 in Barnes and Truitt 1997, p. 114) reported flocks mid-May of 10,000 "red-breasted snipe," which likely refers to red knots based on the feeding habits and timing of occurrence described in that account.

Based on Mackay (1893) and other sources, Harrington *et al.* (2010a, pp. 189–190) reported a 75 percent decrease on Cape Cod from 1850 to 1890, followed by another 25 percent decrease from 1890 to 1904. Echoing Mackay (1893), Shriner (1897, p. 94) reported, "This bird was formerly very plentiful in migrations in New Jersey, but it has been killed off to a great extent…" Eaton (1910, p. 307) reported that the red knot breeds in Arctic regions and migrates along the coast and inland waters, wintering from the Gulf coast to South America. "It is one of the best known beach birds on the south coast of Long Island, but like all the shorebirds is much less common than formerly" (Eaton 1910, pp. 307–308).

Forbush (1912, p. 263) provided an early account of red knots migrating through the interior United States, reporting that the red knot "…migrates principally along the Atlantic coast, both spring and fall, but in the spring, numbers of the species arrive in Texas, Louisiana and other southern States, going north through the Mississippi valley region." Forbush (1912, pp. 263–266) described red knots formerly reaching the shores of New England "in immense numbers," but went on to cite Mackay (1893) and numerous other sources in documenting a substantial population decline. Consistent with earlier reports that knots were still common farther south (Mackay 1893, p. 31; Dixon 1895 in Barnes and Truitt 1997, p. 114), Forbush (1912) concluded that the decline occurred earlier in the northern part of the U.S. range than in the Southeast. "The Knot had decreased considerably near Boston before the middle of the last century," but "[up] to about 1900 they were still very plentiful in the Carolinas" (Forbush 1912, pp. 263–264). Despite some recent localized increases, Forbush (1912, p. 264) determined that the "numbers of this bird have decreased tremendously all along the Atlantic coast within the last seventy-five years." Although numerous sources cited by Forbush (1912, pp. 263–266) agreed that over-hunting was the chief cause of the decline (see Factor B), one source attributed a local decrease to a "lack of feed" (Forbush 1912, p. 265), perhaps an early indication of the red knot's sensitivity to reduced prey availability (see Factor E). Forbush (1912, p. 266) concluded, "The decrease is probably due, however, to shooting both spring and fall all along our coasts, and possibly to some extent in South America. Evidently we are doing more than driving the Red-breast off our coast, and while the utter extinction of such a cosmopolitan bird is not imminent, its extirpation from the Atlantic coast of North America is one of the possibilities of the near future."

In a later work, Forbush (1925, pp. 403–404) began to note signs of recovery. "In the days of our grandfathers the Gray-backs or Wahquoits, as they were called, swarmed along the coasts of Cape Cod by the thousand. . . . they collected in immense numbers and rose in "clouds" before the sportsman's gun. As the nineteenth century closed they were becoming rare all along the coast, but now under protective laws their numbers are beginning to increase. . . . The Knot is highly regarded as a game bird and was formerly sold in large numbers in the markets of the Atlantic coast."

By 1929, Bent (1927, p. 132) continued to note signs of red knot population recovery, "Excessive shooting, both in spring and fall reduced this species to a pitiful remnant of its former numbers; but spring shooting was stopped before it was too late and afterwards this bird was wisely taken off the list of game birds; it has increased slowly since then, but is far from abundant now." Bent (1927, p. 132) noted flocks of 1,500 to 3,000 in Long Island in 1910, and described the red knot as "very common" in South Carolina in May. In Massachusetts, a slow increase of red knots began with regulatory protections beginning about 1920 (Harrington *et al.* 2010a, p. 188). Late summer counts (i.e., of southbound migrants) on Cape Cod were 300 in 1929; 1,000 in 1937; 1,500 in 1941; followed by a decrease for eight years and then 2,400 in 1951; and 5,000 in 1954 (Harrington *et al.* 2010a, p. 188).

In spring and fall surveys of New Jersey's Atlantic coast from 1935 to 1938, Urner and Storer (1949, pp. 178–183) noted peak annual spring counts of 500 to 3,000 knots, and peak annual fall counts of 725 to 1,600 knots. These authors found the red knot to be common or locally abundant in both spring and fall, with a large increase in numbers over previous years (Urner and Storer 1949, pp. 185–188). Urner and Storer (1949, pp. 192–193) noted increases in several medium- and large-bodied shorebird species, continuing a trend that began in 1931, and concluded, "…it is obvious that the species which have increased the most are, for the most part, those which in the past bore the brunt of the hunting pressure." Based on his bird studies of Cape May, New Jersey, Stone (1937, p. 465) was "inclined to think that, while its numbers were sadly depleted, the Knot never reached such a low ebb on our coast as some have supposed, and since the abolishing of the shooting of shore birds it has steadily increased in abundance." An indication of continued recovery in the mid-20th century comes from Harrington *et al.* (2010a, pp. 188, 190), who found that overall red knot numbers on the Massachusetts coast increased from the late 1940s to the early 1970s, especially on the mainland (western Cape Cod Bay), with a smaller increase on outer Cape Cod. It is unclear whether the red knot population fully recovered its historical numbers (Harrington 2001, p. 22) following the period of unregulated hunting.

Perhaps an early indication of the red knot's sensitivity to human disturbance, Stone (1937, p. 465) noted that knots avoided those beaches "most frequented by summer visitors." Fraser (2008, p. 3) noted an even earlier report of the effects of disturbance (see Factor E) and development (see Factor A) on shorebirds in the mid-Atlantic, from the August 9, 1890, New York Times: "Once in a while a nice fat snipe or bay bird is knocked over at Sandy Hook and farther down the Jersey coast. They do not come in there in large flights as they did years ago, because of so much building of Summer homes along the beach. Snipe like to keep at a good distance from human beings and are fond of roaming over wild stretches of sand. There is yet quite a barren stretch of sand dunes that keeps the ocean from pouring into the Great South Bay, and this section is the only good snipe ground near the city." According to Roosevelt (1866, p. 93), red knots were among the shorebird species commonly referred to as "bay-snipe," so may have been among the species referred to in the 1890 observations.

Delaware Bay

Delaware Bay was not recognized as a major shorebird stopover area until the early 1980s, despite detailed shorebird studies (e.g., Urner and Storer 1949; Stone 1937) in the South Jersey region (Clark *et al.* 2009, p. 85; Botton *et al.* in Shuster *et al.* 2003, p. 6; Clark in Farrell and Martin 1997, p. 23; Clark *et al.* 1993, p. 695). There were some early anecdotal reports involving horseshoe crabs, as summarized by Botton *et al.* (in Shuster *et al.* 2003, p. 6). Wilson (1829, pp. 145–146) noted that ruddy turnstones in the bay fed "almost wholly on the eggs, or spawn, of the great King Crab," but no similar accounts were made of red knots (Wilson 1829, pp. 140–144). Forbush (1912, p. 267) noted that red knots "are fond of the spawn of the horsefoot crab, which, often in company with the Turnstone, they dig out of the sand…" Stone (1937, p. 400) observed ruddy turnstones and black-bellied plovers (*Pluvialis squatarola*) regularly feeding on dead horseshoe crabs in Delaware Bay. Stone (1937, p. 400) also mentions flights of ruddy turnstones across the Cape May Peninsula in the spring, as happens today when they go to roost at night along the Atlantic coastal marshes (Botton *et al.* in Shuster *et al.* 2003, p. 6). Interestingly, no mention of horseshoe crab eggs as food is found in Stone's (1937) accounts of any shorebird in the Cape May area, or in the decade-long study by Urner and Storer (1949) (Botton *et al.* in Shuster *et al.* 2003, p. 6). During his early studies of horseshoe crabs in 1951, C. Shuster observed many shorebirds feeding along Delaware Bay beaches, including red knots. However, another 30 years elapsed before scientists began to study the shorebird-horseshoe crab relationship in detail, and documented the very large numbers of shorebirds using the bay as a stopover (Botton *et al.* in Shuster *et al.* 2003, pp. 6–7). Lack of earlier scientific documentation cannot be attributed to remoteness. Delaware Bay is located within a few hours' drive of millions of people, and university marine laboratories were established many years ago on both shores of the bay (Botton *et al.* in Shuster *et al.* 2003, p. 6).

It is unclear if the large magnitude of the shorebird-horseshoe crab phenomenon was simply missed by science until 1981, or if the distribution of the red knot and other shorebird species changed over the period of the historical record. For much of the 20th century, this phenomenon in Delaware Bay may have been much reduced (relative to 1980s levels), and therefore easier to miss, due to the occurrence of low points in the abundance of both shorebirds (caused by hunting, see Factor B) and horseshoe crabs (caused by intensive harvest, see Factor E) (Botton *et al.* in Shuster *et al.* 2003, p. 25; Clark in Farrell and Martin 1997, p. 23). Alternatively, it may be that the red knot did not make extensive use of Delaware Bay prior to its population decline a century ago. Under this scenario, red knots came to rely on Delaware Bay because their populations were recovering at the same time that Atlantic-side stopover habitats in the region were becoming developed and the shorelines stabilized (Cohen *et al.* 2008, pp. 23–27) (see Factor A). This second idea is supported by the fact that the spring stopover seems to have been historically more spread out along the mid-Atlantic, rather than highly concentrated as it is now (e.g., in Delaware Bay and Virginia). We have no means to determine how long shorebirds have been reliant on horseshoe crab eggs in Delaware Bay (Botton *et al*. in Shuster *et al.* 2003, pp. 24–26) prior to the early 1980s.

Summary—Historical Distribution and Abundance

The current geographic distribution of the red knot has not changed relative to that recorded in historical writings, with the notable exception of Delaware Bay. Likewise, a number of historical accounts suggest that the timing of the red knot's spring and fall migrations along the U.S. Atlantic coast was generally the same in the past as it is today. Although we lack quantitative data, a sharp red knot population decline from the late 1800s to the early 1900s is noted across numerous historical accounts from the U.S. Atlantic coast, driven by unregulated hunting, and followed by signs of recovery once hunting ceased. Outside of the U.S. Atlantic coast, we have no information regarding the red knot's historical abundance or population trends.

Delaware Bay was not recognized as a major shorebird stopover area until the early 1980s, despite detailed shorebird studies in the South Jersey region. It is unclear if the large magnitude of the shorebird-horseshoe crab phenomenon was missed by science until 1981, or if the distribution of the red knot and other shorebird species changed over the period of the historical record. The middle part of 20th century coincided with recovery of shorebird populations following the regulation of hunting (Urner and Storer 1949, pp. 192–193; Bent 1927, p. 132), a low point in horseshoe crab abundance following a period of intensive harvest (ASMFC 2009a, p. 1), and the large-scale development and stabilization of Atlantic coast beaches in the mid-Atlantic region (Nordstrom and Mauriello 2001, pp. 20–21; Nordstrom 2000, pp, 7–11). Any or all of these factors may have influenced the red knot's use of, and reliance on, Delaware Bay as its primary Atlantic stopover site in spring. (See Migration and Wintering Food—Horseshoe Crab Eggs—Possible Differential Reliance on Horseshoe Crab Eggs.)

POPULATION SURVEYS AND ESTIMATES

Although Harrington (2001, p. 22), Morrison *et al.* (2001a, pp. 33–34), and Morrison *et al.* (2006, pp. 71, 76–77) produced rangewide population estimates for the red knot, we choose not to evaluate these estimates for several reasons: (1) they are not current; (2) scientific understanding of the geographic distribution of *Calidris canutus rufa* versus *C.c. roselaari* has evolved since these references were published, confounding the estimates; and (3) we conclude that the methods and coverage reflected in the various available survey data sets vary too greatly to sum them. Despite these limitations in producing a rangewide population estimate, we do note that Morrison *et al.* (2006, pp. 71, 76–77), as well as Andres (2009) and National Park Service (NPS) (2013), all concluded red knot numbers declined, probably sharply, in recent decades. This is consistent with the conclusions we draw from the available (regional) data sets, which are presented below.

As no current, reliable, rangewide population estimate is available, we have instead evaluated the best available data, which consists of survey data for specific regions. Localized and regional red knot surveys have been conducted across the subspecies' range with widely differing levels of geographic, temporal, and methodological consistency. Thus, we limit our conclusions to trends within each regional data set, although we do note a temporal correlation between declines at Tierra del Fuego and Delaware Bay. Available survey data are presented in detail below, along with the sources and methodologies for each. Some general characterizations of the available data include:

- No population information exists for the breeding range because, in breeding habitats, red knots are thinly distributed across a huge and remote area of the Arctic. Despite some localized survey efforts, (e.g., Bart and Johnston 2012, pp. 141-155; Niles *et al.* 2008, p. 62), there are no regional or comprehensive estimates of breeding abundance, density, or productivity (Niles *et al.* 2008, pp. 52–53).

- Few regular surveys are conducted in fall because southbound red knots tend to be less concentrated than during winter or spring.

- Some survey data are available for most wintering and spring stopover areas. For some areas, long-term data sets have been compiled using consistent survey methodology.

- Because there can be considerable annual fluctuations in red knot counts, longer-term trends are more meaningful. At several key sites, the best available data show that numbers of red knots declined and remain low relative to counts from the 1980s, although the rate of decline appears to have leveled off since the late 2000s; specific data are presented and analyzed below.

- Inferring long-term population trends from various national or regional datasets derived from volunteer shorebird surveys and other sources, NPS (2013), Andres (2009, p. 6), and Morrison *et al.* (2006, pp. 71, 76, 77) also concluded that red knot numbers declined, probably sharply, in recent decades.

Wintering Areas

Counts in wintering areas are particularly useful in estimating red knot populations and trends because the birds generally remain within a given wintering area for a longer period of time compared to the areas used during migration. This eliminates errors associated with turnover or double-counting that can occur during migration counts.

Argentina and Chile

Aerial surveys of Tierra del Fuego (Chile and Argentina) and the adjacent Patagonian coast to the north (Argentina) (see figures 3 and 4) have been conducted since 2000, and previously in the early 1980s, by the same observers using consistent methodology (Morrison *et al.* 2004, p. 62). This is the best available long-term data set for a wintering area. However, as these are not the only red knot wintering areas, the survey results are best interpreted as one indicator of population trends rather than estimates of the total population.

Table 3. Aerial counts of red knots in Chile and Argentina, winters 1982 to 2013 (Dey *et al.* 2014, p. 2; G. Morrison pers. comm. August 31, 2012; Dey *et al.* 2011a, p. 2; Morrison *et al*. 2004, p. 65; Morrison and Ross 1989, Vol. 2, pp. 226, 252).

Year	Tierra del Fuego	Patagonia	Total
1982		14,314	
1985	53,232		
2000	51,255		
2001	29,745*		
2002	27,242	2,029	29,271
2003	29,915	560	30,475
2004	30,778	880	31,658
2005	17,653		
2006	17,211		
2007	17,360		
2008	14,800		
2009	17,780		
2010	16,260		
2011	9,850		
2012	14,200	574	14,774
2013	10,105		10,105

*Only the single largest wintering area (Bahía Lomas) and one small adjacent site were surveyed on Tierra del Fuego in 2001.

As shown in table 3, counts have been markedly lower in recent years. Comparing the average counts for Tierra del Fuego from 1985 and 2000 (52,244) with 2011 to 2013 (11,385), the recent counts are more than 75 percent lower than the earlier counts. An independent population estimate, based on modeling using resighting data from Río Grande, supports the observation that declines did not begin until after 2000. González *et al.* (2004, p. 361, as cited in Niles *et al.* 2008, pp. 65–67) estimated the 1995 wintering population in Argentina and Chile at 74,193 (95 percent confidence interval 50,000 to 110,000). This same model produced population estimates that were within 5 to 15 percent of the aerial counts from 2001 to 2003, giving confidence in the model results.

As shown in table 3, declines were even sharper (about 96 percent) along the roughly 1,000 mi (1,600 km) of Patagonian coast than in the core area on Tierra del Fuego. The population thus appears to have contracted to the core sites, leaving few birds at the "peripheral" Patagonian sites (COSEWIC 2007, p. 11). Reflecting the larger downward trend in Patagonia, local winter counts at Península Valdés went from 8,000 red knots in 1994 to a low of 650 red knots in 2008, with the most recent (2010) at 825 knots (WHSRN 2012).

Northern South America

Counts of wintering red knots along the north coast of South America have been sporadic and have varied in geographic coverage. Morrison and Ross (1989, Vol. 2, p. 183) conducted aerial surveys of the entire South American coast in the 1980s. In northern Brazil, more than 8,000 red knots were found across 3 out of 4 survey segments (figure 5; table 4): North, North-

Central, and Northeast. No red knots were observed in the Amazon survey segment of Brazil, which is between North and North-Central (Morrison and Ross 1989, Vol. 2, p. 183). Using the same surveyor team and methods as the 1986 survey, the North-Central segment of Brazil was again surveyed by air in 2011 (D. Mizrahi pers. comm. November 17, 2012; Morrison *et al.* 2012, p. 126). As shown in table 4, the 2011 total was 3,660 red knots (D. Mizrahi pers. comm. November 17, 2012), which suggested a decline. Redistribution of birds to the west was an unlikely explanation for the lower numbers in 2011, based on surveys of Guyana, Suriname, and French Guiana around the same time (discussed below) (Morrison *et al.* 2012, p. 126).

However, 2013 results raise questions about the red knot's winter abundance in northern Brazil. Again using the same team and methods, a team of Guy Morrison, Ken Ross, Paulo de Tarso Zuquim Antas, and NJAS surveyed the North, Amazon, and North-Central segments in Brazil in 2013. No red knots were observed in the North or Amazon segments a substantially higher count of 15,485 knots were recorded in the North-Central segment in 2013 (G. Morrison pers. comm. August 12, 2014; Dey *et al.* 2014, p. 1). The larger number found in 2013 is likely because the team was able to get favorable tidal conditions throughout the survey period, and this is probably the team's best aerial survey estimate to date (G. Morrison pers. comm. July 30, 2014).

Covering about 30 percent (by linear miles of coastline) of the North-Central Brazil survey segment, Baker *et al.* (2005, p. 12) counted 7,575 knots in western Maranhão, from Baía do Mutuoca to São Luís (figure 5), during an aerial survey in February 2005. In a repeat of this survey in December 2006 (winter of 2007), only 3,000 knots were counted (Niles *et al.* 2008, p. 134) (table 4). The shores of Maranhão are complex and highly fragmented making accurate counting more difficult. To allow for this, aerial coverage was more extensive and included not only the ocean shore but also a variety of back bays and channels (Niles *et al.* 2008, p. 134).

In December 2007 (winter of 2008), ground surveys were conducted at 2 sites in the Brazilian State of Ceará, within and immediately adjacent to the Northeast Brazil survey segment (where only 15 red knots had been counted in 1983). Only small numbers of knots (average peak of 8 ± 8.5) were observed at Ilha Grande, but an average peak count of 481 ± 31 red knots was recorded at Cajuais Bank (Carlos *et al.* 2010, pp. 10–11, 13). Lower numbers (up to 80) of red knots have been observed in winter at 4 other sites in Ceará (Serrano 2007, p. 16).

Due to the difficulty of surveying (Niles *et al.* 2008, p. 134), variability of tidal conditions (G. Morrison pers. comm. July 30, 2014), and availability of only a few data points, we do not infer any trends in the counts from Northern Brazil.

Figure 5. Survey areas along the north coast of South America

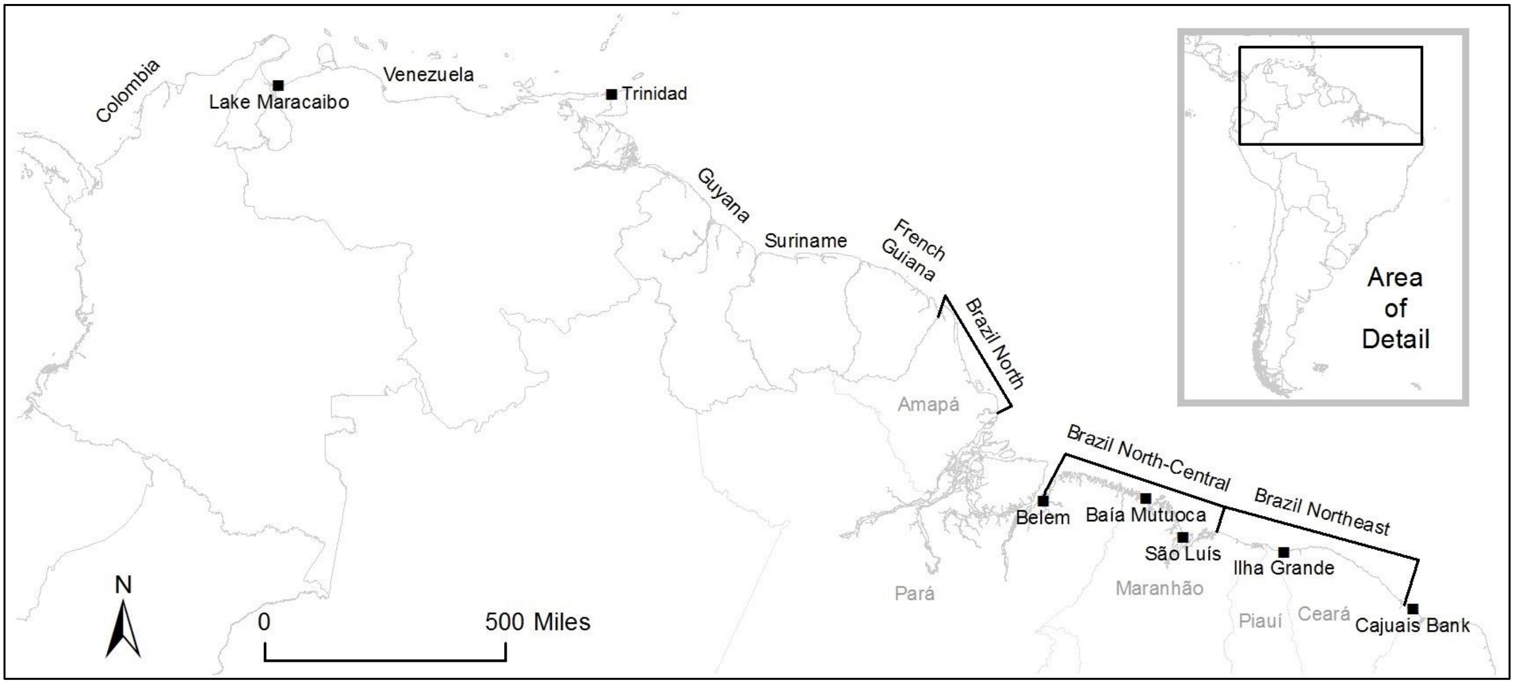


Table 4. Aerial and ground counts of red knots on the northern coast of Brazil (from west to east), winters 1986 to 2013 (Dey *et al.* 2014, p. 1; D. Mizrahi pers. comm. November 17, 2012; Niles *et al.* 2008, p. 134; Baker *et al.* 2005, p. 12; Morrison and Ross 1989, Vol. 2, p. 183)

Winter	**North Survey Segment** (State of Amapá)	**North-Central Survey Segment** (from Belém, State of Pará to Baía de Sanadi, State of Maranhão)	**From Baía do Mutuoca to São Luís** (State of Maranhão, within North-Central Survey Segment)	**Northeast Survey Segment** (from Baía de Sanadi, State of Maranhão to Aracati, State of Ceará)	**Cajuais Bank**** (in the State of Ceará, adjacent to Northeast Survey Segment)
1982	120				
1983				15	
1986		8,191			
2005			7,575		
2007*			3,000		
2008*					481 ± 31
2011		3,660			
2013	0	15,485			

*December of the previous year.

**Ground survey; all others were aerial. Peak monthly average of repeated counts from December to February; all others were one-time counts.

Morrison and Ross (1989, Vol. 2, pp. 145, 149) also documented 520 *Calidris canutus* in western Venezuela in 1982, in the saline lagoon areas at the mouth of the Lake Maracaibo channel. Due to lack of access, Morrison and Ross (1989, Vol. 2, p. 145) did not survey the long beaches running from the mouth of Lake Maracaibo westwards along the Gulf of Venezuela, but large numbers of *C. canutus* (1,000 or more) had previously been reported from this location (Morrison and Ross 1989, Vol. 2, p. 145). Ruiz-Guerra (2011, p. 194) documented 20 *C. canutus* at Musichi (Department of La Guajira) on the Caribbean coast of Colombia near Venezuela in January 2008. It is not known if the birds observed around the Colombia-Venezuela border were all of the *rufa* subspecies, but recent geolocator results suggest at least some of the winter birds in this area are *C. canutus rufa* (Niles *et al.* 2012a, p. 200).

During the 1980s surveys, no red knots were observed between western Venezuela and the west end of Brazil (the North segment), with no knots recorded in eastern Venezuela, Trinidad, Guyana, Suriname, or French Guiana (Morrison and Ross 1989, Vol. 1, p. 41). With the same survey team and methods from the 1980s, aerial shorebird surveys were repeated in Guyana (January 2010), Suriname (December 2008, January 2010, and January 2011), and French Guiana (December 2008 and January 2010) (Morrison *et al.* 2012, p. 121). No red knots were detected in 2011, and a negligible number in December 2008 (i.e., winter 2009) and in 2010 (D. Mizrahi pers. comm. November 17, 2012 and October 16, 2011). Red knots are mostly migrants in French Guiana, although there is a small wintering population between October and March, numbering a few tens of birds, mostly at Kourou (CSRPN 2013). In the 1990s, a population of 40 to 50 knots used to winter in the mudflats along the old Cayenne harbor, French Guiana; however, wintering red knots were no longer using the area by 2005 (CSRPN 2013). Small, isolated groups of wintering red knots may extend along most of the northern coast of South America (L. Niles pers. comm. January 8, 2013).

Northwest Gulf of Mexico

Except for localized areas, there have been no long-term systematic surveys of red knots in Texas or Louisiana, and no information is available about the number of knots that winter in northeastern Mexico. From survey work in the 1970s, Morrison and Harrington (1992, p. 77) reported peak winter counts of 120 red knots in Louisiana and 1,440 in Texas, although numbers in Texas between December and February were typically in the range of 100 to 300 birds. Records compiled by Skagen *et al.* (1999) give peak counts of 2,838 and 2,500 red knots along the coasts of Texas and Louisiana, respectively, between January and June over the period 1980 to 1996, but these figures could include spring migrants. Morrison *et al.* (2006, p. 76) estimated only about 300 red knots wintering along the Texas coast, based on surveys in January 2003 (Niles *et al.* 2008, p. 19). Higher counts of roughly 700 to 2,500 knots have recently been made on Padre Island, Texas, during October, which could include wintering birds (Newstead *et al.* 2013, p. 54; Niles *et al*. 2009, p. 1). There are no current estimates for the size of the Northwest Gulf of Mexico wintering group as a whole (Mexico to Louisiana). The best available current estimates for portions of this wintering region are about 2,000 in Texas (Niles 2012a), or about 3,000 in Texas and Louisiana, with about half in each State and movement between them (C. Hunter pers. comm. September 20, 2012).

Christmas Bird Count data suggest that wintering red knots have declined along the northern Gulf coast from Texas to Florida by 2.3 percent per year (95 percent confidence interval: 0.18 to 4.59 percent per year) (Niven and Butcher 2011, p. 18). However, these authors did not report on trends specific to the western Gulf. Particularly in Louisiana and Texas, the red knot's coastal distribution barely overlaps Christmas Bird Count coverage, thus coastwide trends are strongly biased by counts from Florida (Johnson 2013, p. 4), as discussed below under Southeast United States and Caribbean.

Purrington (2012, p. 65) lists the red knot as an uncommon to scarce (i.e., present in low numbers) winter visitor in Louisiana. (Purrington *et al.* (2008, p. 2) defined "uncommon" as 0 to 4 birds per day.) Winter occurrences in Louisiana are erratic, and intense survey coverage may be needed to detect knots. Nonetheless they are regarded as somewhat regular in winter. Their gregarious habits (they are frequently found in flocks of 15 to 100 individuals) contribute to their "spotty" distribution in Louisiana (Purrington 2012, p. 65). The birds seem to disappear in the coldest winters, perhaps moving down the Texas coast or even farther south. Most wintering birds are recorded from the Grand Terre/Grand Isle region west to Raccoon Island, Terrebonne Parish, but presumably some may winter offshore on the seldom-visited Chandeleur Island chain. A high count of 70 knots was recorded on Timbalier Island in February 2011, with more typical winter counts of 1 to 10 birds. Wintering birds appear to be largely absent from the southwestern Louisiana beaches where they are regular during spring and fall migration. The Christmas Bird Count at Grand Isle recorded red knots in 7 of the 10 years from 2004 to 2013, ranging from 0 to 92 birds, and averaging 13.6 birds per year (Russell 2014, p. 3).

Although available data specific to Louisiana are very limited, they do suggest a decline. Louisiana Christmas Bird Counts within coastal areas show that red knot counts have declined from 1.6 birds per party-hour (i.e., an hour of survey effort by a group, or party, of surveyors) in the 1980s, to 0.2 birds per party-hour in the 1990s, to 0.05 birds per party-hour in the 2000s (Johnson 2013, p. 1). Red knots were perhaps never terribly common in Louisiana as passage migrants or during wintering, compared to estimates from Texas and Florida (Johnson 2013, p. 12). However, available historical data suggest counts greater than 100 were fairly regular in the 1980s. Today, red knots are rarely seen in numbers greater than 50 at a time in Louisiana. Although red knots were perhaps always more common along the barrier island systems in southeastern Louisiana than along the chenier plain coastline of southwestern Louisiana, they are now extremely unlikely to be seen in southwestern Louisiana, except perhaps a few scattered birds during migration. Although available data are quite limited, the cumulative evidence suggests substantial declines have occurred in Louisiana since the 1980s. However, with such limited data available, it is not possible to determine how much of the decrease may reflect a shift in distribution versus an overall regional population decline (Johnson 2013, p. 12).

Foster *et al.* (2009, pp. 1081, 1084) found a mean daily abundance of 61.8 red knots on Mustang Island, Texas, based on surveys every other day from 1979 to 2007. Similar winter counts (26 to 120 red knots) were reported by Dey *et al.* (2011a, p. 2) for Mustang Island from 2005 to 2011. From 1979 to 2007, mean abundance of red knots on Mustang Island decreased 54 percent. This may have been a localized response to increasing human disturbance, coastal development, and changing beach management practices (Newstead *et al.* 2013, p. 54; Foster *et al.* 2009, p. 1079) (i.e., it is possible these birds shifted elsewhere in the region). However, it is not possible to confidently attribute any such factors as the cause of this sharp decline on Mustang Island (D. Newstead pers. comm. May 8, 2014).

Similarly, reports from Bolivar Flats Shorebird Sanctuary on the upper Texas coast also suggest that the red knot is far less abundant than it was in previous decades, to the point that it is now only present occasionally and in low numbers. As a shorebird sanctuary, the decline at this site cannot be attributed to an increase in direct anthropogenic factors that may potentially explain the declines on Mustang Island (D. Newstead pers. comm. May 8, 2014).

Southeast United States and Caribbean

Harrington *et al.* (1988, p. 440) reported that the mean count of birds wintering in Florida was 6,300 birds (± 3,400, one standard deviation) based on 4 aerial surveys conducted from October to January in 1980 to 1982. These surveys covered the Florida Gulf coast from Dunedin to Sanibel-Captiva, sometimes going as far south as Cape Sable (B. Harrington pers. comm. November 12, 2012). Based on these surveys and other work, the Southeast wintering group was estimated at roughly 10,000 birds in the 1970s and 1980s (Harrington 2005a, p. 1).

Based on resightings of birds banded in South Carolina and Georgia from 1999 to 2002, the Southeast wintering population was estimated at 11,700 ± 1,000 (standard error) red knots. Although there appears to have been a gradual shift by some of the southeastern knots from the Florida Gulf coast to the Atlantic coasts of Georgia and South Carolina, population estimates for the Southeast region in the 2000s were at about the same level as during the 1980s (Harrington 2005a, pp. 1, 15).

Based on modeling using resightings of marked birds staging in Georgia in fall, combined with stable isotope data to determine the wintering areas of these fall migrants, the northern-wintering group was recently estimated at 20,364 birds (95 percent confidence interval: 18,014 to 22,990) (GDNR 2013). This is considered a minimum estimate, because it only includes the (unknown) proportion of northern-wintering birds that stopped at Georgia's Altamaha River during fall migration in 2011 (GDNR 2013). (See Population Surveys and Estimates—Fall Stopover Areas.) This estimate is consistent with an earlier, unpublished study that also estimated the northern-wintering population at about 20,000 knots (B. Harrington pers. comm. November 12, 2012). However, these higher estimates have not been corroborated by a targeted winter field survey covering the entire Southeast region. Further, we do not have data from these studies to clarify the geographic extent of this "northern" group (i.e., if, in addition to Southeast-wintering birds, these estimates also include birds wintering in the Caribbean or even Texas or northern Brazil).

As discussed above under Wintering Areas—Northwest Gulf of Mexico, Christmas Bird Count data suggest that wintering red knots have declined along the northern Gulf coast from Texas to Florida by 2.3 percent per year (95 percent confidence interval: 0.18 to 4.59 percent per year) (Niven and Butcher 2011, p. 18). However, these authors did not report on trends specific to the eastern Gulf that is part of the Southeastern wintering region. Particularly in Louisiana and Texas, the red knot's coastal distribution barely overlaps Christmas Bird Count coverage, thus coastwide trends are strongly biased by counts from Florida (Johnson 2013, p. 4). Therefore, we conclude that at least some of this decline was likely caused by the shifting of some southeastern knots from Florida's Gulf coast to the Atlantic coasts of Georgia and South Carolina, as discussed above.

Red knots in the Southeast were counted incidentally to the International Piping Plover Census in 2006 and 2011, a ground survey from Louisiana to Virginia (table 5). Because knots were not the focus of the piping plover survey, the level of effort varied and red knot numbers may be potentially underestimated. Table 5 also shows results from Georgia of regular Statewide ground counts conducted during a narrow window in late January or early February (Schneider and Winn 2010, p. 3).

Table 5. Red knot counts in the in the Southeast United States, winters 1997 to 2011 (L. Patrick pers. comm. August 31, 2012; M. Bimbi pers. comm. November 1, 2012; Niles *et al.* 2006, p. 89)

	Virginia	North Carolina	South Carolina	Georgia	Florida	Louisiana-Alabama-Mississippi (estimates)	Total
1997				411			
1999				175			
2001				4,689			
2002				1,080			
2003				1,247			
2004				586			
2005				3,363			
2006	26	455	583	485	3,020	500	5,069
2007				1,083			
2008				208			
2009				1,748			
2010				577			
2011		157	1,201	1,160	1,046	250 to 375	3,814 to 3,939

<u>Florida</u>

Extensive data for Florida are available from the International Shorebird Survey (ISS) and other sources. However, geographic coverage has been inconsistent, ranging from 1 to 29 sites per year from 1974 to 2004. Statewide annual totals ranged from 5 knots (1 site in 1976) to 7,764 knots (7 sites in 1979). The greatest geographic coverage occurred in 1993 (4,265 knots at 25 sites) and 1994 (5,018 knots at 29 sites) (Niles *et al.* 2008, pp. 177–180).

Sprandel *et al.* (1997) identified the top 60 sites for wintering shorebirds in Florida and surveyed these areas in 1994. Red knots were found at 27 sites, mainly on the central Gulf coast. Adding the average number of birds counted at each site, these authors estimated a Statewide total of 1,452 red knots, with the following regional estimates: 124 in the Panhandle (3 sites); 0 in the Big Bend (0 sites); 1,092 in the Southwest (18 sites); 183 in the Everglades (4 sites); and 53 in the Northeast (2 sites) (Sprandel *et al.* 1997, p. 33). During frequent surveys of 9 sites along about 55 mi (89 km) of the central Florida Panhandle, Smith (2010, p. 48) found a mean of about 84 wintering red knots in the winter of 2007. Smith (2010, p. 45) covered roughly 25 percent of the Panhandle region as delineated by Sprandel *et al.* (1997, p. 6), with the survey sites clustered on the eastern end of that region.

Niles (2009) conducted winter aerial and ground counts along Florida's Gulf coast from 2006 to 2010 (table 6), covering essentially the same area in which Harrington *et al.* (1988, p. 440) had reported an average of 6,300 red knots (± 3,400) in the winters of 1980 to 1982. As the more recent aerial counts were lower, red knot numbers may have decreased in west Florida, perhaps due to birds shifting elsewhere within the larger Southeast wintering region (Harrington 2005a, p. 2). However, a comparison of the geographic coverage of Sprandel *et al.* (1997, p. 6) with Niles (2009, p. 2) suggests that red knot numbers did not change much from 1994 to 2010. The Panhandle and Big Bend regions (124 red knots total in 1994) generally correspond to the Honeymoon Key to St. George Island segment (2 flocks in 2009). The Southwest region (1,092 red knots in 1994) roughly corresponds to the Anclote Key to Cape Romano segment (5-year average of 1,451 from 2006 to 2010).

Table 6. Red knot counts along the Gulf coast of Florida, winters 2006 to 2010 (Niles 2009, p. 4; Dey *et al.* 2011a, p. 2)

Survey Segment	2006	2007	2008	2009	2010
Florida Gulf coast, Honeymoon Key to St. George Island (aerial)				2 flocks (Cedar Key and St. Mark's NWR)	
Florida Gulf coast, Anclote Key to Cape Romano (aerial)	2,301	1,530	515	1,532	1,378
Florida Gulf coast, Anclote Key to Cape Romano (ground)	2,142		458	1,463	
Florida Gulf coast, Cape Romano to Cape Sable (aerial)		0	0	0	Included in the total for Anclote Key to Cape Romano

Other Areas

Small numbers of wintering red knots have been reported from Maryland, United States, to Nova Scotia, Canada (Burger *et al.* 2012b, p. 6; BandedBirds.org 2012; eBird.org 2014; H. Hanlon pers. comm. November 22, 2012; A. Dey pers. comm. November 19, 2012), but no systematic winter surveys have been conducted in these northern areas. In surveys of 5 sites within North Carolina's Outer Banks in 1992 and 1993, Dinsmore *et al.* (1998, p. 178) found over 500 red knots per year.

Two more recent winter estimates are available for the central Gulf of Mexico. As shown in table 5, 250 to 500 knots were counted from Alabama to Louisiana during the International Piping Plover Censuses in 2006 and 2011 (L. Patrick pers. comm. August 31, 2012). From work related to the Deepwater Horizon oil spill, an estimated 900 red knots were reported from the Florida Panhandle to Mississippi (C. Hunter pers. comm. September 20, 2012). Older surveys recorded similar numbers from the central Gulf coast, with peak counts of 752 red knots in Alabama (1971) and 40 knots in Mississippi (1979) (Morrison and Harrington 1992, p. 77).

The red knot is an uncommon to rare winter resident or visitor in Mississippi, mainly on the offshore islands where winter visits by observers are scarce. Knots are recorded from all major islands from Cat Island east to Petit Bois Island, with peak counts in winter of only five birds at Horn Island. The peak mainland count is 74 birds at Long Beach in January 1986 (Russell 2014, p. 4). The red knot is an uncommon winter resident on the Alabama coast, occurring on both Mobile Bay and Dauphin Island. A high count of 70 birds, most likely a wintering flock, was recorded on March 3, 1971 at Dauphin Island. During recent decades, the highest count was a flock of 38 birds in December 2008 on the west side of Dauphin Island. Southern Alabama eBird records show knots 5 out of 10 years in winter. The Christmas Bird Count at Dauphin Island recorded red knots in 4 of the 10 years from 2004 to 2013, ranging from 0 to 46 birds and averaging 7.6 birds per year (Russell 2014, p. 2).

Numbers of red knots wintering in the Caribbean are essentially unknown, but in the course of piping plover surveys in February 2011 in the Bahamas, 70 red knots were observed on the Joulters Cays just north of Andros Island, and 7 knots were observed on the Berry Islands. In December 2012 (i.e., winter 2013), 52 red knots were observed in the Green Turtle Cay flats in Abaco, Bahamas (M. Jeffery pers. comm. February 13, 2013). Roughly 50 red knots occur annually on Green Turtle Cay (eBird.org 2014; T. Pover pers. comm. December 9, 2012).

Spring Stopover Areas

Records of migrating red knots have been collected at many sites along the Atlantic coast, not all of which are discussed in this document (see below). Not all migration areas are well surveyed, and considerable turnover of individuals occurs as birds migrate through an area. Consequently, using counts of migrating red knots as a basis for population estimates may lead to inaccuracies due to errors associated with turnover or double-counting. However, long-term counts made at a specific location are good indicators of usage trends for that area and, considered together, may reflect trends in the overall population of the red knot.

In this document, we focus on geographically large spring stopovers with multiple years of survey data, but we note that other important spring stopover areas are known (e.g., from ISS data, eBird, localized surveys). See Southeast United States, below, for examples of spring counts in North Carolina, South Carolina, Georgia, and Florida. For another example, red knot numbers in Louisiana increase in April and May, with a peak count of 530 birds on Grand Isle on May 1, 2004 (Russell 2014, p. 3). Likewise, a distinct passage occurs along the Gulf coast of Texas from late April to mid-May, with peak counts of 2,000 to 2,500 birds at Padre and Mustang Islands and 2,000 in the Galveston area (Morrison and Harrington 1992, p. 77). In South America, several hundred knots were observed using coastal mud flat habitat in the Commewijne district of Suriname during spring migration in April 2012 (NJAS *et al.* 2013). Reports of up to 1,700 *Calidris canutus* have been made in early May on the beaches west of the mouth of Lake Maracaibo in western Venezuela (figure 5) (Morrison and Harrington 1992, p. 77), at least some of which were likely rufa red knots (see Subspecies Nonbreeding Distributions).

South America

Peak counts of red knots declined at three South American stopover sites from the 1990s through the mid-2000s, as shown in table 7. Trends at stopover areas can reflect changing usage of the site. At Fracasso Beach, for example, Bala *et al.* (2002, p. 29) correlated the decline in 1999 relative to previous years to poor prey availability. Nonetheless, the prolonged duration and timing of these declines over roughly the same period as those in Tierra del Fuego and Delaware Bay (late 1990s to early 2000s) is more suggestive of an overall decrease in the southern-wintering population. At Fracasso Beach on Península Valdés in Argentina, ground surveys were conducted weekly from February through April (González 2005, p. 4). At Bahía San Antonio in Argentina, the surveys were ground-based counts conducted January to April, weekly through 1999, but varying from daily to every 10 days from 2000 to 2005 (González 2005, p. 4). Counts at Lagoa do Peixe in Brazil were obtained during expeditions that covered the peak spring passage in April (Niles *et al.* 2008, p. 58). Other observers noted 5,000 red knots at Lagoa do Peixe in April 2005 (Fedrizzi and Carlos in Lanctot 2009, p. 132) suggesting that usage of this site had partially rebounded.

Table 7. Peak counts of red knots at three Atlantic coast stopover sites in South America, spring 1990 to 2005 (Fedrizzi and Carlos in Lanctot 2009, p. 132; Niles *et al.* 2008, p. 58; González 2005, p. 14; Harrington *et al.* 1986, p. 50)

	Fracasso Beach, Argentina	**Bahía San Antonio, Argentina**	**Lagoa do Peixe, Brazil**
1984			13,750*
1990		19,700	
1992		15,000	
1994	8,000		
1995	2,625		10,000
1996	3,200	20,000	6,200
1997	2,000	15,000	7,500
1998		9,000	5,500
1999	3,020	10,500	7,000
2000	3,000	10,000	
2001		7,000	1,500
2002	80	12,000	1,200
2003	1,000	5,000	900
2004	2,000	5,500	
2005	500	6,500	5,000*

*Different observer(s) than the rest of the data for Lagoa do Peixe.

Southeast United States

From 2006 to 2012, coordinated red knot surveys were conducted from Florida to Delaware Bay during two consecutive days from May 20 to 24 (table 8). This period is thought to represent the peak of the red knot migration. There was some variability in methods, observers, and areas covered. From 2006 to 2010, there was no change in counts that could not be attributed to varying geographic survey coverage (Dey *et al.* 2011a, p. 12); thus, we do not consider any apparent trends in these data before 2010.

Table 8. Red knot counts along the Southeast coasts of the United States, May 20 to 24, 2006 to **2012** (A. Dey pers. comm. April 21, 2014)

State	2006	2007	2008	2009	2010	2011	2012
New Jersey	7,860	4,445	10,045	16,229	8,945	7,737	22,025
Delaware	820	2,950	5,350		5,530	5,067	3,433
Maryland			663	78	5	83	139
Virginia	5,783	5,939	7,802	3,261	8,214	6,236	8,482
North Carolina	235	304	1,137	1,466	1,113	1,868	2,832
South Carolina		125	180	10	1,220	315	542
Georgia	796	2,155	1,487		260	3,071	1,466
Florida			868	800	41		10
Total	15,494	15,918	27,532	21,844	25,328	24,377	38,929

Because red knot numbers peak earlier in the Southeast than in the mid-Atlantic (M. Bimbi pers. comm. June 27, 2013), the late-May coast-wide survey data likely reflect the movement of some birds north along the coast, and may miss other birds that depart for Canada from the Southeast along an interior (overland) route prior to the survey window. Thus, greater numbers of red knots may utilize Southeastern stopovers than suggested by the data in table 8. For example, Dinsmore *et al.* (1998, pp. 174, 178) found a mean of 1,363 (±725) red knots at 5 sites in North Carolina during spring 1992 and 1993, with a peak count of 2,764 birds. The NCWRC (2013) reported that Ocracoke Island, Portsmouth Island, North Core Banks, and South Core Banks are important areas for red knots in North Carolina during spring migration, with peak numbers occurring in May. A peak count of over 8,000 red knots was documented in South Carolina during spring 2012, with the largest flocks on Kiawah Island, Harbor Island, and in Cape Romain NWR (Wallover *et al.* 2014, p. 6; Kiawah Conservancy 2013; SCDNR 2013, p. 28; SCDNR 2012, p. 24). Significant numbers of both northern- and southern-wintering red knots stopover on the Georgia coast into late May (GDNR 2013); as shown in table 8 the late May coordinated survey flight routinely documents several thousand knots even though this survey lags behind Georgia's peak numbers. Peak counts of over 1,000 birds were observed at just a single Florida site (Cape Romano) between April 1 and June 10 of 1983, 1987, and 1989, and several other Florida sites routinely supported hundreds of red knots (peak counts) during spring from the 1980s through the 2000s (Niles *et al.* 2008, pp. 173–176).

Virginia

Aerial surveys of the entire chain of barrier island beaches in Virginia have been conducted since 1995 using consistent methods and observers (table 9). Although the number of surveys has varied from one to six per year, the aerial survey effort has consistently covered the peak period during the last week of May (B. Watts pers. comm. November 15, 2012). Since 2007, Karpanty e*t al.* (2012, p. 2) have estimated total red knots based on ground counts at 100 to 150 randomly selected points throughout Virginia's barrier island beaches including peat banks, with each location visited from one to three times per stopover season (table 10). Because of differences in methodology and timing, the two data sets (in tables 9 and 10) are not comparable.

Because birds pass in and out of a stopover area, the peak count (the highest number of birds seen on a single day) for a particular year is lower than the total passage population (i.e., the total number of birds that stopped at that site over the course of that migration season). Using resightings of marked birds, several attempts have been made to estimate the total passage population of Virginia through mathematical modeling (table 11), which should not be confused with the peak counts given in tables 9 and 10.

Although the data cannot be compared among tables 9, 10, and 11, each of these data sets show a short-term increase between 2006 and 2010. However, this time period is too short to represent the long-term spring red knot population trend in Virginia. Spanning a longer time period, the aerial counts (table 9) have been relatively steady since the mid-1990s. A recent analysis of data from 1995 to 2014 shows that the peak single surveys are not significantly different across decades, clearly due to the few samples in the 1990s and the high variance at that time. However, accumulated use ("total bird days") for the season is significantly different ($p< 0.05$) and the decline between 1990s and 2000s is about 25 percent (B. Watts pers. comm. August 22, 2014; Watts 2014).

Table 9. Peak counts of red knots in Virginia from aerial surveys, spring 1995 to 2013 (B. Watts pers. comm. August 19, 2014; Watts 2013; B. Watts pers. comm. November 15, 2012)

Year	Count
1995	7,958
1996	8,922
2005	9,150
2006	5,783
2007	5,939
2008	7,802
2009	6,079
2010	8,167
2011	6,086
2012	8,482
2013	6,200
*2014	5,547

*The 2014 survey may have been slightly before the true peak in bird numbers.

Table 10. Estimated peak abundance of red knots in Virginia from ground sampling, spring 2007 to 2013 (Karpanty *et al.* 2014, pp. 5–6; Karpanty *et al.* 2012; Cohen *et al.* in prep)

Year	Estimate	95 Percent Confidence Interval
2007	3,888	1,279 to 6,497
2008	5,176	1,926 to 8,426
2009	6,046	2,535 to 9,558
2010	6,271	2,496 to 10,045
2011	9,257	3,208 to 15,307
2012	11,781	5,236 to 18,327
2013*	5,799	Not available

*Preliminary results. Sharp decrease from 2012 may have been caused, at least in part, by a foraging shift from sandy intertidal zone to peat banks, which are less accessible to surveyors.

Table 11. Estimates of total passage population in Virginia, spring 2006 to 2010

Year	Peak Count	Estimated Passage Population	95 Percent Confidence Interval	Source
2006	5,783	7,224	6,460 to 7,986	Cohen *et al.* 2009, p. 942
2006	5,783	9,785		Duerr *et al.* 2011, p. 12
2007	5,939	8,332	6,924 to 9,740	Cohen *et al.* 2009, p. 942
2007	5,939	12,611		Duerr *et al.* 2011, p. 12
2008*	8,465	14,688		Duerr *et al.* 2011, p. 12
2009	6,079	13,398		Duerr *et al.* 2011, p. 12
2010*	8,172	12,959		Duerr *et al.* 2011, p. 12

* Peak count differs slightly from peak counts reported by B. Watts (pers. comm. November 15, 2012).

Delaware Bay

Aerial surveys have been conducted in Delaware Bay since 1981 (table 12). Methods and observers were consistent from 1986 to 2008. The methodology during this period involved weekly counts; thus, it was possible the absolute peak number of birds was missed in some years. However, since most shorebirds remain in Delaware Bay at least a week, it is likely that the true peak was captured in most years (Clark *et al.* 1993, p. 700). The surveys covered consistent areas of New Jersey and Delaware from the first week of May to the second week of June. All flights were conducted 3 to 4 hours after high tide, a period when birds are usually feeding on the beaches (Clark *et al.* 2009, p. 87).

Methodologies and observers changed several times since 2009. Flights are now flown only during the end of May. Another change since 2009 is that aerial counts are now adjusted with ground counts from Mispillion Harbor, Delaware to more accurately reflect large concentrations of birds at this key site (Dey *et al.* 2011a, p. 4). Further, problems in 2009, 2012, and 2013 prevented accurate aerial counts, and ground counts have been substituted. Caution should be used in comparing ground and aerial counts (see Research Methods, above, and Laursen *et al*. 2008, p. 165). Differences between the two methods may account for the markedly higher count in 2009. Although aerial counts had typically been higher than ground counts prior to 2009, this was likely because many areas that could be surveyed by air were inaccessible on the ground. Since 2009, ground survey crews have attempted to minimize the access problem by using boats in remote areas (A. Dey pers. comm. April 26, 2013; K. Clark pers. comm. March 4, 2013).

Comparing 4 different time periods, average red knot counts in Delaware Bay declined from 59,946 (1981 to 1983), to 46,886 (1986 to 1994), to 34,060 (1995 to 2004), to 18,387 (2005 to 2014). Average counts for the last time period (2005 to 2014) are about 70 percent lower than during the earliest period (1981 to 1983). However, numbers appear to have stabilized or increased slightly from 2009 to 2014, despite our lower confidence in the data over this later period due to shifts in methodology and surveyors. There may have been declines in the Delaware Bay stopover population prior to 2001, but variability in the data that makes it difficult to detect trends. In contrast, the decline in Delaware Bay red knot counts in the 2000s was sufficiently pronounced and sustained that we have confidence in the downward trend over this time period despite the variability in the data.

As with other stopover areas, it is impossible to separate population-wide trends from trends in usage of a particular spring site. Thus, differences in the number of birds in Delaware Bay may reflect stopover patterns rather than (or in addition to) trends in the overall red knot population (Clark *et al*. 1993, p. 702). However, comparing tables 9 and 12 shows the red knot decline in Delaware Bay cannot be explained by birds switching to Virginia as the final spring stopover, since Virginia counts did not change appreciably when comparing the years since 2005 with the mid-1990s, and in fact red knot usage of Virginia declined over this time period (B. Watts pers. comm. August 22, 2014; Watts 2014). We cannot rule out the possibility that some or all of the decline in Delaware Bay could have been caused by birds switching to other U.S. Atlantic stopover areas that, besides Virginia, have not been consistently surveyed. However, we consider this unlikely based on similarities in the magnitude and timing of the declines in Delaware Bay relative to Tierra del Fuego (table 3) and the South American stopovers (table 7). Thus, we conclude that the declines in Delaware Bay reflect (or contributed to, or both) a true and pronounced population decline in the Argentina-Chile wintering region, particularly considering that several lines of evidence suggest southern-wintering red knots are more reliant on Delaware Bay than are northern-wintering birds (see Migration and Wintering Food—Possible Differential Reliance on Horseshoe Crab Eggs). Because some northern-wintering knots (from northern Brazil, the Southeast) are also known to utilize Delaware Bay in spring (table 2), it is possible that Delaware Bay declines also reflect (or contributed to, or both) a population decline in one or more of these other wintering regions; however, the corresponding data sets from these northern wintering regions are insufficient to support this conclusion. Because birds pass in and out of a stopover area, the peak count for a particular year is lower than the total passage population. Using resightings of marked birds, several attempts have been made to estimate the total passage population of Delaware Bay through mathematical modeling (table 13), which should not be confused with the peak counts given in table 12. Because of differences in modeling methodology and sporadic temporal coverage, we do not infer any trends from the total passage population estimates given in table 13.

Table 12. Peak counts of red knots in Delaware Bay from aerial and ground surveys, spring 1981 to 2013 (A. Dey pers. comm. June 30, 2014; Dey *et al.* 2014, p. 3; Dey *et al.* 2011a, p. 3; Clark *et al.* 2009, p. 88; Kochenberger 1983, p. 1; Dunne *et al.* 1982, p. 67; Wander and Dunne 1982, p. 60)

Year	Peak Count	Year	Peak Count
1981*	67,450	1999	49,805
1982	95,530	2000	43,145
1983	16,859	2001	36,125
1986	58,156	2002	31,695
1987	38,790	2003	16,255
1988	34,750	2004	13,315
1989	95,490	2005	15,345
1990	45,860	2006	13,445
1991	27,280	2007	12,375
1992	25,595	2008	15,395
1993	44,000	2009**	24,000
1994	52,055	2010	14,475
1995	38,600	2011	12,804
1996	19,445	2012**	25,458
1997	41,855	2013**	25,596
1998	50,360	2014	24,980

*Only New Jersey was surveyed in 1981. For reference, the total numbers of red knots in Delaware Bay was relatively evenly distributed between New Jersey and Delaware from 1986 to 1992 (Clark *et al.* 1993, p. 700), suggesting that the true peak count for the bay could have been roughly double the number recorded in 1981.

** Data from 2009, 2012, and 2013 are from ground counts, while all other years are from aerial counts. For 2009, the actual peak ground count was 27,187, but Niles *et al.* (2010b, p. 10) chose to report 24,000 as the low end of an estimated 10 percent error range. The peak ground count in 2012 was also adjusted down (from roughly 29,400 to 25,458) based on concerns that some flocks in New Jersey were double counted.

Table 13. Estimates of total passage population in Delaware Bay, spring 1998 to 2012

Year	Peak Count	Estimated Passage Population	Range	Source
1989	95,490	152,900	±50,300 Standard Deviation	Harrington 2001, p. 22
1998-2001	36,125-50,360	77,000 (per year)	28,000 to 126,000 (per year) 95% Confidence Interval	Atkinson *et al.* 2002, p. 11
2004	13,315	17,108	14,515 to 19,701 95% Confidence Interval	Cohen *et al.* 2009, p. 942
2004	13,315	17,707	12,800 to 22,614 95% Confidence Interval	Gillings *et al.* 2009, p. 58
2006	13,445	19,555	17,927 to 21,184 95% Confidence Interval	Cohen *et al.* 2009, p. 942
2011	12,804	43,570	40,880 to 46,570 95% Confidence Interval	J. Lyons pers. comm. September 3, 2013
2012	25,458	44,100	41,860 to 46,79095% Confidence Interval	J. Lyons pers. comm. September 3, 2013
2013	25,596	48,955	*39,119 to 63,130 95% Confidence Interval	J. Lyons pers. comm. September 3, 2013

*The confidence interval was larger in 2013 in part because approximately 21 to 25 percent fewer marked individuals than were detected than in 2011 and 2012.

Fall Stopover Areas

Few regular surveys are conducted in fall because southbound red knots tend to be less concentrated than during winter or spring. No regular surveys are conducted in Hudson Bay or James Bay, Canada. However, aerial surveys of the Ontario coastlines of James Bay and Hudson Bay in the late 1970s produced totals of 7,000 to 10,000 red knots, with more recent surveys reporting 5,000 to 10,000 (Morrison and Harrington 1992, p. 74). There were numerous reports of 100 to 1,300 red knots at James Bay (Ontario) in August 2011, and one report of nearly 4,000 birds in this area (eBird.org 2014). These James Bay numbers are consistent with much earlier reports of “huge flocks” of 200 to 500 red knots at one James Bay site in July 1942 (Hope and Short 1944, p. 574).

Some fall survey data are available from southeastern Canada. Based on intensive field work and analysis of resightings of marked birds, at least 7,200 red knots are estimated to have used the Mingan Islands Archipelago (Canada) in fall 2008 (USFWS 2011b, p. 8; Wilson *et al.* 2010, p. 192). Using daily checklist data submitted by birdwatchers during fall migration from 1976 to 1998 in southern Quebec, Canada, Aubry and Cotter (2001, pp. 21–22) found a statistically significant decline in sightings of red knots. In surveys of Eastern Canada (New Brunswick, Nova Scotia, Prince Edward Island, and Newfoundland), Morrison *et al.* (1994) found that fall counts of red knots dropped 5.3 to 15.3 percent per year (depending on the statistical method used) from 1974 to 1991, with considerably greater decreases later in the study period; however, the findings were not statistically significant (Morrison *et al.* 1994, pp. 436, 439). Analyzing more years from this same data set, from 1974 to 1998, Morrison *et al.* (2001b, pp. 35–36) found a statistically significant (*p* less than 0.05) annual decrease of 17.6 percent in Eastern Canada. In the French Overseas Territory of Saint-Pierre-et-Miquelon, off the south coast of Newfoundland, Canada, the red knot is considered common in fall, though no trend information is available from this location (G. Humbert pers. comm. November 29, 2013).

Fall peak counts from ISS sites along the U.S. Atlantic coast ranged from 6,000 to 9,000 red knots during the mid- to late-1970s (Morrison and Harrington 1992, p. 75). In Maine, surveys documented between 300 and 600 red knots passing through as fall migrants in the late 1980s and early 1990s. However, fewer red knots are now evident during fall migration in Maine. Although site-specific numbers during the 2013 fall migration are not yet available, overall abundance declined to less than 150 red knots statewide, suggesting a decline (Maine Audubon 2013; Maine Department of Inland Fisheries and Wildlife (MDIFW) 2013).

In 9 years of surveys in the 1980s, an annual average of 1,661 ± 724 red knots used a stopover area on the mainland of western Cape Cod Bay during peak migration (Harrington *et al.* 1988, p. 440). Harrington *et al.* (2010a, pp. 188, 190) found that overall red knot numbers on the Massachusetts coast increased from the late 1940s to the early 1970s, especially on the mainland (western Cape Cod Bay), with a smaller increase on outer Cape Cod (see Historic Distribution and Abundance, above). After 1975, counts declined significantly on the mainland, but increased significantly on outer Cape Cod (Harrington *et al.* 2010a, p. 190). Maximum counts of red knots on western Cape Cod Bay were below 1,000 birds per year from 1987 to 2004, compared to annual peaks of about 1,000 to over 7,000 birds in earlier decades of the 20th century (Harrington *et al.* 2010a, p. 190). Conversely, maximum counts on Cape Cod were typically 1,000 to 2,000 birds per year from 1994 through 2004, compared to peak counts frequently below 1,000 in earlier decades (Harrington *et al.* 2010a, p. 190). Evidence suggests that both the mainland and the Cape Cod areas were historically used by southern-wintering knots having Argentina-Chile destinations, but that more recently the Cape Cod locations have increasingly been used by northern-wintering knots with destinations in the Southeast United States, roughly balancing out the declining numbers of knots with Argentina-Chile wintering destinations (Harrington *et al.* 2010a, p. 188).

Although the numbers of southern-wintering knots are now greatly reduced in Massachusetts, especially at the sites on western Cape Cod Bay, this decline began earlier (i.e., the late 1970s) than the more recent declines documented in Argentina and Chile (table 3) and Delaware Bay (table 12), suggesting that the declines in Massachusetts may have related to shifts in distribution rather than (or in addition to) changes in population size (Harrington *et al.* 2010a, p. 192; Harrington *et al.* 2010b, p. 361). Possibly reflecting another change in distribution, the increasing numbers of northern-wintering birds using Cape Cod may represent a shifting of fall stopover areas from Georgia to Massachusetts, based on declining September numbers in Georgia (Harrington *et al.* 2010a, p. 192; Harrington *et al.* 2010b, p. 361).

No regular fall counts are currently conducted in Massachusetts (S. Koch pers. comm. November 19, 2012), but flocks of over 100 knots are routinely reported from Monomoy NWR (eBird.org 2014). Anecdotal information from the Jamaica Bay NWR in western Long Island, New York suggests a localized decline in fall migrants, with up to 200 August birds in early 1980s compared to only 2 or 3 August birds since 2004 (R. Kurtz pers. comm. October 18, 2013). About 1,500 red knots were present in Avalon on the Atlantic coast of New Jersey in the fall of 2011 (USFWS 2011c, p. 27). Also on the Atlantic coast of New Jersey, hundreds of red knots are regularly reported from North Brigantine and Stone Harbor, sometimes in flocks of over 500 (eBird.org 2014). Dinsmore *et al.* (1998, pp. 174) found a mean of 267 (±27) red knots at 5 sites in North Carolina during fall 1992 and 1993. In South Carolina, a peak count of 1,185 birds in August 2008 was documented on Raccoon Key, an island within Cape Romain NWR (Wallover *et al.* 2014, p. 6). In 2012, a flock of 100 birds was observed in August on Botany Bay and a flock of 400 birds was documented in September on Harbor Island, both in South Carolina (SCDNR 2013, p. 28).

Islands at the mouth of the Altamaha River, Georgia, support a large, late summer and fall staging site, attracting as many as 12,000 knots at one time (Schneider and Winn 2010, p. 2), although these Georgia numbers may have declined, possibly as a result of northern-wintering knots shifting their fall stopover to Massachusetts (Harrington *et al.* 2010a, p. 192; Harrington *et al.* 2010b, p. 361). The importance of the mouth of the Altamaha River to southbound migrating red knots has been known since 1997 (GDNR 2013). Numbers typically peak in mid- to late September, with peak counts of 5,000 and 10,000 birds and flocks of 3,000 to 8,000 birds (GDNR 2013; Harrington *et al.* 2007, p. 37). An extensive resighting study on the Altamaha River in the fall of 2011 allowed for an estimate of the total passage population of red knots moving through the system (which is larger than the peak count due to turnover). Between early August and late October, the estimated total stopover population size was 22,900 (95 percent confidence interval: 20,944 to 25,115). Based on resight and isotope data, the proportion of these birds that were part of the northern-wintering group was between 83 and 96 percent. Incorporating both uncertainties in the percentage of northern-wintering birds, and in the total number of birds on the Altamaha, the size of the northern-wintering group using the Altamaha during fall 2011 was estimated at 20,364 (95 percent confidence interval: 18,014 to 22,990). Without knowing what percent of the northern-wintering knots were in Georgia during the fall of 2011, it is not possible to estimate the total northern population; thus, this is considered a minimum estimate (GDNR 2013). See Population Surveys and Estimates—Wintering Areas—Southeast United States and Caribbean.

The Texas coast is not systematically surveyed in fall, but a high count of 1,600 red knots was recorded in October 2005 along a 40 mi (64 km) stretch of Padre Island National Seashore’s gulf beach, and other recent fall counts on the same beach range as high as 2,547 (Newstead *et al.* 2013, p. 54).

Raffaele *et al.* (1998, p. 277) described red knots as generally rare (occurring less than twice per year) through the West Indies in September and October during the southbound migration, and very rare (occurring less than every 5 years) in the Lesser Antilles except on Barbados where they occur regularly. Many birds likely overfly the West Indies, particularly while migrating southward. These authors list the red knot as rare in the Bahamas, Cuba, and Saint Barthélemy, and as very rare in Dominca, Guadeloupe, and Martinique. Red knots were classified as uncommon (occurring at least twice per year) in Puerto Rico, and common (one or more seen daily) in Barbados (Raffaele *et al.* 1998, pp. 464–465). Additional fall sightings data are available for certain parts of the Caribbean, as discussed below.

In the Bahamas, up to 50 knots have been reported from Abaco in October and November (T. Pover pers. comm. October 23, 2013; eBird.org 2014). In the Bahamas in October 2012, 124 red knots were documented on the Joulter Cays, one with a band, and 43 knots were observed on the Berry Islands, some roosting in mangroves (M. Jeffery pers. comm. February 13, 2013). Puerto Rico and the some of the Lesser Antilles (e.g., St. Croix in the U.S. Virgin Islands, Guadeloupe, Martinique, Barbados, and Trinidad) are also used as fall stopover areas (G. Humbert pers. comm. November 29, 2013; Niles *et al.* 2010a, pp. 125–126; eBird.org 2014), with birds occurring regularly but in small numbers. A 1976 account notes that the red knot was not well known by hunters in the French Antilles, and did not even have a creole name (G. Humbert pers. comm. November 29, 2013). Until 1950, red knot was catalogued as rare, if not exceptional in the West Indies. Since then, however, red knots have been noted to pass through the islands at least occasionally (G. Humbert pers. comm. November 29, 2013). In Guadeloupe, the red knot is an uncommon but regular visitor during fall migration, typically in small groups of up to 3 birds, but as many as 16 have been observed in a flock (A. Levesque pers. comm. October 11, 2011). Although not as regular in Martinique as in Guadeloupe, several sightings are known from there, including two fall eBird records (G. Humbert pers. comm. November 29, 2013). In Barbados, the red knot is a fairly regular fall transient in small numbers, usually occurring as single individuals and in small groups, but very exceptionally knots may occur in flocks of up to a dozen birds, and a group of 63 birds was recorded in 1951. Detailed records from 1950 to 1965 show an average of about 20 red knots per year in Barbados (Hutt and Hutt 1992, p. 70). Flocks of up to a dozen red knots were reported from Trinidad each year from 2008 to 2011, with multiple sightings each fall (eBird.org 2014) (figure 5).

Recent evidence suggests at least 2,000 red knots pass through the Guianas during southbound migration (NJAS *et al.* 2013). Flocks of 340, 500, and 1,000 knots have been reported in French Guiana during fall, all near Mana. In late August 2012, 1,700 knots (the largest flock reported to date) were observed in rice fields near Mana, French Guiana (CSRPN 2013; Niles 2012b). A large number of these birds at Mana had been marked in the Chile portion of Tierra del Fuego (Niles 2012b). Based on these reports and geolocator results, French Guiana is emerging as an important fall stopover area (Niles 2012b). Adjacent Suriname and Brazil are also used in fall (Niles *et al.* 2010a, pp. 125–127; Spaans 1978, p. 72), but little information is available regarding the numbers of birds in these areas. In Suriname, a total of nearly 160 red knots were counted during 2 surveys conducted in late August of 1970 to 1973. Larger red knot numbers apparently do not occur in Suriname as the habitat is not ideal (B. Harrington pers. comm. March 31, 2006); see Migration and Wintering Habitat, above. However, there are sandy beach habitats in Suriname's Marowijne District, less than 31 mi (50 km) from Mana, French Guiana (NJAS *et al.* 2013), suggesting that southbound migrating knots may also use this portion of nearby Suriname. In September 2007, the average peak count of red knots at Cajuais Bank in the Brazilian State of Ceará was 434 ± 95 (Carlos *et al.* 2010, pp. 10–11) (see figure 5).

Inland Areas—Spring and Fall

Red knots are restricted to ocean coasts during winter, and occur primarily along the coasts during migration. As described above under Migration, however, the known spring and fall migratory paths of the rufa red knot include flights over the interior portions of both North and South America (Newstead *et al.* 2013, entire; Niles *et al.* 2012a, pp. 197–200; Niles 2011a; Harrington 2005a, p. 1; Morrison and Harrington 1992, p. 77). We generally lack sightings data for inland portions of South America, but present below available sightings data for inland portions of the United States and portions of southern Canada. Small to moderate numbers of rufa red knots are reported annually across the interior United States and Canada (i.e., greater than 25 miles from the Gulf or Atlantic Coasts) during spring and fall migration—these reported sightings are concentrated along the Great Lakes and Northern Great Plains of southern Canada, but multiple reports have been made from every interior State (Central Flyway Council 2013; eBird.org 2014). (See Migration—Midcontinent, above).

Definitions of the terms "casual, rare, uncommon, and irregular" vary . Typically, however, "casual" and "irregular" imply less than annually recorded; "rare" implies near-annual occurrence but usually single individuals to very small flocks of 2 to 5 birds; and "uncommon" usually implies annual occurrence, but not always to be encountered daily and from 1 to 10 birds, seldom more (Russell 2014, p. 1). In many inland parts of its range, the rufa red knot meets one or more of these definitions. In contrast, we understand the term "vagrant" to mean a bird that has strayed or been blown far from its usual range or migratory route; synonymous with "accidental." According to Russell (2014, p. 1), "accidental" implies an extraordinary record, out of the normal pattern, and unlikely to occur again. We do not consider the rufa red knot "vagrant" or "accidental" within any U.S. or Canadian parts of its range as described under Species Nonbreeding Distributions, above.

Midcontinental (Central) Flyway

Canada

Large numbers (1,000 to 2,500) of red knots have been reported in Saskatchewan and Alberta, Canada, between January and June (Skagen *et al.* 1999), typically in the latter half of May (Morrison and Harrington 1992, p. 77). In southern Alberta, Saskatchewan, and Manitoba, eBird.org (2014) shows about 160 records of red knot observations since 1968. Reports are annual or nearly annual, including 28 records of 10 or more birds and 8 records of 100 to 258 birds. Localities with multiple records include Beaverhill Lake, Brooks, Chaplin Lake, Frank Lake, Kitsam Reservoir, Oak Hammock Marsh, Rumsey, and Reed Lake (eBird.org 2014).

Montana

The status of the red knot has not been ranked in Montana as it is rarely recorded in the State. The Montana Natural Heritage Program's database shows 34 detections for red knot between 1982 and 2013, averaging 2.9 birds per year across the past 30 years. The number of individuals recorded generally ranged from one to four birds and on only three occasions were eight or more birds recorded. Red knots were detected both during spring migration (20 records in May) and fall migration (14 records between late July and mid-September). While *Calidris canutus* records come from locations across the State, including west of the continental divide, a majority of records (roughly 64 percent) come from three areas in the northern part of the State: Freezeout Lake and Benton Lake NWR near Great Falls, Bowdoin NWR near Malta, and scattered lakes in the northeast corner of the State, including Medicine Lake NWR. Even in these areas there are many years in which red knots are not recorded—there is no evidence that these locations are used annually or frequently as stopover sites (Montana Fish, Wildlife, and Parks (MFWP) 2013). However, from a relatively small sample of Texas-wintering knots from which geolocator data have been retrieved, two stopped in northern Montana during migration (D. Newstead pers. comm. May 16, 2014).

Figure 6. Major rivers of the western United States

North Dakota

The red knot is considered a rare migrant in North Dakota (NDGFD 2013). The species may occur annually somewhere in the State although in extremely low numbers. The archives of the ND-BIRDS listserv contain 20 records of the red knot in North Dakota from 2000 to 2013. There are approximately six records in eBird over that same time period. *North Dakota Bird Note* records prior to 2000 reveal similar numbers of observations (i.e., roughly one to two observations per year). Most observations occur in mid-May and mid-September. Almost all sightings are of one to three individual birds, with a few exceptions. In May 2005, 25 red knots were observed at Long Lake NWR. In July 2009, 13 were observed at the West Fargo Sewage Lagoons, and in May 2012, 8 red knots were observed at New John's Lake. While there is no coordinated monitoring effort for red knots or other migratory shorebirds in North Dakota, the NDGFD (2013) believes the limited birding records of red knots support its status as a rare migrant. However, from a relatively small sample of Texas-wintering knots from which geolocator data have been retrieved, four have stopped in northern North Dakota during migration (D. Newstead pers. comm. May 16, 2014; Newstead *et al.* 2013, p. 560). At least one of these geolocator birds stopped in a saline environment (e.g., natural alkaline lake or wetland) (D. Newstead pers. comm. May 16, 2014), and such habitats are remote and surveyed little (or not at all) during the red knot's spring migration season (C. Aron pers. comm. May 19, 2014). In addition, along a portion of the Missouri River in North Dakota with only one reported eBird record, red knots have been "regularly" observed, but not recorded, incidental to other research since 2005 (C. Dovichin pers. comm. May 6, 2014) (see figure 6).

Wyoming

The red knot is classified a rare migrant in Wyoming. It is a species for which the Wyoming Bird Records Committee requests documentation on all sightings. Since 1979, observations of the red knot have been documented in 9 of the State's 28 latilongs, which are rectangular areas between adjacent meridians of longitude and parallels of latitude, averaging about 3,200 mi^2 (8,436 km^2). Red knot observations in the Wyoming Game and Fish Department's (WGFD) database total 10 individual birds, reported in 1982, 1997, 1988, and 2008 (WGFD 2013).

South Dakota

The South Dakota Ornithologists' Union's sightings database contains uncommon and sporadic sightings of the red knot since 1970, consisting of 26 sightings in the past 43 years. Of these, 17 were in spring, 1 in summer, and 8 in fall. Based on data published in *South Dakota Bird Notes*, the highest numbers of red knots were 5 birds at LaCreek NWR in southwestern South Dakota in May 2003 and 30 birds at Lake Preston in eastern South Dakota during August of 2007. Other reports ranged from one to three birds. Fifteen South Dakota counties had red knot records as of November 2013, all at the lowest category of less than 0.25 sightings per year (Central Flyway Council 2013). All but three of the database records are on the East River or near the Missouri River, but this may reflect areas of higher bird watching activity (N. Drilling pers. comm. May 13, 2014) (see figure 6).

Nebraska

In Nebraska, the red knot is considered a casual spring and fall migrant, meaning it has occurred at least twice for a particular season, but does not occur annually (Central Flyway Council 2013). In a comprehensive review of Nebraska bird records covering more than 100 years, there were 15 documented records for red knot as of 2001. The Rainwater Basin in south-central Nebraska is recognized as a landscape of hemispheric importance by WHSRN because of the region's role in supporting midcontinental shorebird populations. During intensive shorebird surveys in this region from 1997 to 2001, red knot was observed on only one occasion. A more recent (2012) review of Rainwater Basin avifauna shows that red knot has been recorded on only three occasions. The red knot was not recorded during shorebird surveys at Nebraska's eastern saline wetlands in 1997, and as of 1994 there were only two records at Lake McConaughy and the North Platte River valley (Central Flyway Council 2013) (see figure 6). There are no Nebraska records in eBird.org (2014). However, from a relatively small sample of Texas-wintering knots from which geolocator data have been retrieved, one may have stopped in Nebraska during migration (D. Newstead pers. comm. May 16, 2014).

Colorado

The red knot is classified as a rare fall migrant and very rare spring migrant in eastern Colorado (Central Flyway Council 2013). Since 1953, the Colorado Bird Records Committee of the Colorado Field Ornithologists' has verified 27 records of red knot in Colorado. Six of those observations were during the spring migration and the remaining 21 during fall migration. Records of red knots in Colorado are highly variable both temporally and spatially and in many years no observations are recorded. Red knots have been recorded primarily at scattered reservoirs in the South Platte and Arkansas River drainages (see figure 6) as individuals or small flocks, though no reservoirs are locations of consistent observations (Central Flyway Council 2013). Skagen *et al.* (1999) reported a spring peak count (January and June over the period 1980 to 1996) of 38 knots at Longmont, Colorado.

Kansas

The red knot is considered a rare spring and fall transient in Kansas, with most observations limited to wetlands associated with Cheyenne Bottoms State Wildlife Area and Quivira NWR in the central region of the State (Central Flyway Council 2013). At these two sites and a few others along the Arkansas River (see figure 6), eBird.org (2014) shows roughly 100 records of red knot observations reported annually or nearly annually since 1976. Although most of these eBird records are of small numbers of birds, 19 records are for 10 or more birds, including maximum counts of 81, 128, and 160 red knots. More than 70 percent of these observations were reported during fall migration, most as part of the ISS (eBird.org 2014). Skagen *et al.* (1999) reported a fall peak count (July through December over the period 1980 to 1996) of 182 knots at Cheyenne Bottoms. Both Cheyenne Bottoms and Quivira NWR are designated as Ramsar wetlands of international importance and WHSRN sites, and have extended histories of wetland protection and management for wildlife (Central Flyway Council 2013). Historically, knots have been considered an irregular, rare transient in fall at Cheyenne Bottoms. There are 15 counties in Kansas with confirmed records for red knot. In a recent 5-year survey of over 50 shorebird stopover sites in Kansas, only one occurrence of red knots was documented, at Quivira NWR. Most knot observations have occurred in mid-August through September, with spring observations typically in May, especially the last half of the month (Central Flyway Council 2013).

Oklahoma

The red knot is considered to be a rare and irregular migrant in Oklahoma (Central Flyway Council 2013). The ODWC (2013) has 40 records of red knots in Oklahoma from 1941 to 2012, with 39 of these documented since 1962 for an annual rate of occurrence of 0.78 records per year (or 0.39 records per migration event). In contrast, eBird.org (2014) has only five records from 1979 to 2013. The 40 ODWC (2013) records represent 18 unique locations in 15 counties. Despite evidence that a population of red knots annually migrates through the Great Plains (Newstead *et al.* 2013, entire), the observational data on the ground indicate that red knots make landfall in Oklahoma only infrequently (ODWC 2013). No sites are used annually, and only 3 Oklahoma locations have more than 2 documented occurrences—Hefner Reservoir (14 observations), Salt Plains NWR (7 observations), and Oologah Reservoir (3 observations) (Central Flyway Council 2013). The majority of these observations (30 out of 40) consist of single birds, while the remaining observations are of small groups of 2 to 7 birds each (ODWC

2013). However, Skagen *et al.* (1999) reported fall maximum counts (July to December over the period 1980 to 1996) of 45 red knots at the Oklahoma's Oologah Reservoir and 22 red knots on the Arkansas River at Bixby, Oklahoma (see figure 6). Most of the observations reported by ODWC (2013) have occurred during fall migration (34 out of 40 records).

All but one of the 40 red knot records for Oklahoma are associated with man-made impoundments, including 32 on large reservoirs; 3 on large, managed wetland complexes; and 3 on sewage lagoons and fish hatchery ponds (Central Flyway Council 2013). Two large reservoirs in the Oklahoma City area, which are heavily visited by birders, account for nearly half of the total records (16 out of 40) (Central Flyway Council 2013). Red knots have been observed on Oklahoma City reservoirs when low water levels exposed substantial areas of mudflat and shoreline, and during or immediately after inclement weather (ODWC 2013). However, if weather conditions alone were the causative factor in these landfalls, we would expect to see a mix of group sizes and larger flocks of birds. Because most of Oklahoma's records are of single birds, an alternative hypothesis is that these are weakened birds or, in fall, inexperienced juvenile birds not capable of making the sustained 2- or 3-day migration flight in the face of unfavorable weather (ODWC 2013).

Texas

Inland records in Texas are sparse and the species there is considered very rare to casual. The Texas Clearinghouse database included five inland observations of red knots (in Tarrant, Travis, and Waller counties) from 1990 to 2006, involving a total of only nine individual birds (Central Flyway Council 2013).

Mississippi Flyway

The red knot is a regular, normally "rare" (near-annual but usually single individuals or very small flocks of 2 to 5 birds) spring and fall migrant along the shores of the Great Lakes, and a "casual" (less than annual) migrant inland throughout the Mississippi Flyway (Russell 2014, p. 1). Between 25 and 100 birds are recorded annually in spring away from the Gulf coast, with autumn numbers in the 100- to 200-bird range, the majority along the shores of Lakes Michigan and Erie. The knot is a low-density migrant and most records are of singles, pairs, or small flocks of 3 to 10 birds. The species appears to be very opportunistic and can occur almost anywhere along the Great Lakes shores or inland on mudflats of falling reservoirs in late summer and autumn or flooded fields in spring. The northern shoreline of Ohio, particularly at Ottawa NWR and the Point Mouille region of southeastern Michigan, are visited regularly, particularly in fall migration. On rare occasions, spring flocks heading north, likely from Delaware Bay, have strayed into northern Ohio due perhaps to adverse winds from the northeast, while on the west side of the region the same thing has occurred from presumably northbound birds from the western Gulf coast wintering population, which perhaps encounter headwinds and drop down in western Missouri and western Iowa (Russell 2014, pp. 1–2). The following State summaries are from Russell (2014, entire) unless otherwise noted.

Minnesota

The red knot is a rare, low-density migrant annually recorded somewhere in the State, most frequently at Park Point, Duluth, where it has been seen in 6 of 10 years, and along the larger inland lakeshores such as Upper Red Lake, Leech Lake, Mille Lacs, and Lake of the Woods. Occasionally, this species appears at sewage treatment plants in the southern third of the State and at other wetlands in the prairie region. A maximum flock size of 15 (date unspecified) has been noted at Duluth. Recent reports are usually of only 1 to 2 birds.

Wisconsin

The red knot is an annual but rare to "uncommon" (annual occurrence but not always encountered daily, seldom more than 10 birds) migrant, mainly recorded from the Great Lakes shorelines in spring, and from both the Great Lakes shorelines and inland sites in fall, mainly on State or Federal wildlife refuges. Although annually found along the Lake Michigan shoreline in fall, no one locale regularly concentrates the birds. They occur about every other year at Wind Point near Racine and nearly that frequently along the Milwaukee lakefront. Small flocks of 6 to 15 birds occasionally occur in spring, but numbers in recent years seem reduced with mostly reports of 1 to 4 individuals. Areas along the Great Lakes shores where the red knot was reported more than once include Manitowoc, Raspberry Island in the Apostle Island chain, southern Chequamegon Bay near the mouth of Whittlesey Creek, and in southern Green Bay.

Michigan

The red knot is a regular, low-density spring migrant on the shores of the Great Lakes, more frequently recorded in fall on Lakes Erie and Lake Michigan. The only specific site where the species is recorded annually is at the Point Mouillee State Game Area on the western shore of Lake Erie. Another likely regular site, but not birded as frequently, is Tawas Point State Park where eBird records knots 7 out of 10 years spring and fall. Other areas of occasional concentration are the Muskegon County Wastewater Facility, northwestern Lake Michigan in the Stonington and Garden Peninsula areas (especially fall), and at Whitefish Point in the eastern Upper Peninsula. Spring migrants are most likely to occur along the eastern border at Point Mouillee, Tawas Point, and Whitefish Point. Fall migrants are most frequently recorded along Lake Michigan or at Point Mouillee with fewer reports from Tawas Point and Whitefish Point. The peak total was 72 birds in Monroe County on May 23, 1989. Most recent reports are of single birds with a recent high of 10 at Tawas Point State Park on May 23, 2004.

Iowa

The red knot is a casual migrant, with 14 fall and 3 spring records. Recent eBird reports suggest that the species does not annually occur in the State. A high count of 14 birds in alternate plumage was made on May 21, 1934 but since that time most reports are of only 1 to 4 individuals. No regularly used site is evident, although large reservoirs such as Coralville and Saylorville appear to be attracting the species in recent years more than are isolated lakes and wetlands.

Illinois

The red knot is an "irregular" (less than annual) spring migrant and a regular, low-density fall migrant along the shores of Lake Michigan. Inland, this species is a casual spring migrant and an irregular, low-density fall migrant, occurring mainly in the vicinity of major reservoirs such as Rend Lake and Carlyle Lake and at refuges in the Illinois River Valley (see figure 6) such as at Chautauqua NWR. Sites along Lake Michigan where the red knot has occasionally been recorded include the Lake County shoreline from Illinois Beach State Park south to Waukegan and the Great Lakes Naval Training Base harbor, Wilmette Beach, and the Chicago lakefront. A high count of 48 knots was recorded at Waukegan on May 19, 1983, but most spring numbers are below 5 birds and there has been a recent diminution of spring reports.

Indiana

The red knot is a regular, low-density fall migrant along the Lake Michigan shoreline from Michigan City to Gary and casual inland. It is a very rare spring migrant in the northern two-thirds of the State with only a handful of records in 100 years. A peak of 18 knots was recorded between Gary and Indiana Dunes State Park on August 21, 1920. Most reports are of single individuals or small flocks, typically appearing only 3 out of 10 years.

Ohio

The red knot is usually a rare spring migrant with most records clustered in the two western Lake Erie counties of Ottawa and Lucas. Although most reports document 2 to 8 birds, on rare occasions significant numbers have occurred including, 150 at Bay Point, Ottawa County, and 49 at Ottawa NWR on May 17, 1980. Based on departure dates and peak numbers of spring migrants from the western Gulf coast, these birds are most likely originating from Delaware Bay staging birds, which peak in the latter third of May and on rare occasions stray to the west after departing the mid-Atlantic region. The destination of these birds in fall migration though, remains unclear as many fall Lake Erie shorebirds are known to head eastward towards the Atlantic while others move to the Gulf of Mexico shores. It is possible that both Gulf coast wintering birds and Atlantic coast migrants mix in Ohio in the July to September period. Fall migration is more widespread along the southern Lake Erie shoreline, with birds occurring along the entire shoreline from Toledo in the west to the Pennsylvania line in the east. Knots are uncommon fall migrants, with 10 to 20 reports most years, mostly singles or small flocks, usually of 6 or fewer. Peak fall counts include an exceptional 43 at Cleveland on September 9, 1984. Knots are rare migrants inland in Ohio spring and fall. Rarely, small flocks have occurred, with 60 in Wayne County on May 19, 1983, 17 in Wayne County on May 19, 1997, and 21 at Lake St. Marys on October 10, 1956. Inland numbers in recent decades have not approached these numbers and may reflect the overall decline in the species' numbers.

Missouri

The red knot is a rare transient in Missouri, not recorded annually in eBird (4 out of 10 years). Reports appear evenly divided between spring and fall. Apart from a single April and a single early June record, all spring records are from between May 11 and 26. There are only about a dozen spring records from 1950 to 1990 and five records since then. A high count of 30 on May 11, 1980, at Swan Lake NWR is perhaps indicative of casual eastward movement and stopover of Great Plains migrating birds. Most reports are from Swan Lake NWR, Squaw Creek NWR, Horseshoe Lake in Buchanan County, or the Mississippi River area near St. Louis.

Kentucky

The red knot is an extremely rare spring migrant and rare fall migrant. Older records were from the Falls of the Ohio at Louisville, but recent records are largely from the western part of the State. All four spring records are from the ephemeral lakes of Warren County. Found at Kentucky Lake 2 out of 10 years. The species is not annually recorded in the State. Knots are usually encountered as singles or pairs, rarely in small flocks.

Arkansas

The red knot is a rare fall transient in August and September with only two spring records. The largest numbers (10) have appeared at large reservoir drawdowns such as at Lake Millwood. Most records though have occurred at fish farms and sod farms in Prairie and Lonoke Counties. This species is not annually recorded, with eBird reports in 4 of 10 years. The species may occur occasionally on Mississippi River sandbars when water conditions permit, but those areas are very seldom surveyed.

Tennessee

The red knot is a rare fall migrant, but not annually recorded, and a casual spring migrant in Tennessee. As of 1990, there were 19 State records, all but one from fall (K. McDonald pers. comm. April 8, 2014; Russell 2014, p. 5). However, the State's previous high count of seven birds was in spring. Most of these 19 reports are from eastern or western Tennessee, with only 3 from the central portion of the State. Since 1990, there have been 20 additional reports in Tennessee, all in fall. Of these 20 reports, 14 are from counties bordering the Mississippi River, 5 are from East Tennessee, and 1 is from Middle Tennessee. These 20 newer reports also include a new State high count of 14 birds on September 8, 2004, at South Holston Lake in Sullivan County (South Fork Holston River). This species has been reported 3 out of 10 years in eBird. Mississippi River sandbars occasionally attract birds in the far west when water conditions permit. Sites with multiple records include Memphis (Mississippi River), Gallatin Steam Plant (Cumberland River), Kingston Steam Plant (Tennessee River; see figure 6), Pace Point (Tennessee NWR, Big Sandy Unit, Tennessee River), Island 13 in Lake County (Mississippi River), and Rankin Bottoms (French Broad River) in Cocke County. Biologists also report red knots over a span of several years at Rankin Bottoms Wildlife Management Area in East Tennessee (on Douglas Lake, French Broad River) and at sites on Old Hickory Lake along the Cumberland River (Snow Bunting Peninsula and Gallatin Steam Plant). All but one of these reports from Douglas and Old Hickory Lakes are during fall migration, and may include a new high count of about 15 birds in September 2011 at Snow Bunting Peninsula (K. McDonald pers. comm. April 8, 2014; Russell 2014, p. 5). Douglas and Hickory Lakes, and all Tennessee counties bordering the Mississippi River, may serve as frequent interior red knot stopover locations (K. McDonald pers. comm. April 8, 2014).

Gulf Coast States

Outside of coastal areas (i.e., north of Interstate 10 to the Arkansas border), the red knot is casual in Louisiana during migration (Russell 2014, p. 3). Inland Louisiana records include Bonnet Carre Spillway and two fall reports from New Orleans (Purrington 2012, p. 65), as well as older reports from Louisiana State University, Shreveport, and the Natchitoches Fish Hatchery

(Lowery 1974, pp. 309–309). In inland parts of Mississippi, the knot is a casual migrant with fewer than 10 records. In inland parts of Alabama, the red knot is casual everywhere except at Wheeler NWR in the Tennessee River Valley in the northeast, where there are a handful of records (see figure 6).

Central America and Pacific South America

As discussed above (Subspecies Nonbreeding Distributions), we lack data to determine the subspecies composition of *Calidris canutus* in much of Central America and on the Pacific coast of South America, but we conclude at least some of these birds (e.g., in Panama, Chiloé Island, Chile) are rufa red knots. Thus, we have evaluated what limited abundance data are available for this region, presented below.

In Laguna Superior (see figure 2) and other lagoons on Mexico's southern Pacific coast (State of Oaxaca), *Calidris canutus* are frequently seen during winter, sometimes exceeding 300 birds (Newstead 2014b, p. 1). Five birds marked in Texas during spring have been observed in Laguna Superior in winter, but no data are available to indicate if these five birds are *C.c. rufa* or *C.c roselaari* (Newstead 2014b, pp. 1–2, 4; D. Newstead pers. comm. May 13, 2014). However, we conclude that at least some of the *C. canutus* in southern Mexico are the *roselaari* subspecies based on the one geolocator track available to date for this subspecies (D. Newstead pers. comm. February 13, 2014).

Carmona *et al.* (2013, p. 171) compiled *Calidris canutus* abundance data from Christmas Bird Counts, eBird, and other sources. From Guatemala to Panama, these authors found numerous eBird records in Costa Rica and Panama, but none in other Central American countries (Carmona *et al.* 2013, p. 171). However, we reviewed records from eBird.org (2014) and also found reports of small numbers of *C. canutus* from Belize, El Salvador, Guatemala, Honduras, and Nicaragua. Most records from Costa Rica and Panama involved very small (typically 1 to 2 birds) or unreported numbers of birds, and all 4 counts of 20 or more birds were from Golfo Nicoya on the Pacific coast of Costa Rica (Carmona *et al.* 2013, p. 172) (see figure 2). Smith and Stiles (in Pitelka 1979, pp. 41–43) previously reported *C. canutus* wintering in Golfo Nicoya in Costa Rica, with peaks between 250 and 500 birds in January and February. *Calidris canutus* has generally been considered an uncommon spring and autumn migrant through coastal areas of Guatemala, Honduras, El Salvador, Nicaragua, Costa Rica and Panama (Carmona *et al.* 2013, p. 174). We have no data regarding the subspecies composition of *C. canutus* on the Gulf and Caribbean coasts Central America south of Tamaulipas, Mexico through Panama. On the Pacific coast of Central America, from Guatemala through Costa Rica, we conclude that at least some of the *C. canutus* are the *roselaari* subspecies based on the limited available resighting and geolocator data, but have no data to indicate whether or not *C.c. rufa* also occurs in this region (see Subspecies Nonbreeding Ranges).

No *Calidris canutus* were detected in aerial surveys Panama's Pacific coast in February 1988, October 1991, or January 1993, but many unidentified medium-sized shorebirds were recorded (Carmona *et al.* 2013, p. 174). On the southern (Pacific) coast of Panama, Buehler (2002, p. 43) counted 200 *Calidris canutus* near Panama City in January 2002 (see figure 2). In February 2002, Buehler (2002, p. 43) counted 100 *C. canutus* at this same site near Panama City, another 100 near Chitré, and at least 20 more at Rio Pacora. Another researcher, B. Watts (pers. comm. August 28, 2012), has also surveyed this area, and agrees with an estimate of about 200 wintering *C. canutus*. In spring, Buehler (2002, p. 43) counted 250 to 300 *C. canutus* near Panama City in March and April 2002, although we do not have information to determine if these birds were wintering birds, migrants, or both. *Calidris canutus* has also been observed during summer months in Panama, with the highest counts exceeding 200 birds (Carmona *et al.* 2013, p. 174). During aerial surveys of Panama Bay in the fall of 1997, Watts (1998, p. 11) documented a peak count of 2,460 *C. canutus* in September. Watts (1998, p. 38) reported that these fall *C. canutus* counts in Panama were likely underestimates. Based on data presented above (Subspecies Nonbreeding Distributions), we conclude that *C.c. roselaari* and *C.c. rufa* both likely occur in Panama at least in winter, although we do not have information to determine if the two subspecies utilize the same habitats at the same time within Panama (D. Newstead pers. comm. May 13, 2014), nor can we determine the relative abundance of the two subspecies in Panama or the subspecies composition of migrants moving through Panama.

Most sources have considered *Calidris canutus* a migrant along the Pacific coast of South America (Carmona *et al.* 2013, p. 175). Although Morrison and Ross (1989) did not observe *C. canutus* along the Pacific coast of South America during aerial surveys conducted in January 1985 and February 1986, they observed substantial numbers of unidentified medium-sized shorebirds in several locations, including some that now show *C. canutus* eBird records (Carmona *et al.* 2013, p. 175). Carmona *et al.* (2013, p. 172) found that, of 95 records of *Calidris canutus* from the Pacific coast of South America, 63 were recorded during winter (November to February) and 32 during migration periods. Only 3 records exceeding 20 birds were reported during migration (Paraiso, Peru, and 2 counts at Bahía de Caulín, Chile), but none exceeded 150 birds. In contrast, there were 23 counts of 20 or more *C. canutus* from 5 sites during winter, none of which exceeded 150 birds: Manglares de San Pedro de Vice, Peru; and Putemún, Bahía de Caulín, Bahía Pullao, and Estación Esturial de Quempillén, Chile (Carmona *et al.* 2013, p. 172).

Ruiz-Guerra (2011, p. 194) reported that *C. canutus* winters and is a regular migrant on the Pacific coast of Colombia, mainly in intertidal habitats in the Iscuandé River Delta. Ruiz-Guerra (2011, p. 194) observed *C. canutus* in Pacific Colombia on at least 12 dates from 2007 to 2011 with the following high counts: 30 in autumn, 31 in winter, 16 in spring (Carmona *et al.* 2013, p. 175). Hughes (1979, pp. 51–52) reported 82 *C. canutus* at Mollendo, Peru, in 1971 and 3 in 1975, and described this species as "exceptional" (i.e., very rare) on the south coast of Peru. Current data from eBird.org (2014) show 59 winter records from Los Lagos Chile, mainly on Chiloé Island, with an average of 37 birds per record and a high count of 150 birds. A typical range of 100 to 300 may be likely for Chiloé Island, though the record high spring peak may be about 1,000 (B. Andres pers. comm. July 21, 2014). Based on data presented above (Subspecies Nonbreeding Distributions), we conclude that at least some of the *C. canutus* wintering on Chiloé Island, Chile are *C.c. rufa*. However, we cannot determine if *C.c. roselaari* also use this area (i.e., if the two subspecies mix on Chiloé Island in winter), nor can we ascertain the subspecies composition of *C. canutus* that occur farther north along the Pacific coasts of Chile, Peru, Ecuador, or Colombia.

Summary—Population Surveys and Estimates

We have carefully reviewed available survey data from areas regularly used by substantial numbers of red knots in spring, fall, and winter. For some areas, available data are insufficient to substantiate any conclusions regarding trends over time. For other areas, there are apparent trends, but associated with relatively low confidence. For a few key areas, the consistency of geographic coverage, methodologies, and surveyors lead us to greater confidence in apparent trends.

- Patagonia and Tierra del Fuego wintering region (table 3) – pronounced declines through the 2000s, possibly stabilizing at a relatively low level since 2008, associated with high confidence.

- North-Central Brazil wintering region (table 4 and figure 5) – no apparent trend, given the very high count in 2013 but substantially lower counts in 2011 and 2007. This conclusion is associated with lower confidence due to the complexity of the shoreline that makes surveying difficult (Niles *et al.* 2008, p. 134), the variability of tidal conditions (G. Morrison pers. comm. July 30, 2014), and the availability of only a few data points.

- Northwest Gulf of Mexico wintering region – insufficient data for trend analysis, but anecdotal reports, localized surveys, and limited available data sets suggest that this population may have declined.

- Southeast wintering region – on Florida’s Gulf coast, an apparent decline comparing aerial surveys from 1980 to 1982 with similar surveys (using different surveyors) of approximately the same area from 2006 to 2010, but apparently stable comparing the 2006 to 2010 aerial counts with 1994 ground surveys. Any decline on Florida’s Gulf coast may reflect birds simply shifting elsewhere within this large wintering region. The two regionwide survey efforts to date (from the 2006 and 2011 piping plover surveys) are associated with lower confidence inherent in the methodology (red knots are not the focus of this survey), but do tend to support the perception that knots shift from state to state within this region among years. A long-term data set from Georgia, showing wide inter-annual fluctuations, also supports this perception that birds shift within the Southeast wintering region. Considering the Southeast as a whole, a 2013 winter population estimate based on resightings data (of birds moving through Georgia) suggest this region may have been stable since the early 1980s, but confidence in this conclusion is low because it relies on only two data points derived from dissimilar methodologies and geographic coverage. Data from the Caribbean are insufficient to infer any trends.

- South American spring stopover sites – apparent declines at three key stopover sites from the late 1990s through the mid-2000s, associated with moderate confidence because we have little information regarding the consistency of methodologies or surveyors and because no data are available after 2005.

- Southeast coasts spring window survey – apparent increase from 2010 to 2012, but associated with lower confidence because, despite improvements, methodology and geographic coverage were still being refined and because only 3 years of (relatively consistent) data are available.

- Virginia barrier islands spring stopover area – no apparent trend based on aerial surveys since 1995, associated with high confidence. However, accumulated use ("total bird days") for the season is significantly different ($p < 0.05$) between 1990s and 2000s, showing a decline of about 25 percent (B. Watts pers. comm. August 22, 2014; Watts 2014). A newer data set based on ground surveys suggested an increase from 2007 to 2012, but decreased sharply in 2013, although the decrease may have been partly caused by shift in foraging habitat rather than a true decline in this year.

- Delaware Bay spring stopover area – highly variable data set showing possible declines in the 1990s, and more consistent and substantial declines through the mid-2000s, associated with high confidence during the core years of 1986 to 2008. Numbers appear to have stabilized or increased slightly from 2009 to 2014, despite our lower confidence in the data over this later period due to multiple shifts in methodology and surveyors.

- Fall stopover areas – insufficient data for trend analysis in most areas. Since the 1970s, there were probable declines in some parts of eastern Canada and changes in red knot usage of Massachusetts (mainland versus Cape Cod, proportion of birds bound for Southeast versus Argentina-Chile wintering destinations).

- Inland areas spring and fall – insufficient data for trend analysis.

- Central America and Pacific South America year round - insufficient data for trend analysis.

In conclusion, we have high confidence in two data sets from key red knot areas, Tierra del Fuego and Delaware Bay, showing declines over roughly the same period. Data sets associated with lower confidence from three South American spring stopovers and eastern Canada in fall also suggest declines roughly over this same timeframe. We conclude that red knot use of the Virginia spring stopover declined during this period (the 2000s). We do not conclude that the Southeast wintering region declined, due to the likelihood that knot usage shifted geographically within this region from year to year.

In summary, our analysis of the best available data concludes that an overall, sustained decline of red knot numbers occurred at Tierra del Fuego and Delaware Bay in the 2000s, and that these red knot populations may have stabilized at a relatively low level in the last few years. Although we lack sufficiently robust data to conclude if other wintering and stopover areas also declined, we conclude it is likely that declines at Tierra del Fuego and Delaware Bay drove an overall population decline (i.e., lower total numbers) because these two sites supported a large majority of rangewide knots during the baseline 1980s period. This conclusion is consistent with other analyses of red knot population trends. For example, the NPS (2013) evaluated vetted eBird data points where birding effort was reported, in order to assess gross trends in occurrence of red knots across NPS-managed lands. In the six NPS units where most red knot occurrences were reported, which range from Massachusetts to Florida, a clear declining trend in red knot observations was detected since 1980 (NPS 2013). Likewise, inferring long-term population trends from various national or regional datasets derived from volunteer shorebird surveys and other sources, Andres (2009, p. 6) and Morrison *et al.* (2006, pp. 71, 76, 77) also concluded that red knot numbers declined, probably sharply, in recent decades.

CLIMATE CHANGE

BACKGROUND

Our analyses under the Act include consideration of ongoing and projected changes in climate. The terms "climate" and "climate change" are defined by the Intergovernmental Panel on Climate Change (IPCC). "Climate" refers to the mean and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements, although shorter or longer periods also may be used (IPCC 2007a, p. 78). The term "climate change" thus refers to a change in the mean or variability of one or more measures of climate (e.g., temperature or precipitation) that persists for an extended period, typically decades or longer, whether the change is due to natural variability, human activity, or both (IPCC 2007a, p. 78).

Scientific measurements spanning several decades demonstrate that changes in climate are occurring, and that the rate of change has increased since the 1950s. Based on extensive analyses of global average surface air temperature, the most widely used measure of change, the IPCC concluded that warming of the global climate system over the past several decades is unequivocal (IPCC 2013a, p. 4; IPCC 2007a, p. 2) and accelerating (IPCC 2007b, p. 104). In addition to rising air temperatures, substantial regional increases or decreases in precipitation, shifts in the ranges of plant and animal species, increasing ground instability in permafrost regions, increasing acidification of the oceans, conditions more favorable to the spread of invasive species and of some diseases, and changes in amount and timing of water availability are occurring in association with changes in climate (U.S. Global Change Research Program (USGCRP) 2009, pp. 27, 79–88; IPCC 2007a, pp. 2–4, 9, 30–33; Solomon *et al*. in IPCC 2007b, pp. 35–54, 82–85).

Results of scientific analyses presented by the IPCC show that most of the observed increase in global average temperature since the mid-20th century cannot be explained by natural variability in climate, and is "very likely" (see table 14) due to the observed increase in greenhouse gas (GHG) concentrations in the atmosphere as a result of human activities, particularly carbon dioxide emissions from fossil fuel use (IPCC 2013a, p. 13; IPCC 2007a, pp. 5–6 and figures SPM.3 and SPM.4; Solomon *et al*. in IPCC 2007b, pp. 21–35). Further confirmation of the role of GHGs comes from analyses by Huber and Knutti (2011, p. 4), who concluded it is "extremely likely" that approximately 75 percent of global warming since 1950 has been caused by human activities.

Scientists use a variety of climate models, which include consideration of natural processes and variability, as well as various scenarios of potential levels and timing of GHG emissions, to evaluate the causes of changes already observed and to project future changes in temperature and other climate conditions (e.g., IPCC 2013a, pp. 743–745; Prinn *et al*. 2011, pp. 527, 529; Ganguly *et al*. 2009, pp. 11555, 15558; Meehl *et al*. in IPCC 2007b, pp. 749–782). All combinations of models and emissions scenarios yield very similar projections of average global warming until about 2030. Although projections of the magnitude and rate of warming differ after about 2030, the overall trajectory of all the projections is one of increased global warming through the end of the 21st century, even for projections based on scenarios that assume that

GHG emissions will stabilize or decline. Thus, there is strong scientific support for projections that warming will continue through the 21st century, and that the magnitude and rate of change will be influenced substantially by the extent of GHG emissions (IPCC 2013a, pp. 955–957, 1031–1033; Prinn *et al*. 2011, pp. 527, 529; Ganguly *et al*. 2009, pp. 15555–15558; IPCC 2007a, pp. 44–45; Meehl *et al*. in IPCC 2007b, pp. 760–764).

Table 14. Standard terms used by the IPCC to define levels of confidence and likelihood regarding climate change (Solomon *et al*. in IPCC 2007b, pp. 22–23). When used in this context, these terms are given in quotes in this document.

Confidence Terminology	**Degree of Confidence in Being Correct**
Very high confidence	At least 9 out of 10 chance
High confidence	About 8 out of 10 chance
Medium confidence	About 5 out of 10 chance
Low confidence	About 2 out of 10 chance
Very low confidence	Less than 1 out of 10 chance
Likelihood Terminology	**Likelihood of the occurrence or outcome**
Virtually certain	greater than 99 percent probability
Extremely likely	greater than 95 percent probability
Very likely	greater than 90 percent probability
Likely	greater than 66 percent probability
More likely than not	greater than 50 percent probability
About as likely as not	33 to 66 percent probability
Unlikely	less than 33 percent probability
Very unlikely	less than 10 percent probability
Extremely unlikely	less than 5 percent probability
Exceptionally unlikely	less than 1 percent probability

In addition to basing their projections on scientific analyses, the IPCC reports projections using a framework for treatment of uncertainties (table 14). Some of the IPCC's key projections of global climate and its related effects through 2100 include: (1) it is "virtually certain" that there will be warmer and more frequent hot days and nights over most of the earth's land areas; (2) it is "very likely" that there will be increased frequency or duration of warm spells and heat waves over most land areas; (3) it is "very likely" that the frequency of heavy precipitation events, or the proportion of total rainfall from heavy falls, will increase over most areas; and (4) it is "likely" that the intensity or duration of droughts will increase; (5) it is "more likely than not" that intense tropical cyclone activity will increase in the North Atlantic; and (6) it is "very likely" that there will be increased incidence of extreme high sea level (IPCC 2013a, p. 7). The IPCC (2013a, p. 27) also projects a global increase in ocean acidification (see Factor E—Reduced Food Availability) under all GHG emissions scenarios. Thus, there is a high degree of certainty regarding the overall trajectory of climate changes over the next few decades.

However, the possibility of abrupt changes in the climate or other environmental systems adds uncertainty to more specific projections about how the effects of climate change are likely to unfold. The Earth's temperature is now demonstrably higher than it has been for several hundred years, and GHG concentrations are now higher than they have been in at least 800,000 years. These sharp departures from historical climate regimes raise the possibility that "tipping points" or thresholds for stability might be crossed as the climate system warms, leading to rapid or abrupt changes in climate (National Research Council (NRC) 2010, p. 41). Likewise, with increasing warming, some physical systems or ecosystems may be at risk of abrupt and irreversible changes (i.e., passing tipping points or thresholds) (Summary for Policymakers in IPCC 2014, p. 12). Several components or phenomena in the climate system could potentially exhibit abrupt or nonlinear changes, and some are known to have done so in the geologic past (IPCC 2013a, p. 1033; NRC 2013, p. 1). Paleoclimate records indicate that the climate system can experience abrupt changes in as little as a decade (NRC 2010a, p. 41). Arctic sea ice is one component of the climate system that may already be undergoing an abrupt change (NRC 2013, p. 1). However, there is "low confidence" and little consensus on the likelihood of abrupt climate change (i.e., crossing thresholds) over the 21st century (IPCC 2013a, p. 1033). In general, we have only a limited understanding of where tipping points may exist, when they might be crossed, or what the consequences might be (NRC 2013, p. 1; NRC 2010a, p. 42).

Over recent decades, temperatures have increased about twice as fast in the Arctic as in the middle latitudes, a phenomenon known as "polar amplification" or "Arctic amplification" (IPCC 2013a, p. 398; National Aeronautics and Space Administration (NASA) 2013; NRC 2011, p. 9). "High confidence" exists for polar amplification based on robust and consistent evidence from temperature reconstructions of past climates, recent instrumental temperature records, and climate model simulations of past, present and future climate changes (IPCC 2013a, p. 398). Polar amplification is of global concern due to the potential effects of future warming on ice sheet stability and, therefore, global sea level and carbon cycle feedbacks such as those linked with permafrost melting (IPCC 2013a, p. 396). (Also see Coastal Storms and Extreme Weather, below.)

Global climate projections are informative, and, in some cases, the only or the best scientific information available. However, projected changes in climate and related impacts can vary substantially across and within different regions of the world (IPCC 2007a, pp. 8–12). Therefore, we use "downscaled" projections when they are available and have been developed through appropriate scientific procedures because such projections provide higher resolution information that is more relevant to the spatial scales used for species analyses (see Glick *et al.* 2011, pp. 58–61, for a discussion of downscaling).

EFFECTS ON SPECIES

Various changes in climate may have direct or indirect effects on species. These effects may be positive, neutral, or negative, and they may change over time, depending on the species and other considerations, such as the interactions of climate with other variables such as habitat fragmentation (for examples, see Chen *et al*. 2011, entire; Forister *et al*. 2010, entire; Galbraith *et al*. 2010, entire; IPCC 2007a, pp. 8–14, 18–19; Franco *et al.* 2006, entire). In addition to considering individual species, scientists are evaluating possible climate change-related impacts to, and responses of, ecological systems, habitat conditions, and groups of species; these studies include acknowledgement of uncertainty (e.g., Galbraith *et al.* 2014, entire; Fraser *et al.* 2013, entire; Schmidt *et al.* 2012, p. 4421; Beaumont *et al*. 2011, entire; Hale *et al.* 2011, entire; McKelvey *et al*. 2011, entire; Rogers and Schindler 2011, entire; Berg *et al*. 2010, entire;

Sinervo *et al*. 2010, entire; Euskirchen *et al.* 2009, entire; McKechnie and Wolf 2009, entire; Deutsch *et al*. 2008, entire; Ims and Fuglei 2005, entire; Lindström and Agrell, entire).

Many analyses involve climate change vulnerability assessments. In relation to climate change, vulnerability refers to the degree to which a species (or system) is susceptible to, and unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the type, magnitude, and rate of climate change and variation to which a species is exposed, the species' sensitivity, and its adaptive capacity (Glick *et al.* 2011, pp. 19–22; IPCC 2007a, p. 89). There is no single method for conducting such analyses that applies to all situations (Glick *et al.* 2011, p. 3). We use our expert judgment and appropriate analytical approaches to weigh relevant information, including uncertainty, in our consideration of various aspects of climate change.

Projecting the responses of species and ecosystems to climate change is complicated by the likelihood of thresholds (or "tipping points") being crossed and feedback mechanisms operating (U.S. Climate Change Science Program (CCSP) 2009a, pp. 1–2). The identification and prediction of thresholds in natural systems presents one of the greatest challenges facing scientists investigating climatic and environmental change, since the intrinsic properties can be nonlinear and abrupt (NRC 2011, p. 9). In addition to the possibility of abrupt changes within the climate system itself (discussed above), even gradual climate changes can result in crossing thresholds in natural systems (NRC 2013, p. 1). An ecological threshold is the point at which there is an abrupt change in an ecosystem quality, property, or phenomenon, or at which a small change in one or more external conditions produces a large and persistent response in an ecosystem. Ecological thresholds occur when external factors, positive feedbacks, or nonlinear instabilities in a system cause changes to propagate in a domino-like fashion that are potentially irreversible. Once an ecological threshold is crossed, the ecosystem in question is not likely to return to its previous state. Positive feedbacks are those that tend to increase alteration of the nature of the system, while negative feedbacks tend to minimize these changes. Ecosystems include both positive and negative feedbacks (CCSP 2009a, pp. 1–2). Risks associated with tipping points become moderate at even low levels of additional warming, due to early warning signs that that both warm-water coral reef and arctic ecosystems are already experiencing irreversible regime shifts ("medium confidence") (Summary for Policymakers in IPCC 2014, p. 12).

Globally, there are several ecosystems for which conditions suggest an approaching climate-related threshold, including the arctic tundra, coral reefs, prairie pothole wetlands, and southwestern forests. In the arctic tundra, for example, a series of positive feedback mechanisms may trigger a relatively sudden, domino-like chain of events that result in conversion from low tundra vegetation to shrubland, initiated by a relatively slight increase in temperature (CCSP 2009a, pp. 1–2). In the polar regions, there is considerable risk of passing thresholds and tipping points caused by the rapid response of the cryosphere (those portions of Earth's surface where water is in solid form) (NRC 2011, p. 9). (See Factor A—Breeding Habitat, and Factor E—Asynchronies—Breeding Grounds .) The potential to cross thresholds of rapid change is also of particular concern for coastal ecosystems. Beyond these tipping points, coastal ecosystems exist in a dramatically altered state or are lost entirely from the area, and, in some cases, these changes will be irreversible (Melillo *et al.* 2014, p. 592). Risks increase disproportionately as warming increases, due to the potential for a large and irreversible sea level rise from ice sheet loss

(Summary for Policymakers in IPCC 2014, p. 12). (See Factor A—Sea Level Rise.) As is the case with all threats that we assess, even if we conclude that a species is affected or is likely to be affected in a negative way by one or more climate-related impacts, it does not necessarily follow that the species meets the definition of endangered species or threatened species under the Act. If a species is listed as endangered or threatened, knowledge regarding its vulnerability to, and known or anticipated impacts from, climate-associated changes in environmental conditions can be used to help devise appropriate strategies for its recovery.

COASTAL STORMS AND EXTREME WEATHER

Several threats to the red knot are related to the possibility of changing storm and weather patterns. While variation in weather is a natural occurrence and is normally not considered a threat to the survival of a species, persistent changes in the frequency, intensity, or timing of storms at key locations where red knots congregate can pose a threat (see Factor E). Extreme weather events have been implicated as mechanistic drivers of broad ecological responses to climatic trends (e.g., the mechanism by which climate change that actually bring about species' varying responses, such as range shifts, through changes in factors such as abundance, morphology, behavior, reproduction) (Parmesan *et al.* 2000, entire).

The IPCC (2012) produced a summary report regarding global trends and predictions for extreme events including storms. There is "low confidence" in any observed long-term (i.e., over the past 40+ years) increases in tropical cyclone (e.g., hurricane) activity (i.e., intensity, frequency, duration), after accounting for past changes in observing capabilities (IPCC 2012, p. 8). The North Atlantic illustrates the difficulty of gaging past trends in storm activity. (Regarding tropical storms, "north" is used in the sense of above the equator.) Holland and Webster (2007, p. 2697) and Mann and Emanuel (2006, p. 238) found increasing trends in tropical cyclone activity in the North Atlantic basin extending back to 1900 and 1880, respectively (National Oceanic and Atmospheric Administration (NOAA) 2013a, p. 39). However, assessing trends in storm frequency over the 20th century is confounded by increasing storm detection rates brought on by technological advances, beginning with aircraft in the mid-1940s and increasing further with satellites in the late 1960s (NOAA 2013a, pp. 39–40; Landsea 2007, p. 197). When adjusted for these reporting and monitoring biases, the time series of Atlantic basin tropical cyclone frequency shows only a slight upward trend from 1878 to 2008 (NOAA 2013a, p. 40; Landsea *et al.* 2010, p. 2508).

Looking only at the satellite era, 1970 to 2004, however, Webster *et al.* (2005, pp. 1845–1846) found that the North Atlantic from 5º to 25º north latitude (northern South America to the Florida Keys) showed an increasing trend in hurricane frequency and duration that is significant at the 99 percent confidence level, but these authors concluded that the role of global climate change in these patterns is still unclear. Holland and Bruyère (2013, p. 1) found that the bulk of the current anthropogenic warming has occurred in the past four decades, within the satellite era, which enables improved confidence in assessing hurricane changes as it removes many of the data issues from previous eras. These authors found no anthropogenic contribution to changes in global hurricane frequencies, but found strong evidence of a human contribution (e.g., from GHG emissions) to changing hurricane strengths (e.g., changing proportions of both weaker and stronger hurricanes). Specifically, the proportion of Category 4 and 5 hurricanes has increased at a rate of 25 to 30 percent for every 1.8ºF (1 ºC) of global warming after accounting for analysis and observing system changes. This has been balanced by a similar decrease in Category 1 and 2 hurricane proportions (Holland and Bruyère 2013, p. 1). These findings are consistent with several other studies that found increasing frequencies of high-intensity tropical storms (e.g., Category 4 and 5 hurricanes), as well as increases in the "accumulated cyclone energy index," a metric that incorporates cyclone intensity (wind speed) and duration. These increases in the most powerful storms have taken place since the 1970s, and are attributed to improved monitoring technology, multi-decade climate variability, and human-caused global warming (NOAA 2013a, p. 40; Emanuel 2005, p. 686; Webster *et al.* 2005, pp. 1845–1846). The increase in the number and strength of hurricanes has occurred at times and in areas used by red knots (Committee on the Status of Endangered Wildlife in Canada (COSEWIC) 2007, p. 36).

Predictions about future storm patterns are associated with only "low to medium confidence" (IPCC 2012, pp. 8, 13). Average tropical cyclone maximum wind speed is "likely" to increase, although these wind speed increases may not occur in all ocean basins. Heavy rainfalls associated with tropical cyclones are "likely" to increase with continued warming. Globally, it is "likely" that the frequency of tropical cyclones will either decrease or remain essentially unchanged (IPCC 2012, pp. 8, 13), though there may be regional differences in some ocean basins. However, one modeling effort by Emanuel (2013, p. 6) projected an increase in global tropical cycle activity, including in the North Atlantic. Based on best available data, we cannot draw any conclusions regarding future trends in the total number of tropical storms (e.g., all category 1 through 5 hurricanes) within the range of the red knot. We do conclude, however, that the number and strength of high-intensity tropical storms (e.g., category 4 and 5 hurricanes) will probably continue to increase (e.g., Melillo *et al.* 2014, p. 20; Emanuel 2013, p. 6; Knutson *et al.* 2010, p. 157). The IPCC (2013a, p. 7) concluded that increases in intense tropical cyclone activity in the North Atlantic through 2100 are "more likely than not." Notwithstanding these conclusions, the observed increase in Category 4 and 5 hurricanes may not continue at the same rate with future global warming. Following an initial increase in intense hurricane proportions, a saturation level may be reached beyond which any further global warming will have little effect (Holland and Bruyère 2013, p. 1).

The IPCC (2012, pp. 8,13) found it is "likely" that there has been a poleward shift in the main Northern and Southern Hemisphere extra-tropical storm tracks, meaning these storms, on average, are taking place farther from the equator than in the past (IPCC 2012, pp. 8, 13). (In the Northeast United States, a common type of extra-tropical storm is the nor'easter, which is a winter storm characterized by continuously strong northeasterly winds blowing from the ocean). Due to the poleward shift in extra-tropical storms since the 1970s, nor'easters are now more frequent and intense in the New England region of the United States, but less frequent in the mid-Atlantic United States (Frumhoff *et al.* 2007, pp. 30–31). While there is "low confidence" in detailed geographical projections of future extra-tropical cyclone activity, there is "medium confidence" in a continued poleward shift of extra-tropical storm tracks. There is "medium confidence" that there will be a reduction in the number of extra-tropical cyclones averaged over each hemisphere (IPCC 2012, pp. 8, 13). For extra-tropical storms, current research suggests a decrease in the total number of events, but an increase in the number of intense events within the next century (National Fish, Wildlife and Plants Climate Adaptation Strategy 2012, p . 10).

The frequency and intensity of extreme precipitation events (from both coastal and noncoastal storms) are increasing. There have been statistically significant trends in the number of heavy precipitation events in some regions of the world. Although there are strong regional variations, it is “likely” that more of these regions have experienced increases than decreases in the number of heavy precipitation events, and there is “medium confidence” that anthropogenic (human caused) influences have contributed to intensification of extreme precipitation at the global scale (IPCC 2012, pp. 8–9). Across most of the United States, the heaviest rainfall events have become heavier and more frequent. Since 1991, the amount of rain falling in very heavy precipitation events has been more than 30 percent above the average from 1901 to 1960. This increase has been greatest in the Northeast, Midwest, and upper Great Plains (Melillo *et al.* 2014, p. 36). It is “likely” that the frequency of heavy precipitation, or the proportion of total rainfall from heavy falls, will increase in the 21st century over many areas of the globe. This is particularly the case in the high latitudes and tropical regions, and in winter in the northern mid-latitudes. There is “medium confidence” that, in some regions, increases in heavy precipitation will occur despite projected decreases in total precipitation in those regions. Based on a range of emissions scenarios, a 1-in-20 year annual maximum daily precipitation amount is likely to become a 1-in-5 to 1-in-15 year event by the end of the 21st century in many regions (IPCC 2012, p. 13). Projections of future U.S. climate suggest that the recent trend towards increased heavy precipitation events will continue (Melillo *et al.* 2014, p. 37). In the Northeast United States, for example, increases in precipitation intensity of 8 to 9 percent are projected by mid-century, and 10 to 15 percent by the end of the century. The number of heavy precipitation events is projected to increase 8 percent by mid-century and 12 to 13 percent by the end of the century (Frumhoff *et al.* 2007, p. 8). However, there is “low confidence” in projections of small spatial-scale weather phenomena (IPCC 2012, p. 13) (i.e., we cannot anticipate local weather patterns).

New studies are linking Arctic amplification with weather changes in North America, brought about by changes in atmospheric circulation patterns (e.g., changes in the speed and “waviness” of the jet stream). Overland *et al.* (2012, pp. 1, 6) found changes in early summer Arctic wind patterns from 2007 to 2012 relative to previous decades, and implicated these arctic changes in the recent increases in the initiation, persistence, and severity of weather extremes at lower latitudes of North America. Observational analysis by Francis and Vavrus (2012, p. 1) suggested that rapid climate change in the Arctic could lead to increased probabilities of extreme weather events (e.g., droughts, flood, cold spells, heat waves) in the middle latitudes of the Northern Hemisphere. Some researchers have found evidence that Arctic amplification contributed to the unusual conditions surrounding Hurricane Sandy (Greene *et al.* 2013, entire; Eaton 2012). Petoukhov *et al.* (2013, entire) developed equations that describe atmospheric wave motions in the middle latitudes. These authors found that certain types of waves have become trapped and amplified more frequently since 1980 and that this phenomenon is linked to extreme weather events around the world, such as regional heat waves and floods. The increase in these specific atmospheric patterns is associated with rapid warming in the Arctic (Potsdam Institute for Climate Impact Research 2013). The 32-year period studied by Petoukhov *et al.* (2013) provides a good indication of the mechanism involved in increasing extreme weather events, but is too short for definite conclusions (Potsdam Institute for Climate Impact Research 2013). Coumou *et al.* (2014, p. 1) found similar results that changes in atmospheric waves lead to persistent surface weather conditions and therefore to midlatitude synchronization of extreme heat and rainfall events. These authors found changes in atmospheric circulation since the onset of rapid Arctic amplification around 2000, providing new insights regarding the link between Arctic changes and midlatitude extremes (Coumou *et al.* 2014, p. 1).

However, the scientific link between Arctic amplification and extreme weather at lower latitudes is not yet widely accepted (Wallace *et al.* 2014; Ogburn 2013; Eaton 2012). Barnes (2013, entire) did not find that atmospheric waves were getting wavier, as other research has suggested, and also failed to find strong evidence for a slowdown in the speed of such waves. Modeling by Barnes *et al.* (2013, entire) found that the atmospheric conditions that led to Hurricane Sandy's turn into the New Jersey coast are actually less likely as the climate changes, not more likely. Screen and Simmonds (2013, entire) found some statistically significant trends in atmospheric wave height in some seasons in some places. These authors found that the possible connections among Arctic amplification, atmospheric waves, and middle latitude weather are complex and sensitive to the assumptions that underpin the modeling, and that more research is needed to understand these connections (Screen and Simmonds 2013, p. 959). Many researchers agree that the science on this issue is unsettled because it is a new field of investigation, available data sets are short, and the climate system is highly complex (Ogburn 2013).

ANALYSIS OF LISTING FACTORS

Section 4 of the Act (16 U.S.C. 1533), and its implementing regulations at 50 CFR part 424, set forth the procedures for adding species to the Federal Lists of Endangered and Threatened Wildlife and Plants. Under section 4(a)(1) of the Act, we may list a species based on any of the following five factors: (A) The present or threatened destruction, modification, or curtailment of its habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) the inadequacy of existing regulatory mechanisms; and (E) other natural or manmade factors affecting its continued existence. Listing actions may be warranted based on any of the above threat factors, singly or in combination. Each of these factors is discussed below.

OVERVIEW OF THREATS RELATED TO CLIMATE CHANGE

We discuss the ongoing and projected effects of climate change, and the levels of certainty associated with these effects, in the appropriate sections of the five-factor analysis. For example, habitat loss from sea level rise is discussed under Factor A, and asynchronies ("mismatches") in the timing of the annual cycle are discussed under Factor E. Here we present an overview of threats stemming from climate change, which are addressed in more detail in the sections that follow.

The natural history of Arctic-breeding shorebirds (e.g., many are long-distance migrants, reliant on both coastal and Arctic habitats) makes this group of species particularly vulnerable to global climate change (e.g., Meltofte *et al.* 2007, entire; Piersma and Lindström 2004, entire; Rehfisch and Crick 2003, entire; Piersma and Baker 2000, entire; Zöckler and Lysenko 2000, entire; Lindström and Agrell 1999, entire). Relatively low genetic diversity, which is thought to be a consequence of survival through past climate-driven population bottlenecks, may put shorebirds at more risk from human-induced climate variation than other avian taxa (Meltofte *et al.* 2007, p. 7); low genetic diversity may result in reduced adaptive capacity as well as increased risks when population sizes drop to low levels.

In the short term, red knots may benefit if warmer temperatures result in fewer years of delayed horseshoe crab spawning in Delaware Bay (Smith and Michels 2006, pp. 487–488) or fewer occurrences of late snow melt in the breeding grounds (Meltofte *et al.* 2007, p. 7). However, there are indications that changes in the abundance and quality of red knot prey are already under way (Escudero *et al.* 2012, pp. 359–362; Jones *et al.* 2010, pp. 2255–2256), and prey species face ongoing climate-related threats from warmer temperatures (Jones *et al.* 2010, pp. 2255–2256; Philippart *et al.* 2003, p. 2171; Rehfisch and Crick 2003, p. 88), ocean acidification (International Geosphere-Biosphere Programme (IGBP) *et al.* 2013, p. 16; NRC 2010a, p. 286; Fabry *et al.* 2008, p. 420), and possibly increased prevalence of disease and parasites (Ward and Lafferty 2004, p. 543). In addition, red knots face imminent threats from loss of habitat caused by sea level rise (Galbraith *et al.* 2014, p. 7 and Supplement 1; Iwamura *et al.* 2013, p. 1; NRC 2010a, p. 44; Galbraith *et al.* 2002, pp. 177–178; Titus 1990, p. 66), and increasing asynchronies ("mismatches") between the timing of their annual breeding, migration, and wintering cycles and the windows of peak food availability on which the birds depend (Galbraith *et al.* 2014, p. 7 and Supplement 1; Smith *et al.* 2011a, pp. 575, 581; McGowan *et al.* 2011a, p. 2; Meltofte *et al.* 2007, p. 36; van Gils *et al.* 2005a, p. 2615; Baker *et al.* 2004, p. 878). Except when breeding, most red knots prey entirely on marine species. An extensive literature review found pervasive changes to marine species already observable across various taxa and oceans, as climate change is already having a coherent and significant impact across all ecosystems (coastal to open ocean), latitudes (polar to tropical), and trophic levels (phytoplankton to top predators) (Richardson *et al.* 2012, p. 908).

Several threats are related to the possibility of changing storm patterns. Variation in weather is a natural occurrence and is normally not considered a threat to the survival of a species. For example, while Hurricane Sandy destroyed shorebird habitat in some areas, it actually created shorebird habitat in other areas, through natural coastal processes (B. Maslo pers. comm. March 5, 2014; Niles *et al.* 2012b, p. 1). However, persistent changes in the frequency, intensity, or timing of storms at key locations where red knots congregate can pose a threat (see Factor E and "Coastal Storms and Extreme Weather" under Climate Change). Storms impact migratory shorebirds like the red knot both directly and indirectly. Direct impacts include energetic costs from a longer migration route as birds avoid storms, blowing birds off course, and outright mortality (Niles *et al*. 2010a, p. 129). Indirect impacts include changes to habitat suitability, storm-induced asynchronies between migration stopover periods and the times of peak prey availability, and possible prompting of birds to take refuge in areas where shorebird hunting is still practiced (Niles *et al.* 2012b, p. 1; Dey *et al*. 2011b, pp. 1–2; Nebel 2011, p. 217; see Factor B).

With arctic warming, vegetation conditions in the red knot's breeding grounds are changing, which is expected to eventually cause the zone of nesting habitat to shift and contract. Although vegetative change may take decades to unfold (Feng *et al*. 2012, p. 1366; Meltofte *et al.* 2007, p. 36; Kaplan *et al.* 2003, p. 10), the red knot's arctic breeding grounds also have the potential for a series of positive feedback mechanisms to trigger a relatively sudden, domino-like chain of events that result in conversion from low tundra vegetation to shrubland, initiated by a relatively slight increase in temperature (CCSP 2009a, pp. 1–2). Ecological shifts in the Arctic may appear sooner. Because of their simplicity, arctic food webs may especially be prone to exhibit nonlinear dynamics in response to climate warming and to show abrupt changes due to threshold effects and feedback processes (Gauthier *et al*. 2013, p. 10) (see Climate Change—

Background). High uncertainty exists about when and how changing interactions among vegetation, predators, competitors, prey, parasites, and pathogens may affect the red knot, but the extent of ecosystem change is likely profound (Fraser *et al.* 2013, entire; Gauthier *et al.* 2013, p. 10; Olofsson *et al.* 2013, entire; Schmidt *et al.* 2012, p. 4421; NRC 2011, pp. 1, 44; Meltofte *et al.* 2007, p. 35; Ims and Fuglei 2005, entire; Lindström and Agrell 1999, entire). Further, certainty is high that arctic ecosystem changes are already underway and will continue (Summary for Policymakers in IPCC 2014, pp. 12, 32; Chapter 28 in IPCC 2014, p. 3); see Factor A, Factor C, Factor E, and Cumulative Effects.

Recent assessments of the red knot's vulnerability to climate change indicated a large increase in extinction risk due to the likely loss of breeding (from arctic warming) and nonbreeding habitat (from sea level rise), as well as the knot's high degree of habitat specialization, long migration distance, and high degree of dependence on ecological synchronicities (Galbraith *et al.* 2014, p. 7 and Supplement 1; National Wildlife Federation (NWF) 2013, p. 28; Whitman *et al.* 2013, pp. 2, 19, 64). As discussed throughout Factors A and E, below, climate change is expected to affect red knot fitness and, therefore, survival through direct and indirect effects on breeding and nonbreeding habitat, food availability, and timing of the birds' annual cycle. Ecosystem changes in the arctic (e.g., changes in predation patterns and pressures) may also reduce reproductive output. Together, these anticipated changes will likely negatively influence the long-term survival of the rufa red knot.

FACTOR A. PRESENT OR THREATENED DESTRUCTION, MODIFICATION, OR CURTAILMENT OF ITS HABITAT OR RANGE

In this section, we present and assess the best available scientific and commercial data regarding ongoing threats to the quantity and quality of red knot habitat. Within the nonbreeding portion of the range, red knot habitat is primarily threatened by the highly interrelated effects of sea level rise, shoreline stabilization, and coastal development. Lesser threats to nonbreeding habitat include agriculture and aquaculture, invasive vegetation, and beach maintenance activities. Within the breeding portion of the range, the primary threat to red knot habitat is from climate change. With arctic warming, vegetation conditions in the breeding grounds are changing, which is expected to eventually cause the zone of nesting habitat to shift and contract. Arctic freshwater systems—foraging areas for red knots during the nesting season—are particularly sensitive to climate change, and fundamental ecosystem changes are expected to continue across the Arctic.

Factor A—Accelerating Sea Level Rise

For most of the year, red knots live in or immediately adjacent to intertidal areas. These habitats are naturally dynamic, as shorelines are continually reshaped by tides, currents, wind, and storms. Coastal habitats are susceptible to both abrupt (storm-related) and long-term (sea level rise) changes. Sea level rise stemming from climate change will greatly alter littoral ecosystems, causing habitat change and loss for coastal species (Iwamura *et al.* 2013, p. 1). Outside of the breeding grounds, red knots rely almost entirely on these coastal areas to fulfill their roosting and foraging needs, making the birds vulnerable to the effects of habitat loss from rising sea levels. Because conditions in coastal habitats are also critical for building up nutrient and energy stores for the long migration to the breeding grounds, sea level rise affecting conditions on staging areas also has the potential to impact the red knot's ability to breed successfully in the Arctic (Meltofte *et al.* 2007, p. 36).

In the sections that follow, we evaluate the potential for the loss of red knot habitat from sea level rise. For migratory species, however, the impact of habitat loss depends not only on its extent, but also on where it occurs (Iwamura *et al.* 2013, p. 1). Modeling 10 long-distance migrant shorebirds (including *Calidris canutus rogersi* and *C.c. piersmai*) using the East Asian–Australasian Flyway, Iwamura *et al.* (2013, pp. 1–2) found that reductions in population flow in these migratory networks far exceeded the proportion of habitat lost from sea level rise. Iwamura *et al.* (2013, pp. 1, 3, 4) estimated that sea level rise will inundate 23 to 40 percent of intertidal habitat area along these migration routes, but cause a reduction in population flow of up to 72 percent across the species evaluated in this study. Declines in population flow exceeded the proportional declines in habitat extent at all sea level rise scenarios (from 1.6 to 9.8 ft (0.5 to 3 m)) and across all taxa. In an optimistic scenario, where an upshore shift (i.e., inland migration) of intertidal habitat into all nonurban areas was assumed, the rate of predicted habitat loss was, unsurprisingly, smaller for all sea level scenarios; however, the declines in population flow always remained higher than those predicted by habitat loss alone. The disproportional population effects (relative to habitat loss) were particularly strong for taxa whose migration routes contain bottlenecks—sites through which a large fraction of the population travels (Iwamura *et al.* 2013, p. 1). With an estimated 50 to 80 percent of the total population using Delaware Bay each spring (Brown *et al.* 2001, p. 10), rufa red knots exhibit such bottlenecking (Buehler and Piersma 2008, p. 252). The results of Iwamura *et al.* (2013, pp. 1, 6) emphasize the importance of incorporating migratory connectivity into estimates of habitat loss impacts, and indicate that migratory shorebirds are at greater risk from sea level rise than previously realized.

Sea Level Rise—Rates

The rate of sea level rise since the mid-19th century has been faster than the mean rate during the previous two millennia ("high confidence"). Over the period 1901 to 2010, global mean sea level rose by 0.56 to 0.69 ft (0.17 to 0.21 m) (IPCC 2013a, p. 11). The rate of global sea level rise has accelerated since the mid-19th century (NRC 2010a, p. 43). It is "very likely" that the mean rate of global averaged sea level rise was 0.06 to 0.07 in (1.5 to 1.9 mm) per year between 1901 and 2010; 0.07 to 0.09 in (1.7 to 2.3 mm) per year between 1971 and 2010; and 0.11 to 0.14 in (2.8 to 3.6 mm) per year between 1993 and 2010 (IPCC 2013a, p. 11).

Sea levels will continue to rise during the 21st century. Under all GHG emissions scenarios, the rate of sea level rise will "very likely" exceed that observed between 1971 and 2010 (IPCC 2013a, p. 25). While there is widespread agreement that the rate of sea level rise will continue to increase during the 21st century, great uncertainty surrounds its future magnitude (Horton *et al.* 2014, p. 1). The IPCC (2013a, p. 25) projects additional (relative to the period 1986 to 2005) global mean sea level rise of 0.85 to 3.22 ft (0.26 to 0.98 m) by 2100 with "medium confidence." However, projected rates of sea level rise remain rather uncertain, due mainly to limits in scientific understanding of glacier and ice sheet dynamics (e.g., rates and patterns of ice growth versus loss) (Horton *et al.* 2014, entire; Parris *et al.* 2012, p. 2; NRC 2010a, p. 44; Pfeffer *et al.* 2008, p. 1342). In recent decades, the dominant contributors to global sea level rise have been ocean warming (i.e., thermal expansion) and ice sheet loss (i.e., melting) (Parris *et al.* 2012, p. 1). The relative magnitude of each of these factors in the future remains highly uncertain. Many studies, reflected by the IPCC, assume thermal expansion to be the dominant contributor. However, advances in satellite measurements indicate ice sheet loss as a greater contribution to global sea level rise than thermal expansion over the period of 1993 to 2008 (Parris *et al.* 2012, pp. 1–2). Yet, the greatest uncertainty surrounding estimates of future global sea level rise continues to be the rate and magnitude of ice sheet loss, primarily from Greenland and West Antarctica (Parris *et al.* 2012, p. 2) (e.g., see Mengel and Levermann 2014, entire; McMillan *et al.* 2014, entire; Morlighem *et al.* 2014, entire; Rignot *et al.* 2014, entire). Some research suggests that sea levels could potentially rise another 2.5 to 6.5 ft (0.8 to 2 m) by 2100, which is roughly twice as large as the IPCC estimates (Melillo *et al.* 2014, p. 45; Horton *et al.* 2012; NRC 2010a, p. 44; Vermeer and Rahmstorf 2009, p. 21527; Pfeffer *et al.* 2008, p. 1340).

The IPCC (2013a, p. 26) concluded there is "low confidence" in sea level rise projections over 3.3 ft (1 m). However, for the most recent National Climate Assessment (Melillo *et al.* 2014), Parris *et al.* (2012, p. 2) evaluated various sea level rise scenarios that reflect different degrees of ocean warming and ice sheet loss and have "very high confidence" that global mean sea level will be between 0.7 and 6.6 ft (0.2 and 2.0 m) by 2100, which is the range we consider in the sections that follow. The conclusions of this National Climate Assessment include a realistic low end of about 1 ft (0.3 m), a plausible high end of about 4 ft (1.2 m), and some risk of up to 6.6 ft (2.0 m) by 2100 (Melillo *et al.* 2014, p. 45). A 2014 survey of 90 sea level rise experts produced a median "likely" range of 2.0 to 3.3 ft (0.6 to 1.0 m) by 2100 under a low emissions scenario, and 2.3 to 3.9 ft (0.7 to 1.2 m) with higher GHG emissions (Horton *et al.* 2014, p. 1). This survey reflects the substantial uncertainty that remains around future sea level rise projections, with 13 of the experts estimating a 17 percent probability of exceeding a 6.6 ft (2.0 m) sea level rise by 2100 under high emissions (Horton *et al.* 2014, p. 5). Risks associated with climatic "tipping points" increase disproportionately as temperature increases, due to the potential for a large and irreversible sea level rise from ice sheet loss (Summary for Policymakers in IPCC 2014, p. 12).

The amount of sea level change varies regionally because of different rates of subsidence (settling or "sinking") or uplift of the land, and because of differences in ocean circulation (NRC 2010a, p. 43). The net effect of these different factors is known as relative (in contrast to global) sea level. In the last century, for example, relative sea level rise along the U.S. mid-Atlantic and Gulf coasts exceeded the global average by 5 to 6 in (13 to 15 cm) because coastal lands in these areas are subsiding (USEPA 2013b). Land subsidence also occurs in some areas of the Northeast, at current rates of 0.02 to 0.04 in (0.5 to 1 mm) per year across this region (Ashton *et al.* 2007, pp. 5–6), primarily the result of slow, natural geologic processes (NOAA 2013b, p. 28). Due to regional differences, a 2-ft (0.6-m) rise in global sea level by the end of this century would result in a relative sea level rise of 2.3 ft (0.7 m) at New York City, 2.9 ft (0.9 m) at Hampton Roads, Virginia, and 3.5 ft (1.1 m) at Galveston, Texas (U.S. Global Change Research Program (USGCRP) 2009, p. 37). Table 15 shows that local rates of sea level rise in the range of the red knot over the second half of the 20th century were generally higher than the global rate of 0.07 in (1.8 mm) per year over roughly this same period (NRC 2010a, p. 237).

Table 15. Local sea level trends from within the range of the red knot (NOAA 2012a)

Station	Mean Local Sea Level Trend (mm per year)	Data Period
Pointe-Au-Père, Canada	-0.36 ± 0.40	1900–1983
Woods Hole, Massachusetts	2.61 ± 0.20	1932–2006
Cape May, New Jersey	4.06 ± 0.74	1965–2006
Lewes, Delaware	3.20 ± 0.28	1919–2006
Chesapeake Bay Bridge Tunnel, Virginia	6.05 ± 1.14	1975–2006
Beaufort, North Carolina	2.57 ± 0.44	1953–2006
Clearwater Beach, Florida	2.43 ± 0.80	1973–2006
Padre Island, Texas	3.48 ± 0.75	1958–2006
Punto Deseado, Argentina	-0.06 ± 1.93	1970–2002

Available information on the effects of sea level rise varies in specificity across the range of the red knot. At the international scale, only a relatively coarse assessment is possible. At the national scale, the U.S. Geological Survey's (USGS) Coastal Vulnerability Index (CVI) provides information at an intermediate level of resolution (Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999). Finally, more detailed regional, state, and local information is available for certain red knot wintering or stopover areas.

Sea Level Rise—International

International—Overview

We conducted an analysis to consider the possible effects of a 3.3-ft (1-m) increase in sea level in important nonbreeding habitats outside the United States, using global topographic mapping from the University of Arizona (Arizona Board of Regents 2012; J. Weiss pers. comm. November 13, 2012; Weiss *et al.* 2011, p. 637). This visualization tool incorporates only current topography at a horizontal resolution of 0.6 mi (1 km) (Arizona Board of Regents 2012). We did not evaluate Canadian breeding habitats for sea level rise because red knots nest inland above sea level (at elevations of up to 492 ft (150 m)) and, while in the Arctic, knots forage in freshwater wetlands and rarely contact salt water (Burger *et al.* 2012a, p. 26; Niles *et al.* 2008, pp. 27, 61).

We selected a 3.3-ft (1-m) sea level increase based on the availability of a global dataset, and because it falls within the current range of 0.7 to 6.6 ft (0.2 to 2.0 m) projected by 2100 (Parris *et al.* 2012, p. 2). Along with topography (e.g., land elevation relative to sea level), the local tidal regime is an important factor in attempting to forecast the likely effects of sea level rise (Strauss *et al.* 2012, pp. 2, 6–8). Therefore, we also considered local tidal ranges (the vertical distance between the high tide and the succeeding low tide) and other factors that may influence the extent or effects of sea level rise when site-specific information was available and appropriate. In the 1990s, some studies (e.g., Gornitz *et al.* 1994, p. 330) classified coastlines with a large tidal range ("macrotidal") (i.e., with a tidal range greater than 13 ft (4 m)) as more vulnerable to sea level rise because a large tidal range is associated with strong tidal currents that influence coastal behavior (Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999). More recently, however, the USGS inverted this ranking such that a macrotidal coastline is classified as low vulnerability. This change was based primarily on the potential influence of storms on coastal evolution, and the impact of storms relative to the tidal range. For example, on a tidal coastline, there is only a 50 percent chance of a storm occurring at high tide. Thus, for a region with a 13.1-ft (4-m) tidal range, a storm having a 9.8-ft (3-m) surge height is still up to 3.3 ft (1 m) below the elevation of high tide for half of the duration of each tidal cycle. A microtidal coastline (with a tidal range less than 6.6 ft (2 m)), on the other hand, is essentially always "near" high tide and, therefore, always at the greatest risk of significant storm impact (Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999).

Notwithstanding uncertainty about how tidal range will influence overall effects of sea level rise on coastal change, tidal range is also important due to the red knot's dependence on intertidal areas for foraging habitat. Along macrotidal coasts, large areas of intertidal habitat are exposed during low tide. In such areas, some intertidal habitat is likely to remain even with sea level rise, whereas a greater proportion of intertidal habitats may become permanently inundated in areas with smaller tidal ranges.

International—Analysis

Although no local modeling is available, large tidal ranges in the southernmost red knot wintering areas suggest extensive tidal flats will persist, although a projected 3.3-ft (1-m) rise in sea level will likely result in some habitat loss. Despite decreases in recent decades, Bahía Lomas in the Chile portion of Tierra del Fuego is still the largest single red knot wintering site. Extensive intertidal flats at Bahía Lomas are the result of daily tidal variation on the order of 20 to 30 ft (6 to 9 m), depending on the season. The Bahía Lomas flats extend for about 30 mi (50 km) along the coast, and during spring tides the intertidal distance reaches 4.3 mi (7 km) in places (Niles *et al.* 2008, p. 50). Some lands in the eastern portion of Bahía Lomas would potentially be impacted by a 3.3-ft (1-m) rise in sea level but not lands in the western portion. In the Argentina portion of Tierra del Fuego, red knots winter chiefly in Bahía San Sebastián and Río Grande (Niles *et al.* 2008, p. 17). Tides in Bahía San Sebastián are up to 13 ft (4 m). Tides in Río Grande average 18 ft (5.5 m), with a maximum of 27.6 ft (8.4 m) (Escudero *et al.* 2012, p. 356). At high tides, some lands throughout Bahía San Sebastián and Río Grande would potentially be impacted by a 3.3-ft (1-m) rise in sea level; red knot habitat could be reduced at these sites.

On the Patagonian coast of Argentina, key red knot wintering and stopover areas include the Río Gallegos estuary and Bahía de San Antonio (San Antonio Oeste) (Niles *et al.* 2008, p. 19). Tides at Río Gallegos can rise 29 ft (8.8 m) (NOAA 2013c), and low tide exposes extensive intertidal silt-clay flats that in some places extend out for 0.9 mi (1.5 km) (WHSRN) 2012). With a 3.3-ft (1-m) sea level rise, extensive areas on the north side of the Río Gallegos estuary, west of the City of Río Gallegos, would potentially be impacted. At Fracasso Beach, the tidal range is 19.7 ft (6 m) and the intertidal area is 0.6 mi (1 km) at its center (Bala *et al.* 2002, p. 27). At Bahía de San Antonio, the tidal range is 30.5 ft (9.3 m), and at low tide the water can withdraw as far as 4.3 mi (7 km) from the coastal dunes. Extensive tidal flats will persist at the lower tidal levels, even with a projected 3.3-ft (1-m) rise in sea level.

Despite decreases in recent decades, Lagoa do Peixe is a key spring stopover site for red knots on the east coast of Brazil. The lagoon is connected to the Atlantic Ocean through wind action and rain and sometimes through pumping or an artificial inlet (WHSRN 2012; Niles *et al.* 2008, p. 48). The shallow waters and mudflats that support foraging red knots are exposed irregularly by wind action and rain. The Atlantic coastline fronting Lagoa do Peixe would be impacted by a 3.3-ft (1-m) rise in sea level, which could potentially result in more extensive inundation of the lagoon through the inlet or via storm surges.

Coastal areas in North-Central Brazil in the State of Maranhão are used by migrating and wintering red knots, which forage on sandy beaches and mudflats and use extensive areas of mangroves (Niles *et al.* 2008, p. 48). In this region, local tidal ranges of up to 32.8 ft (10 m) are associated with strong tidal currents (Muehe 2010, p. 177). The largest concentrations of red knots have been recorded along the islands and complex coastline just east of Turiaçú Bay (Niles *et al.* 2008, pp. 71, 153), which has a tidal range of up to 26.2 ft (8 m) (Rebelo-Mochel and Ponzoni 2007, p. 684). Despite the large tidal ranges, topographic mapping suggests that nearly all the low-lying islands and coastline now used by red knots could become inundated by a 3.3-ft (1-m) sea level rise. As this region has low human population density (Rebelo-Mochel and Ponzoni 2007, p. 684), landward migration of suitable red knot habitats may be possible as sea levels rise. Muehe (2010, p. 177) suggested that the mangroves might be able to compensate for rising sea levels by migrating landward and laterally in some places, but movement could be frequently limited by the presence of cliffs along the open coasts and estuaries. Mangrove adaptation may not be sustained at rates of sea level rise higher than 0.3 in (7 mm) per year (Muehe 2010, p. 177), as would occur under the 3.3-ft (1-m) sea level rise scenario (CCSP 2009b, p. XV). In French Guiana, sandy beaches have experienced some substantial events of marine erosion and lost large tracks, possibly as a result of sea level rise (CSRPN 2013).

The IPCC (2007c, p. 58) evaluated the effects of a 1.6-ft (0.5-m) rise in sea level on small Caribbean islands, and found that up to 38 percent (±24 percent standard deviation) of the total current beach could be lost, with lower, narrower beaches being the most vulnerable. The IPCC did not relate this beach loss to shorebirds, but did find that sea turtle nesting habitat (the basic characteristics of which are similar to, and which often overlaps with, shorebird habitat) would be reduced by one-third under this 1.6-ft (0.5-m) scenario, which is now considered a low estimate of the sea level rise that is likely to occur by 2100 (Parris *et al.* 2012, p. 2; NRC 2010a, p. 44). In the Bahamas, ocean acidification (discussed further under Factor E, below) may exacerbate the effects of sea level rise by interfering with the biotic and chemical formation of carbonate-based sediments (Hallock 2005, pp. 25–27; Feely *et al.* 2004, pp. 365–366).

In Canada, the islands of the Mingan Archipelago could be inundated by a 3.3-ft (1-m) sea level rise. The topographic mapping shows some inundation of the adjacent mainland coastline (Mingan Archipelago National Park), as well as the Nelson River delta and the shores of James Bay, but, except where blocked by topography, red knot habitat in these areas may have more potential to migrate than on the islands. With a 3.3-ft (1-m) sea level rise, little intertidal area would be lost in the Bay of Fundy, which has the greatest tidal ranges in the world (up to 38.4 ft (11.7 m)) (NOAA 2013c), although some habitats around the mouths of rivers may become inundated. These areas are important stopover sites for red knots during migration (Newstead *et al.* 2013, pp. 56–57; Niles *et al.* 2010a, pp. 125–136; Niles *et al.* 2008, p. 94).

International—Summary

Based on our analysis of topography, tidal range, and other factors, some habitat loss in Tierra del Fuego is expected with a 3.3-ft (1-m) rise in sea level, but considerable foraging habitat is likely to remain due to very large tidal ranges. Several key South American and

Canadian stopover sites we examined are likely to be affected by sea level rise. In both Canada and South America, red knot coastal habitats are expected to migrate inland under a mid-range estimate (3.3-ft; 1-m) of sea level rise, except where constrained by topography, coastal development, or shoreline stabilization structures. The north coast of Brazil, low-lying Caribbean beaches, and Canada's Mingan Islands Archipelago may be exceptions and may experience more substantial red knot habitat loss even under moderate sea level rise. The upper range (6.6 ft; 2 m) of current predictions was not evaluated but would be expected to exceed the migration capacity of many more red knot habitats than the 3.3-ft (1-m) scenario. Thus, sea level rise is expected to result in localized habitat loss at several non-U.S. wintering and stopover areas. Cumulatively, these losses could affect the ability of red knots to complete their annual cycles that in turn may possibly affect fitness and survival.

Sea Level Rise—United States

United States—Mechanisms of Habitat Loss

Comparing topography to best available scenarios of sea level rise (as we did for our international analysis, above) provides an estimate of the land area that may be vulnerable to the effects of sea level rise (e.g., from inundation), but does not incorporate local variation in tidal regimes (Strauss *et al.* 2012, p. 2), coastal processes (e.g., barrier island migration), or environmental changes (e.g., salt marsh deterioration) that may occur as sea level rises (CCSP 2009b, p. 44). Because the majority of the U.S. Atlantic and Gulf coasts consist of sandy shores, inundation alone is unlikely to reflect the potential consequences of sea level rise. Instead, long-term shoreline changes will involve contributions from both inundation and erosion, as well as changes to other coastal environments such as wetland losses (Hinkel *et al.* 2013, p. 151; CCSP 2009b, p. 44). For example, the natural infilling of coastal embayments that occurs as sea levels rise can potentially cause erosion rates an order of magnitude faster than those predicted by standard models, implying the potential for major coastal instability at inlets (IPCC 2007c, p. 324). Most portions of the open coast of the United States will be subject to significant physical changes and erosion over the next century because the majority of coastlines consist of sandy beaches, which are highly mobile and in a state of continual change (CCSP 2009b, p. 44).

There is consensus that sea level rise will exacerbate coastal erosion (Hinkel *et al.* 2013, p. 150). Although local responses will vary, an acceleration in sea level rise will widely exacerbate beach erosion around the globe, with shoreline recession in the range 50 to 200 times the rise in relative sea level (IPCC 2007c, p. 324). Data from along the U.S. Atlantic coast suggest a relationship between rates of sea level rise and long-term erosion rates; thus, long-term coastal erosion rates are likely to increase as sea level rises (Florida Oceans and Coastal Council 2010, p. 6). However, there is not a simple relationship between sea level rise and horizontal movement of the shoreline (IPCC 2007c, p. 324). Coastal responses to climate change will not be homogeneous along the coast due to local differences in geology and other factors (Ashton *et al*. 2007, p. 9). Moreover, predicting the effect of sea level rise on red knot habitat is even more complex because, even if wetland or upland coastal lands are lost, sandy or muddy intertidal habitats can often migrate or reform along the shoreline's new position. Forecasting how such changes may unfold is complex and uncertain.

Although scientists agree that the predicted sea level rise will result in severe beach erosion and shoreline retreat through the next century, quantitative predictions of these changes are hampered by limited understanding of coastal responses and the innate complexity of the coastal zone (Ashton *et al*. 2007, p. 9). Potential effects of sea level rise on beaches vary regionally due to subsidence or uplift of the land, as well as the geological character of the coast and nearshore (U.S. Climate Change Science Program (CCSP) 2009b, p. XIV; Galbraith *et al.* 2002, p. 174). Precisely forecasting the effects of sea level rise on particular coastal habitats will require integration of diverse information on local rates of sea level rise, tidal ranges, subsurface and coastal topography, sediment accretion rates, coastal processes, and other factors; such integration is beyond the capability of current models (CCSP 2009b, pp. 27–28; Frumhoff *et al.* 2007, p. 29; IPCC 2007c, p. 324; Thieler *et al.* 2000, entire; Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999). Furthermore, human manipulation of the coastal environment through beach nourishment, hard stabilization structures, and coastal development may negate forecasts based only on the physical sciences (Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999) (see Factor A—U.S. Shoreline Stabilization and Coastal Development). In addition to direct effects on red knot habitats, rising sea levels may also indirectly affect red knots through impacts on prey resources; see Factor E—Food Availability—Other Aspects of Climate Change.

By altering coastal geomorphology, sea level rise will cause significant and often dramatic changes to coastal landforms including barrier islands, inlets, beaches, and intertidal flats (CCSP 2009b, p. 13; IPCC 2007c, p. 324; Rehfisch and Crick 2003, p. 89), primary red knot habitats (see Migration and Wintering Habitats). Beach losses accumulate over time, mostly during infrequent, high-energy events, both seasonal events and rare extreme storms (Ashton *et al*. 2009, p. 7). Even the long-term coastal response to sea level rise depends on the magnitudes and timing of stochastically unpredictable storm events (Ashton *et al*. 2009, p. 9). Thus, future changes to the U.S. Atlantic and Gulf coasts will depend upon both sea level rise and the intensity and frequency of storms, each of which will be affected by climate change (Horton *et al.* 2012). Extreme weather events will continue to be the primary driver of the highest water levels. However, a consensus has not yet been reached on how the frequency and magnitude of storms may change in coastal regions of the United States (Horton *et al.* 2012; Parris *et al.* 2012, p. 3). As discussed under Climate Change—Coastal Storms and Extreme Weather, increased magnitude and changing geographic distributions of coastal storms are predicted, but projections about changing storm patterns are associated with only “low to medium confidence” levels (IPCC 2012, p. 13). Several studies indicate that changes in the behavior or frequency of storms—along with beach protection strategies (see Factor A—U.S. Shoreline Stabilization and Coastal Development)—can be more important than the projected acceleration of sea level rise in determining future beach erosion rates (IPCC 2007c, p. 324).

It is certain that higher mean sea levels increase the frequency, magnitude, and duration of flooding associated with a given storm (Parris *et al.* 2012, p. 3). Due to increasing sea levels, storm-surge-driven floods now qualifying as 100-year events are projected to occur as often as every 10 to 20 years along most of the U.S. Atlantic coast by 2050, with even higher frequencies of such large floods in certain localized areas (Miller *et al.* 2013, pp. 3, 14; Tebaldi *et al.* 2012, pp. 7–8). Rising sea levels not only increase the likelihood of coastal flooding, but also change the template for waves and tides to sculpt the coast, which can lead to land loss orders of magnitude greater than that from direct inundation alone (Ashton *et al*. 2007, p. 1). Individual storms generally lead to rapid short-term erosion followed by rapid short-term accretion, and the net change can be negligible. However, if sediment deficiencies persist, chronic long-term erosion can result, and sea level change is one component altering coastal sediment budgets worldwide (Hinkel *et al.* 2013, p. 150). (See Factor A—U.S. Shoreline Stabilization and Coastal Development regarding factors other than sea level rise that also contribute to coastal sediment deficiencies.)

Impacts from coastal inundation and erosion will "likely" be first apparent by sea level rise adding to storm surges, making extreme water levels higher and more frequent to attack beaches and dunes (Chapter 5 in IPCC 2014, p. 15). With sea level rise, increased erosion is caused by longer storm surges and greater wave action from both tropical (especially on the southeast Atlantic and Gulf coasts) and extra-tropical storms (Higgins 2008, p. 49). Most erosion events on the Atlantic and Gulf coasts are the result of storms, and the Atlantic and Gulf shorelines are especially vulnerable to long-term sea level rise, as well as any increase in the frequency of storm surges or hurricanes. The slope of these areas is so gentle that a small rise in sea level produces a large inland shift of the shoreline (Higgins 2008, p. 49). In many locations along the U.S. coasts, small increases in sea level over the past few decades already have already increased the height of storm surge and wind-driven waves (Parris *et al.* 2012, p. 3).

In addition to the effects of storm surges, red knot habitats could also be affected by the increasing frequency and intensity of extreme precipitation events (see Climate Change). Since the ecological dynamics of sandy beaches can be linked to freshwater discharge from rivers, global changes in land-ocean coupling via freshwater outflows are predicted to affect the ecology of beaches (Schlacher *et al.* 2008a, p. 84). For example, persistent increases in freshwater discharges could cause localized habitat changes by allowing invasive or incompatible vegetation to become established, changing the seed distribution of native grasses, or altering salinity (F. Weaver pers. comm. April 17, 2013) (also see Factor E—Reduced Food Availability—Other Aspects of Climate Change).

Red knot migration and wintering habitats in the United States generally consist of sandy beaches and other intertidal areas that are dynamic and subject to seasonal erosion and accretion (the accumulation of sediment). Sea level rise and shoreline erosion have reduced the availability of intertidal habitat used for red knot foraging, and in some areas, roosting sites have also been affected (Niles *et al*. 2008, p. 97). With moderately rising sea levels, the red knot's intertidal habitats in many portions of the United States would be expected to migrate or reform rather than be lost, except where they are constrained by coastal development or shoreline stabilization (Titus *et al.* 2009, p. 1) (discussed in subsequent sections). However, if the sea rises more rapidly than the rate with which a particular coastal system can keep pace, it could fundamentally change the state of the coast (CCSP 2009b, p. 2). The upper range (6.6 ft; 2 m) of current sea level rise predictions would be expected to exceed the migration capacity of many more red knot areas than the 3.3-ft (1-m) scenario. For example, in a climate vulnerability assessment for North American shorebirds using a sea level rise of 3.3 to 6.6 ft (1 to 2 m), Galbraith *et al.* (2014) anticipated (with "high" confidence) major loss of coastal wintering habitat for shorebirds, particularly in areas where the land surface is subsiding or accretion rates of intertidal habitats are low (e.g., most Gulf coast sites). If coastal habitats are able to move inland in response to sea level rise, it could offset losses, but at many sites such habitat migration will be precluded by human infrastructure and interventions (Galbraith *et al.* 2014, p. 3).

Likewise, Iwamura *et al.* (2013, p. 6) found that upshore movement (i.e., inland migration) of intertidal habitats in response to sea level rise would greatly reduce the magnitude of shorebird population declines as intertidal habitats are lost due to sea level rise. Facilitating such movements, therefore, seems a critical conservation tool to protect migratory shorebirds from the impacts of habitat loss through sea level rise. However, these authors concluded that a scenario in which intertidal habitats migrate inland over the 21st century seems less likely, given that the realization of such new habitat will depend on appropriate sediment patterns and coastal development regimes, as well as a concomitant shift in food resources. Furthermore, managed realignment to allow existing intertidal habitat room to migrate requires careful coastal zone planning and restriction of development footprints (Iwamura *et al.* 2013, p. 6) (see Factor A—U.S. Shoreline Stabilization and Coastal Development.)

Mechanisms—Estuarine Beaches

As sea level rises, the fate of estuarine beaches (e.g., along Delaware Bay) depends on their ability to migrate and the availability of sediment to replenish eroded sands. Estuarine beaches continually erode, but under natural conditions the landward and waterward boundaries usually retreat by about the same distance. Shoreline protection structures may prevent migration, effectively squeezing beaches between development and the water (CCSP 2009b, p. 81). (See Factor A—U.S. Shoreline Stabilization and Coastal Development.)

Mechanisms—Barrier Island Beaches

The barrier islands of the Atlantic and Gulf coasts have evolved in the context of modest and decelerating sea level rise over the past 5,000 years. If human activities do not interfere, these barrier systems can typically remain intact as they migrate landward, given sea level rise rates typical of those of the last few millennia (CCSP 2009b, p. 186; Ashton *et al.* 2007, p. 2). Without stabilization, many low-lying, undeveloped islands will migrate toward the mainland, pushed by the overwashing of sand eroding from the seaward side that gets re-deposited in the bay (Scavia *et al.* 2002, p. 152). However, even without human intervention, some barrier islands may respond to sea level rise by narrowing or breaking up and drowning in place, rather than migrating (Chapter 5 in IPCC 2014, p. 15; Titus 1990, p. 67). Coastal geologists are not yet able to forecast whether a particular island will migrate or break up, although island disintegration appears to be more frequent in areas with high rates of relative sea level rise (Titus 1990, p. 67); thus, disintegration may occur more often as rates of sea level rise continue to accelerate.

Whether the barrier systems can continue to evolve with accelerated sea level rise is not clear, particularly as human intervention often does not permit the islands to continue to freely move landward (Ashton *et al.* 2007, p. 2). Sea level rise of 3.3 ft (1 m) may cause many narrow barrier islands to disintegrate (USEPA 2012). Because the coastal marshes behind many barrier islands become increasingly inundated, sufficiently high rates of sea level rise could result in threshold behaviors that produce wholesale reorganizations of entire barrier systems (CCSP 2009b, p. 2; Ashton *et al.* 2007, p. 10). Crossing threshold levels of interaction between coastal elevation, sea level, and storm-driven surges and waves can result in dramatic changes in coastal topography, including the loss of some low-lying islands (Courchamp *et al.* 2014, p. 127; Florida Oceans and Coastal Council 2010, p. 7; CCSP 2009b, p. 50; Lavoie 2009, p. 37).

United States—Coastal Vulnerability Index

At the national scale, the USGS CVI combines the coastal system's susceptibility to change with its natural ability to adapt to changing environmental conditions. The output is a relative measure of the system's natural vulnerability to the effects of sea level rise. Classification of vulnerability (very high, high, moderate, or low) is based on variables such as coastal geomorphology, regional coastal slope, rate of sea level rise, wave and tide characteristics, and historical shoreline change rates. The combination of these variables and the association of these variables to each other furnishes a broad overview of regions where physical changes are likely to occur due to sea level rise (Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999).

Table 16. Percent of coastline (by length) in each coastal vulnerability category; important red knot habitats versus the entire coast

	Very High	High	Moderate	Low
Important Red Knot Habitats				
Massachusetts	0	10	23	67
New York	0	7	50	43
New Jersey - Atlantic	69	10	22	0
New Jersey - Delaware Bay	0	77	14	9
Delaware	0	37	0	63
Virginia	99	1	0	0
North Carolina	59	15	25	1
South Carolina	59	23	18	0
Georgia	29	35	27	8
Florida - Atlantic	8	7	79	6
Florida - Gulf	2	41	53	3
Mississippi	100	0	0	0
Louisiana	100	0	0	0
Texas	63	20	17	0
All States combined	49	21	23	7
Entire Coast*				
Atlantic coast	27	22	23	28
Gulf coast	42	13	37	8
Atlantic and Gulf coasts combined	31	19	26	23

* Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999

We conducted a Geographic Information System (GIS) analysis to overlay the CVI mapping with important red knot habitats, which were delineated using data from the ISS (eBird.org 2014) and other sources. By length, about half of the coastline within important red knot habitats is in the "very high" vulnerability category, and about two-thirds is either "very high" or "high" (table 16). Comparing these percentages to the Atlantic and Gulf coasts as a whole (less than one-third "very high," only about half "high" or "very high") suggests that important red knot habitats tend to occur along higher-vulnerability portions of the shoreline. Red knot habitats along the Atlantic coast of New Jersey, Virginia, and the Carolinas and along the Gulf coast west of Florida are at particular risk from sea level rise. The GIS analysis does not reflect the potential for red knot habitats to migrate or reform (which is poorly known under high and accelerating rates of sea level rise) and did not consider human interference with coastal processes (which is discussed in subsequent sections).

United States—Northeast and Mid-Atlantic

From Maine to New Jersey, areas most vulnerable to increasing shoreline erosion with sea level rise include portions of Cape Cod, Massachusetts; Long Island, New York; and most of coastal New Jersey (Cooper *et al*. 2008, p. 488; Frumhoff *et al.* 2007, p. 15). Because of the erosive impact of waves, especially storm waves, the extent of shoreline retreat and wetland loss in this region is projected to be many times greater than the loss of land caused by the rise in sea level itself (Frumhoff *et al.* 2007, p. 15). Planners along the mainland Atlantic coast of Massachusetts, including some areas that support red knot concentrations, anticipate losses of beach and other coastal wildlife habitats over coming decades as a result of sea level rise (Kleinfelder 2013, pp. 50, 89, 122).

Geological and tide-gauge data from New York to Virginia show that sea level rise was more rapid throughout this region since the Industrial Revolution, with a 19th century rise of 0.11±0.02 in (2.7±0.4 mm) per year, and a 20th century rise of 0.15±0.01 in (3.8±0.2 mm) per year. There is a 95 percent probability that the 20th century rate of sea level rise in this region was faster than it was in any century in the last 4,300 years. These records reflect global sea level rise and natural geologic subsidence, compounded at coastal plain locations by groundwater withdrawal and compaction (Miller *et al.* 2013, p. 3). Similarly, Kemp *et al.* (2013, p. 90) found that the sea level rise rate of 0.12 in (3.1 mm) per year in New Jersey since about 1850 represents the most rapid period of change for at least 2,500 years. New Jersey and Delaware experienced a sea level rise of 3.9 to 7.9in (10 to 20 cm) during the twentieth century in addition to natural geologic land subsidence (Engelhart *et al.* 2011, p. 78). From New York to Virginia, Miller *et al.* (2013, p. 3) project a 8.7 in (22 cm) sea level rise by 2030 (range 6.3 to 15.0 in (16 to 38 cm)); 15.7 in (40 cm) by 2050 (range 11.0 to 25.6 in (28 to 65 cm)); and 37.8 in (96 cm) by 2100 (range 26.0 to 66.1 in (66 to 168 cm)), with coastal plain locations having higher rises (1.2, 2.0 to 2.4, and 3.9 to 4.7 in (3, 5 to 6, and 10 to 12 cm) higher, respectively). By 2050 under a moderate sea level rise scenario, a storm with a 10 year recurrence interval will exceed the flood level of all historical storms at Atlantic City, New Jersey (Miller *et al.* 2013, pp. 3, 14).

Along the ocean shores from New York to North Carolina, which are composed of headlands, barrier islands, and spits, it is “virtually certain” that erosion will dominate changes in shoreline as a consequence of sea level rise and storms over the next century. It is “very likely” that coastal landforms will undergo large changes under regional sea level rise scenarios of 1.6 to 3.6 ft (0.5 to 1.1 m) (CCSP 2009b, pp. XV, 43). The response will vary locally and could be more variable than the changes observed over the last century. Under these scenarios, it is “very likely” that some barrier island coasts will cross a threshold and undergo significant changes. These changes include more rapid landward migration or segmentation of some barrier islands (CCSP 2009b, p. 43) that are likely to cause substantial changes to red knot habitats.

Mid-Atlantic—Delaware Bay Shorebird Habitat

Migratory routes often include staging sites where migrants can rest and feed, and the loss of such sites can cause severe "bottleneck" effects on migratory populations. That is, sudden declines in population flow can be triggered by small amounts of overall habitat loss due to migratory connectivity. Iwamura *et al.* (2013, pp. 5, 6) showed that the population effects of habitat loss from sea level rise can be very large, as habitat loss within a bottleneck node, (i.e., a node through which a large proportion of the population passes) can drive large overall declines in population flow even if only a small fraction of total habitat is lost. Supporting an estimated 50 to 80 percent of the population each spring (Brown *et al.* 2001, p. 10), Delaware Bay represents such a bottleneck node for the rufa red knot. Iwamura *et al.* (2013, p. 7) also noted the importance of habitat quality and suggested that the effects of habitat losses at major staging areas (i.e., sites at which significant weight gain occurs and without which the migration would be impossible) would be disproportionately high. Delaware Bay fits these authors' concept of a major staging area for the rufa red knot. Thus, we conclude that habitat loss in Delaware Bay (e.g., from sea level rise) would likely produce a disproportional effect on red knot populations, based on both proportion of knots that use this site and the extent of weight gain during their stay (see Migration Biology and Migration and Wintering Food—Horseshoe Crab Eggs).

The rate of sea level rise in the Delaware Bay over the past century was about 0.12 in (3 mm) per year (table 15; Kraft *et al.* 1992, p. 233; Phillips 1986a, p. 430), resulting in erosion of the bay's shorelines and a landward extension of the inland edge of the marshes. For the period 1940 to 1978, Phillips (1986a, pp. 428–429) documented a mean erosion rate of 10.5 ft (3.2 m) per year (standard deviation of 6 ft (1.85 m) per year) for a 32.3-mi (52-km) long section of the Delaware Bay shoreline in Cumberland County, New Jersey. This is a high rate of erosion compared to other estuaries and is affected by some very high local values (e.g., peninsular points, creek mouths) approaching 49 ft (15 m) per year (Phillips 1986a, pp. 429–430). The spatial pattern of the erosion was complex, with differential erosion resistance related to local differences in shoreline morphology (Phillips 1986b, pp. 57–58). Phillips's shoreline erosion studies (1986a, pp. 431–435; 1986b, pp. 56–60) suggested that bay-edge erosion was occurring more rapidly than the landward-upward extension of the coastal wetlands and that this pattern was likely to persist. Similar to the complex and heterogeneous pattern found by Phillips, Kraft *et al.* (1992, p. 233) found that some bayshore areas in Delaware were undergoing inundation while other areas were accreting faster than the local rate of sea level rise. Accompanying these sedimentary processes were coastal erosion rates up to 22.6 ft (6.9 m) per year along the Delaware portion of the bayshore (Kraft *et al.* 1992, p. 233). Erosion has led to loss of red knot roosting sites, which are already limited, especially around the Mispillion Harbor portion of Delaware Bay (Niles *et al*. 2008, p. 97).

Looking at sedimentary evidence of seven episodes of storm-induced erosion on over the past 2,000 years on the New Jersey side of Delaware Bay, Nikitina *et al.* (2013, p. 11) found that it takes from several decades to almost 200 years for complete salt marsh recovery after storm erosion. Glick *et al.* (2008, p. 31) found that existing marsh along Delaware Bay is predicted to be inundated with greater frequency as sea level rises. Under 2.3 and 3.3 ft (0.7 and 1 m) of sea level rise, 43 and 77 percent of marshes, respectively, are predicted to be lost. The area of estuarine beach is predicted to increase substantially, roughly doubling under all sea level rise scenarios. However, this finding assumes no additional shoreline armoring would take place. Further armoring may be likely, considering 6 to 8 percent of developed and undeveloped dry land is predicted to be lost under the various scenarios evaluated. At the high end (6.6-ft (2-m) sea level rise), 18 percent of developed land would be inundated without further armoring (Glick *et al.* 2008, p. 31).

Galbraith *et al.* (2002, pp. 177–178) examined several different scenarios of future sea level rise and projected major losses of intertidal habitat in Delaware Bay. Under a scenario of 1.1 ft (34 cm) global sea level rise, Delaware Bay was predicted to lose at least 20 percent of its intertidal shorebird feeding habitats by 2050, and at least 57 percent by 2100. Under a scenario of 2.5 ft (77 cm) global sea level rise, Delaware Bay would lose 43 percent of its tidal flats by 2050, but may actually see an increase of nearly 20 percent over baseline levels by 2100, as the coastline migrates farther inland and dry land is converted to intertidal (Galbraith *et al*. 2002, pp. 177–178). However, the net increase would be realized only after a long period (50 years) of severely reduced habitat availability, and assumes that landward migration would not be halted by development or armoring. Sea Level Affecting Marsh Modeling (SLAMM) of a 3.3-ft (1-m) sea level rise at Prime Hook (Delaware) and Cape May (New Jersey) NWRs, key Delaware Bay stopover areas, suggests that estuarine beaches would survive, but with increased vulnerability to storm surges as back marsh areas become inundated (Scarborough 2009, p. 61; Stern 2009, pp. 7–9).

Mid-Atlantic—Delaware Bay Horseshoe Crab Habitat

The narrow sandy beaches used by spawning horseshoe crabs in Delaware Bay are diminishing at sometimes rapid rates due to beach erosion as a product of land subsidence and sea level rise (CCSP 2009b, p. 207). At Maurice Cove, New Jersey, for example, portions of the shoreline eroded at a rate of 14.1 ft (4.3 m) per year from 1842 to 1992. Another estimate for this area suggests the shoreline retreated about 500 ft (150 m) landward in a 32-year period, exposing ancient peat deposits that are considered suboptimal spawning habitat for the horseshoe crab. Particularly if human infrastructure along the coast leaves estuarine beaches little room to migrate inland as sea level rises, further loss of spawning habitat is likely (CCSP 2009b, p. 207).

The degree to which horseshoe crab populations will decline as beaches are lost remains unclear. Botton *et al.* (1988, p. 331) found that even subtle alteration of the sediment, such as through erosion, may affect the suitability of habitat for horseshoe crab reproduction, and that horseshoe crab spawning activity is lower in areas where erosion has exposed underlying peat (Botton *et al*. 1988, p. 325). Through habitat modeling, Czaja (2009, p. 9) found overall horseshoe crab habitat suitability in Delaware Bay was lower with a 3.9-ft (1.2-m) sea level rise than a 2-ft (0.6-m) rise, although this study did not attempt to account for landward migration. Research suggests that horseshoe crabs can successfully reproduce in alternate habitats (other than estuarine beaches), such as sandbars and the sandy banks of tidal creeks (CCSP 2009b, p. 82). However, these habitats may provide only a temporary refuge for horseshoe crabs if the alternate habitats eventually become inundated as well (CCSP 2009b, p. 82). In addition, these alternate spawning habitats may not be conducive to foraging red knots, or may not be available in sufficient amounts to support red knot and other shorebird populations during spring migration.

In 2012, the New Jersey side of Delaware Bay lost considerable horseshoe crab spawning habitat during Hurricane Sandy. A team of biologists found a 70 percent decrease in optimal horseshoe crab spawning habitat since 2002, which was judged to be mostly a result of Hurricane

Sandy (Lathrop *et al.* 2013, pp. 1–2; Niles *et al.* 2012b, p. 1). Several areas were eroded to exposed sod bank or rubble (used in past shoreline stabilization), which do not provide suitable spawning habitat. Following the storm and prior to restoration efforts (Niles *et al.* 2013a, entire; Niles *et al.* 2013b, entire), creek mouths likely constituted the bulk of the remaining intact spawning areas (A. Dey pers. comm. December 3, 2012). However, any conclusions about the long-term effects of this storm are premature due to the highly dynamic nature of the shoreline and the immediate restoration efforts that may or may not occur, due to timing and funding, after future erosional events.

United States—Southeast and the Gulf Coast

Strauss *et al.* (2012, p. 4) found more than 78 percent of the coastal dry land and freshwater wetlands on land less than 3.3 ft (1 m) above local Mean High Water in the continental United States is located in Louisiana, Florida, North Carolina, and South Carolina, indicating the vulnerability of the Southeast and Gulf coasts to effects from sea level rise.

Rates of erosion for the Southeast Atlantic region are generally highest in South Carolina along barrier islands and headland shores associated with the Santee delta. Erosion is also rapid along some barrier islands in North Carolina (Morton and Miller 2005, p. 1). Using salt marsh sediments to reconstruct sea levels for North Carolina, and adjusting for natural geologic subsidence, Kemp *et al.* (2011, p. 11017) found that sea level has risen an average of 0.98 in (2.1 mm) per year since the late 19th century, representing the steepest century-scale increase of the past two millennia. Looking at 17 recreational beaches in North Carolina and 3 local sea level rise scenarios, Bin *et al.* (2007, p. 9) projected 10 to 30 percent increases in beach erosion by 2030, and 20 to 60 percent increases by 2080. These authors assumed a constant coastwide rate of erosion, no barrier island migration, and no beach nourishment or hardening (Bin *et al.* 2007, p. 8).

The barrier islands in the Georgia Bight (southern South Carolina to northern Florida) are generally higher in elevation, wider, and more geologically stable than the microtidal barriers found elsewhere along the Atlantic coast (Leatherman 1989, p. 2-15). This lower vulnerability to sea level rise is generally reflected in the CVI (table 16). The most stable Southeast Atlantic beaches are along the east coast of Florida due to low wave energy, but also due to frequent beach nourishment (Morton and Miller 2005, p. 1), which can have both beneficial and adverse effects on red knot habitat as discussed in the section that follows. Although Florida's Atlantic coast in general is more stable than other portions of the red knot's U.S. range, localized changes from sea level rise can be significant. Modeling (SLAMM 6) of a 3.3-ft (1-m) sea level rise by 2011 at Merritt Island NWR (which supports red knots) projects a 47 percent loss of estuarine beach habitats (USFWS 2011d, p. 13).

In contrast to the more stable southern Atlantic shores of Georgia and Florida, the Gulf coast is the lowest-lying area in the United States and consequently the most sensitive to small changes in sea level (Leatherman 1989, p. 2-15). Sediment compaction and oil and gas extraction in the Gulf have compounded tectonic subsidence (a natural geologic process), leading to greater rates of relative sea level rise (Hopkinson *et al*. 2008, p. 255; Morton 2003, pp. 21–22; Morton *et al.* 2003, p. 77; Penland and Ramsey 1990, p. 323). In addition, areas with small tidal ranges are the most vulnerable to loss of intertidal wetlands and flats induced by sea level rise

(USEPA 2013b; Thieler and Hammar-Klose 2000; Thieler and Hammar-Klose 1999). Tidal range along the Gulf coast is very low, less than 3.3 ft (1 m) in some areas.

In the Florida Keys, SLAMM modeling under a mid-range sea level rise scenario showed minimal changes in sea turtle nesting habitat in 2060, but under a higher scenario of 3.3 ft (1m) by 2100 the modeling showed a 95 percent loss of sea turtle nesting habitat by 2060 (Vargas *et al.* 2013, pp. 22, 80). Although the Keys support only moderate number of red knots, the basic characteristics of sea turtle nesting habitat are similar to, and often overlap with, shorebird habitat. In Alabama, coastal land loss is caused primarily by beach and bluff erosion, but other mechanisms for loss, such as submergence, appear to be minor. Barrier islands in Mississippi are migrating laterally and erosion rates are accelerating; island areas have been reduced by about one-third since the 1850s (Morton *et al.* 2004, p. 29).

Erosion is rapid along some barrier islands and headlands in Texas (Morton *et al.* 2004, p. 4). Texas loses approximately 5 to 10 ft (1.5 to 3 m) of beach per year, as the high water line shifts landward (Higgins 2008, p. 49). Sea level rise was cited as a contributing factor in a 68 percent decline in tidal flats and algal mats in the Corpus Christi area (i.e., Lamar Peninsula to Encinal Peninsula) in Texas from the 1950s to 2004 (Tremblay *et al.* 2008, p. 59). Long-term erosion at an average rate of -5.9 ± 4.3 ft (1.8 ± 1.3 m) per year characterizes 64 percent of the Texas Gulf shoreline. Although only 48 percent of the shoreline experienced short-term erosion, the average short-term erosion rate of -8.5 ft (-2.6 m) per year is higher than the long-term rate, indicating accelerated erosion in some areas. Erosion of Gulf beaches in Texas is concentrated between Sabine Pass and High Island, downdrift (southwest) of the Galveston Island seawall, near Sargent Beach and Matagorda Peninsula, and along South Padre Island. The most stable or accreting beaches in Texas are on southwestern Bolivar Peninsula, Matagorda Island, San Jose Island, and central Padre Island (Morton *et al.* 2004, p. 32).

Rates of erosion for the U.S. Gulf coast are generally highest in Louisiana along barrier island and headland shores associated with the Mississippi delta (Morton *et al.* 2004, p. 4). Louisiana has the most rapid rate of beach erosion in the country (Leatherman 1989, p. 2-15). Subsidence and coastal erosion are functions of both natural and human-induced processes. About 90 percent of the Louisiana Gulf shoreline is experiencing erosion, which increased from an average of -26.9 ± 14.4 ft (-8.2 ± 4.4 m) per year in the long term to an average of -39.4 ft (-12.0 m) per year in the short term. Short sections of the shoreline are accreting as a result of lateral island migration, while the highest rates of erosion in Louisiana coincide with subsiding marshes and migrating barrier islands such as the Chandeleur Islands, Caminada-Moreau headland, and the Isles Dernieres (Morton *et al.* 2004, p. 31). The Mississippi River in Louisiana deposits its load of fine-grained sediments at the edge of the continental shelf in relatively deep water where it is unavailable for forming beaches and barriers. While land loss associated with shoreline change along the Gulf shore and around the margins of large coastal bays is severe (e.g., from natural and anthropogenic subsidence), loss of the interior wetlands is also extensive due to submergence and destruction of the Mississippi River delta plain (e.g., from reduced sediment supplies caused by human alterations along the river) (Morton *et al.* 2004, pp. 25, 31).

Compared to shoreline erosion in some parts of the Gulf, the average long-term erosion rate of -2.5 ± 3.0 ft (-0.8 ± 0.9 m) per year for west Florida is low, primarily because wave energy is low. Although erosion rates are generally low, more than 50 percent of the shoreline is experiencing both long-term and short-term erosion. The highest erosion rates on Florida's Gulf coast are typically localized near tidal inlets (Morton *et al.* 2004, p. 27), a preferred red knot habitat (Harrington 2008, p. 2; Harrington in Guilfoyle *et al.* 2007, pp. 18–19; Winn and Harrington in Guilfoyle *et al.* 2006, pp. 8–11). Long-term and short-term trends and rates of shoreline change are similar where there has been little or no alteration of the sediment supply or littoral system (e.g., Dog Island, St. George Island, and St. Joseph Peninsula). Conversely, trends and rates of change have shifted from long-term erosion to short-term stability or accretion where beach nourishment is common (e.g., Longboat Key, Anna Maria Island, Sand Key, and Clearwater, Panama City Beach, and Perdido Key). Slow but chronic erosion along the west coast of Florida eventually results in narrowing of the beaches (Morton *et al.* 2004, pp. 27, 29).

United States—Summary

Important red knot habitats tend to occur along higher-vulnerability portions of the U.S. shoreline. Red knot habitats along the Atlantic coast of New Jersey, Virginia, and the Carolinas and along the Gulf coast west of Florida are at particular risk from sea level rise. Delaware Bay is projected to lose substantial shorebird habitat by mid-century, even under moderate scenarios of sea level rise. In many areas, red knot coastal habitats are expected to migrate inland under mid-range estimates of sea level rise, except where constrained by topography, coastal development, or shoreline stabilization structures. Some areas may see short- or long-term net increases in red knot habitat, but low-lying and narrow islands become more prone to disintegration as sea level rise accelerates, which may produce local or regional net losses of habitat. The upper range (6.6 ft; 2 m) of current predictions was not evaluated, but would be expected to exceed the migration capacity of many more red knot habitats than the 3.3-ft (1-m) scenario.

Sea Level Rise—Summary

Due to background rates of sea level rise and the naturally dynamic nature of coastal habitats, we conclude that red knots are adapted to moderate (although sometimes abrupt) rates of habitat change in their wintering and migration areas. However, rates of sea level rise have accelerated beyond those that have occurred over recent millennia and continue to increase (IPCC 2013a, pp. 11, 25). In most of the red knot's nonbreeding range, shorelines are expected to undergo dramatic reconfigurations over the next century as a result of accelerating sea level rise (CCSP 2009b, pp. 13, 44, 50). Extensive areas of marsh are likely to become inundated, which may reduce foraging and roosting habitats. Marshes may be able to establish farther inland, but the rate of new marsh formation (e.g., intertidal sediment accumulation, development of hydric soils, colonization of marsh vegetation) may be slower than the rate of deterioration of existing marsh, particularly under the higher sea level rise scenarios (Nikitina *et al.* 2013, p. 11; Glick *et al.* 2008, p. 6). The primary red knot foraging habitats, intertidal flats and sandy beaches, will likely be locally or regionally inundated or eroded, but replacement habitats are likely to reform along the shoreline in its new position (CCSP 2009b, p. 186; Scavia *et al*. 2002, p. 152). However, if shorelines experience a decades-long period of high instability and landward migration (e.g., under higher rates of sea level rise), the formation rate of new beach habitats may be slower than the rate of loss of existing habitats (Iwamura *et al.* 2013, p. 6). In addition, low-lying and narrow islands (e.g., in the Caribbean and along the Gulf and Atlantic coasts) may disintegrate rather than migrate (Chapter 5 in IPCC 2014, p. 15; Titus 1990, p. 67), representing a net loss of red knot habitat. Superimposed on these changes are widespread human attempts to stabilize the shoreline, which are known to exacerbate losses of intertidal habitats by blocking their landward migration. The cumulative loss of habitat across the nonbreeding range could affect the ability of red knots to complete their annual cycles, possibly affecting fitness and survival, and is thereby likely to negatively influence the long-term survival of the rufa red knot (Galbraith *et al.* 2014, p. 7 and Supplement 1; NWF 2013, p. 28).

Factor A—U.S. Shoreline Stabilization and Coastal Development

Much of the U.S. coast within the range of the red knot is already extensively developed. Direct loss of shorebird habitats occurred over the past century as substantial commercial and residential developments were constructed in and adjacent to ocean and estuarine beaches along the Atlantic and Gulf coasts. In addition, red knot habitat was also lost indirectly, as sediment supplies were reduced and stabilization structures were constructed to protect developed areas.

In addition to sea level rise (discussed in the preceding section), human activities within coastal watersheds can lead to long-term reductions in sediment supply to the coast. The damming of rivers, bulk-heading of highlands, and armoring of coastal bluffs have reduced erosion in natural source areas and consequently the sediment loads reaching coastal areas. Although it is difficult to quantify, the cumulative reduction in sediment supply from human activities may contribute substantially to the long-term shoreline erosion rate. Along coastlines subject to sediment deficits, the amount of sediment supplied to the coast is less than that lost to storms and coastal sinks (inlet channels, bays, and upland deposits), leading to long-term shoreline recession (Hinkel *et al.* 2013, p. 150; Coastal Protection and Restoration Authority of Louisiana 2012, p. 18; Florida Oceans and Coastal Council 2010, p. 7; CCSP 2009b, pp. 48–49, 52–53; Defeo *et al.* 2009, p. 6; Morton *et al.* 2004, pp. 24–25; Morton 2003, pp. 11–14; Herrington 2003, p. 38; Greene 2002, p. 3).

In addition to reduced sediment supplies, other factors such as stabilized inlets, shoreline stabilization structures, and coastal development can exacerbate long-term erosion (Herrington 2003, p. 38). Coastal development and shoreline stabilization can be mutually reinforcing. Coastal development often encourages shoreline stabilization because stabilization projects cost less than the value of the buildings and infrastructure built in areas where they are at risk from flooding and erosion. Conversely, shoreline stabilization sometimes encourages coastal development by making a previously high-risk area seem safer for development (CCSP 2009b, p. 87). Protection of developed areas is the driving force behind ongoing shoreline stabilization efforts.

Large-scale shoreline stabilization projects became common in the past 100 years with the increasing availability of heavy machinery. Shoreline stabilization methods change in response to changing new technologies, coastal conditions, and preferences of residents, planners, and engineers. Along the Atlantic and Gulf coasts, an early preference for shore-perpendicular structures (e.g., groins) was followed by a period of construction of shore-parallel structures (e.g., seawalls), and then a period of beach nourishment, which is now favored (Morton *et al.* 2004, p. 4; Nordstrom 2000, pp. 13–14).

Coastal engineering projects, including channel dredging, hard armoring, and beach nourishment, affect survival of migrant and wintering shorebirds (Winn *et al.* 2013, p. 22). Past and ongoing stabilization projects fundamentally alter the naturally dynamic coastal processes that create and maintain beach strand and bayside habitats, including those habitat components that red knots rely upon. Past loss of stopover and wintering habitat likely reduced the resilience of the red knot by making it more dependent on those habitats that remain, and more vulnerable to threats (e.g., disturbance, predation, reduced quality or abundance of prey, increased intraspecific and interspecific competition) within those restricted habitats. (See Factors C and E, below, for discussions of these threats, many of which are intensified in and near developed areas. Also see Historical Distribution and Abundance.)

Shoreline Stabilization—Hard Structures

Hard structures constructed of stone, concrete, wood, steel, or geotextiles have been used for centuries as a coastal defense strategy (Defeo *et al.* 2009, p. 6). The most common hard stabilization structures fall into two groups: structures that run parallel to the shoreline (e.g., seawalls, revetments, bulkheads) and structures that run perpendicular to the shoreline (e.g., groins, jetties). Groins are often clustered in groin fields, and are intended to protect a finite section of beach, while jetties are normally constructed at inlets to keep sand out of navigation channels and provide calm-water access to harbor facilities (U.S. Army Corps of Engineers (USACE) 2002, pp. I-3-13, 21). Descriptions of the different types of stabilization structures can be found in Rice (2009, pp. 10–13), Herrington (2003, pp. 66–89), and USACE (2002, Parts V and VI).

Prior to the 1950s, the general practice in the United States was to use hard structures to protect developments from beach erosion or storm damages (USACE 2002, p. I-3-21). The pace of constructing new hard stabilization structures has since slowed considerably (USACE 2002, p. V-3-9). Many states within the range of the red knot now discourage or restrict the construction of new, hard oceanfront protection structures, although the hardening of bayside shorelines is generally still allowed (Kana 2011, p. 31; Greene 2002, p. 4; Titus 2000, pp. 742–743). Most existing hard oceanfront structures continue to be maintained, and some new structures continue to be built. Eleven new groin projects were approved in Florida from 2000 to 2009 (USFWS 2009, p. 36). In South Carolina since 2006, a new terminal groin has been constructed at one site, three more groins have been approved but not yet constructed in conjunction with a beach nourishment project, and a proposed new terminal groin is under review (M. Bimbi pers. comm. January 31, 2013). The State of North Carolina prohibited the use of hard erosion control structures in 1985; as a result, there are only a few permanent, hard stabilization structures along North Carolina's beaches (K. Matthews pers. comm. May 2, 2014; Rice 2012a, pp. 8–9). However, 2011 legislation authorized an exception for construction of up to four new terminal groins in North Carolina (Rice 2012a, p. 8), and some of North Carolina's coastal communities have begun seeking authorization from the State legislature for additional hard structures (K. Matthews pers. comm. May 2, 2014). One new terminal groin has been proposed in an area of North Carolina known to support red knots, and the project has been reviewed under section 7 of the Endangered Species Act for effects on listed and proposed species (USFWS 2014a, entire). While some states have restricted new construction, hard structures are still among the alternatives in the Federal shore protection program (USACE 2002, pp. V-3-3, 7).

Hard shoreline stabilization projects are typically designed to protect property (and its human inhabitants), not beaches (Kana 2011, p. 31; Pilkey and Howard 1981, p. 2). Hard structures affect beaches in several ways. For example, when a hard structure is put in place, erosion of the oceanfront sand continues, but the fixed back-beach line remains, resulting in a loss of beach area (USACE 2002, p. I-3-21). In addition, hard structures reduce the regional supply of beach sediment by restricting natural sand movement, further increasing erosion problems (Morton *et al.* 2004, p. 25; Morton 2003, pp. 19–20; Greene 2002, p. 3). Through effects on waves and currents, sediment transport rates, aeolian (wind) processes, and sand exchanges with dunes and offshore bars, hard structures change the erosion-accretion dynamics of beaches and constrain the natural migration of shorelines (CCSP 2009b, pp. 73, 81–82; 99–100; Defeo *et al.* 2009, p. 6; Morton 2003, pp. 19–20; Scavia *et al.* 2002, p. 152; Nordstrom 2000, pp. 98–107, 115–118). There is ample evidence of accelerated erosion rates, pronounced breaks in shoreline orientation, and truncation of the beach profile downdrift of perpendicular structures—and of reduced beach widths (relative to unprotected segments) where parallel structures have been in place over long periods of time (Hafner 2012, pp. 11–14; CCSP 2009b, pp. 99–100; Morton 2003, pp. 20–21; Scavia *et al.* 2002, p. 159; USACE 2002, pp. V-3-3, 7; Nordstrom 2000, pp. 98–107; Pilkey and Wright 1988, pp. 41, 57–59). In addition, marinas and port facilities built out from the shore can have effects similar to hard stabilization structures (Nordstrom 2000, pp. 118–119). Hard structures constructed at inlets, such as jetties, often result in loss of inlet beaches that provide shorebird habitat (Nordstrom 2000, p. 116). Compared to unhardened inlets, hardened inlets often have fewer mudflat and overwash areas, important components of shorebird habitat (Kisiel 2009, p. 52). Inlets are a preferred red knot habitat (Harrington 2008, p. 2; Harrington in Guilfoyle *et al.* 2007, pp. 18–19; Winn and Harrington in Guilfoyle *et al.* 2006, pp. 8–11).

In Delaware Bay, hard structures also cause or accelerate loss of horseshoe crab spawning habitat (CCSP 2009b, p. 82; Niles *et al.* 2008, p. 97; Botton *et al*. in Shuster *et al.* 2003, p. 16; Botton *et al.* 1988, entire), or sufficiently alter sediment quality and beach morphology to negatively affect the suitability of the remaining habitat for horseshoe crab spawning (Niles *et al.* 2008, p. 97). Shorebird habitat has been, and likely continues to be, lost where bulkheads have been built in Delaware Bay (Niles *et al.* 2008, p. 97; Clark in Farrell and Martin 1997, p. 24).

Structural development along the shoreline and manipulation of natural inlets upset the naturally dynamic coastal processes and result in loss or degradation of beach habitat (Melvin *et al.* 1991, pp. 24–25). As beaches narrow, the reduced habitat can directly lower the diversity and abundance of biota (life forms), especially in the upper intertidal zone. Shorebirds may be impacted both by reduced habitat area for roosting and foraging, and by declining intertidal prey resources, as has been documented in California (Defeo *et al.* 2009, p. 6; Dugan and Hubbard 2006, p. 10). In an estuary in England, Stillman *et al.* (2005, pp. 203–204) found that a two to eight percent reduction in intertidal area (the magnitude expected through sea level rise and industrial developments including extensive stabilization structures) decreased the predicted survival rates of five out of nine shorebird species evaluated (although not of *Calidris canutus*).

In addition to directly eliminating red knot habitat, hard structures interfere with the creation of new shorebird habitats by interrupting the natural processes of overwash and inlet formation. Where hard stabilization is installed, the eventual loss of the beach and its associated habitats is virtually assured (Rice 2009, p. 3), absent beach nourishment, which may also impact red knots as discussed below. Where they are maintained, hard structures are likely to significantly increase the amount of red knot habitat lost as sea levels continue to rise.

In a few isolated locations, however, hard structures may enhance red knot habitat, or may provide artificial habitat. In Delaware Bay, for example, Botton *et al.* (1994, p. 614) found that, in the same manner as natural shoreline discontinuities like creek mouths, jetties and other artificial obstructions can act to concentrate drifting horseshoe crab eggs and thereby attract shorebirds. Another example comes from the Delaware side of the bay, where a seawall and jetty at Mispillion Harbor protect the confluence of the Mispillion River and Cedar Creek. These structures create a low energy environment in the harbor, which seems to provide highly suitable conditions for horseshoe crab spawning over a wider variation of weather and sea conditions than anywhere else in the bay (G. Breese pers. comm. March 25, 2013). Horseshoe crab egg densities at Mispillion Harbor are consistently an order of magnitude higher than at other bay beaches (Dey *et al.* 2011a, p. 8), and this site consistently supports upwards of 15 to 20 percent of all the knots recorded in Delaware Bay (Lathrop 2005, p. 4). In Florida, A. Schwarzer (pers. comm. March 25, 2013) has observed multiple instances of red knots using artificial structures such as docks, piers, jetties, causeways, and construction barriers; we have no information regarding the frequency, regularity, timing, or significance of this use of artificial habitats. Notwithstanding localized red knot use of artificial structures, and the isolated case of hard structures improving foraging habitat at Mispillion Harbor, the nearly universal effect of such structures is the degradation or loss of red knot habitat. However, under circumstances of extreme land loss due to sea level rise, a combination of hard structures and beach nourishment may be the only available means of maintaining shorebird habitat—this may already be the case in some parts of Louisiana (Louisiana Department of Wildlife and Fisheries (LDFW) 2013) and may become more widespread as rates of sea level rise continue to increase.

Shoreline Stabilization—Mechanical Sediment Transport

Several types of sediment transport are employed to stabilize shorelines, protect development, maintain navigation channels, and provide for recreation (Gebert 2012, pp. 14, 16; Kana 2011, pp. 31–33; USACE 2002, p. I-3-7). The effects of these projects are typically expected to be relatively short in duration, usually less than 10 years, but often these actions are carried out every few years in the same area, resulting in a more lasting impact on habitat suitability for shorebirds. Mechanical sediment transport practices include beach nourishment, sediment backpassing, sand scraping, and dredging, and each practice is discussed below. In addition to affecting habitats by physically relocating sediment, these practices are also known to the preclude formation of new shorebird habitats by interfering with natural coastal processes, and can alter a habitat’s sediment properties, especially grain size. Sediment grain sizes have been shown to affect shorebird community spatial patterns by influencing benthic (bottom-dwelling) macroinvertebrate community compositions, thereby affecting prey distribution and availability (VanDusen *et al.* 2012, p. 1).

Sediment Transport—Beach Nourishment

Beach nourishment is an engineering practice of deliberately adding sand (or gravel or cobbles) to an eroding beach, or the construction of a beach where only a small beach, or no beach, previously existed (NRC 1995, pp. 23–24). Since the 1970s, 90 percent of the Federal appropriation for shore protection has been for beach nourishment (USACE 2002, p. I-3-21), which has become the preferred course of action to address shoreline erosion in the United States (Kana 2011, p. 33; Morton and Miller 2005, p. 1; Greene 2002, p. 5). Beach nourishment requires an abundant source of sand that is compatible with the native beach material. The sand is trucked to the target beach, or hydraulically pumped using dredges (Hafner 2012, p. 21). Sand for beach nourishment operations can be obtained from dry land-based sources; estuaries, lagoons, or inlets on the backside of the beach; sandy shoals in inlets and navigation channels; nearshore ocean waters; or offshore ocean waters; with the last two being the most common sources (Greene 2002, p. 6).

Where shorebird habitat has been severely reduced or eliminated by hard stabilization structures, beach nourishment may be the only means available to replace any habitat for as long as the hard structures are maintained (Nordstrom and Mauriello 2001, entire), although such habitat will persist only with regular nourishment episodes (typically on the order of every 2 to 6 years). Beach nourishment may also be the only means available for maintaining shorebird habitat in hydrologic systems with limited sediment inputs, such as Delaware Bay (DDNREC 2013). In Delaware Bay, beach nourishment has been recommended to prevent loss of spawning habitat for horseshoe crabs (Kalasz 2008, p. 34; Carter *et al*. in Guilfoyle *et al.* 2007, p. 71; ASMFC 1998, p. 28). The State of Delaware restores the bay beaches in accordance with the ASMFC horseshoe crab fishery management plan, using guidelines that provide suitable horseshoe crab spawning habitat and during times of year that do not impact spawning (DDNREC 2013). For example, beach nourishment was part of a 2009 project to maintain important shorebird foraging habitat at Mispillion Harbor, Delaware (K. Kalasz pers. comm. March 29, 2013; Siok and Wilson 2011, entire). On the New Jersey side of Delaware Bay, beach nourishment—also done outside of the spawning season—is being implemented as a means of restoring shorebird habitat following Hurricane Sandy (Niles *et al.* 2013a, entire; Niles *et al.* 2013b, entire). In addition, under circumstances of rapid land loss due to sea level rise (which may be locally or regionally exacerbated by subsidence and reduced sediment inputs), beach nourishment, perhaps even in combination with hard structures, may be the only available means of maintaining shorebird habitat. This may already be the case in some parts of Louisiana (LDFW 2013) and may become more widespread as rates of sea level rise continue to increase.

However, red knots may be directly disturbed if beach nourishment takes place while the birds are present. On New Jersey's Atlantic coast, beach nourishment has typically been scheduled for the fall, when red knots are present, because of various constraints at other times of year. In addition to causing disturbance during construction, beach nourishment often increases recreational use of the widened beaches that, without careful management, can increase disturbance of red knots. Beach nourishment can also temporarily depress, and sometimes permanently alter, the invertebrate prey base on which shorebirds depend. These effects (disturbance, reduced food resources) are discussed further under Factor E, below. Also see Factor E—Food Availability—Sediment Placement regarding possible effects of beach nourishment, both beneficial and adverse, on horseshoe crabs in Delaware Bay.

In addition to disturbing the birds and impacting the prey base, beach nourishment can affect the quality and quantity of red knot habitat (M. Bimbi pers. comm. November 1, 2012; Greene 2002, p. 5). The artificial beach created by nourishment may provide only suboptimal habitat for red knots, as a steeper beach profile is created when sand is stacked on the beach during the nourishment process. In some cases, nourishment is accompanied by the planting of dense beach grasses, which can directly degrade habitat, as red knots require sparse vegetation to avoid predation. By precluding overwash and aeolian transport, especially where large artificial dunes are constructed, beach nourishment can also lead to further erosion on the bayside and promote vegetation growth, both of which can degrade the red knot's preferred foraging and roosting habitats (sparsely vegetated flats in or adjacent to intertidal areas). Importantly, preclusion of overwash also impedes the formation of new red knot habitats through natural coastal processes. Beach nourishment can also encourage further development, bringing further habitat impacts, reducing future alternative management options such as a retreat from the coast, and perpetuating the developed and stabilized conditions that may ultimately lead to inundation where beaches are prevented from migrating (M. Bimbi pers. comm. November 1, 2012; Greene 2002, p. 5).

Following placement of sediments much coarser than those native to the beach, Peterson *et al.* (2006, p. 219) found that the area of intertidal-shallow subtidal shorebird foraging habitat was reduced by 14 to 29 percent at a site in North Carolina. Presence of coarse shell material armored the substrate surface against shorebird probing, further reducing foraging habitat by 33 percent, and probably also inhibiting manipulation of prey when encountered by a bird's bill (Peterson *et al.* 2006, p. 219). (In addition to this physical change from adding coarse sediment, nourishment that places sediment dissimilar to the native beach also substantially increases impacts to the red knot's invertebrate prey base; see Factor E—Reduced Food Availability—Sediment Placement.) Lott (2009, p. viii) found a strong negative correlation between sand placement projects and the presence of piping plovers (nonbreeding) and snowy plovers (*Charadrius alexandrinus*) (breeding and nonbreeding) in Florida.

Sediment Transport—Backpassing and Scraping

Sediment backpassing is a technique that reverses the natural migration of sediment by mechanically (via trucks) or hydraulically (via pipes) transporting sand from accreting, downdrift areas of the beach to eroding, updrift areas of the beach (Kana 2011, p. 31; Chasten and Rosati 2010, p. 5). Currently less prevalent than beach nourishment, sediment backpassing is an emerging practice because traditional nourishment methods are beginning to face constraints on budgets and sediment availability (Hafner 2012, pp. 31, 35; Chase 2006, p. 19). Beach bulldozing or scraping is the process of mechanically redistributing beach sand from the littoral zone (along the edge of the sea) to the upper beach to increase the size of the primary dune or to provide a source of sediment for beaches that have no existing dune; no new sediment is added to the system (Kana 2011, p. 30; Greene 2002, p. 5; Lindquist and Manning 2001, p. 4). Beach scraping tends to be a localized practice. In Florida beach scraping is usually used only in emergencies such as after hurricanes and other storms, but in New Jersey this practice is more routine in some areas.

Many of the effects of sediment backpassing and beach scraping are similar to those for beach nourishment (USFWS 2011c, pp. 11–24; Lindquist and Manning 2001, p. 1), including disturbance during and after construction, alteration of prey resources, reduced habitat area and quality, and precluded formation of new habitats. Relative to beach nourishment, sediment backpassing and beach scraping can involve considerably more driving of heavy trucks and other equipment on the beach including areas outside the sand placement footprint, potentially impacting shorebird prey resources over a larger area (see Factor E for discussion of vehicle impacts on prey resources) (USFWS 2011c, pp. 11–24). In addition, these practices can directly remove sand from red knot habitats, as is the case in one red knot concentration area in New Jersey (USFWS 2011c, p. 27). Backpassing and sand scraping can involve routine episodes of sand removal or transport that maintain the beach in a narrower condition, indefinitely reducing the quantity of back-beach roosting habitat.

Sediment Transport—Dredging

Sediments are also manipulated to maintain navigation channels. Many inlets in the U.S. range of the red knot are routinely dredged and sometimes relocated. In addition, nearshore areas are routinely dredged ("mined") to obtain sand for beach nourishment. Regardless of the purpose, inlet and nearshore dredging can affect red knot habitats. Dredging often involves removal of sediment from sand bars, shoals, and inlets in the nearshore zone, directly impacting optimal red knot roosting and foraging habitats (Harrington 2008, p. 2; Harrington in Guilfoyle *et al.* 2007, pp. 18–19; Winn and Harrington in Guilfoyle *et al.* 2006, pp. 8–11). These ephemeral habitats are even more valuable to red knots because they tend to receive less recreational use than the main beach strand (see Factor E—Human Disturbance, below).

In addition to causing this direct habitat loss, the dredging of sand bars and shoals can preclude the creation and maintenance of red knot habitats by removing sand sources that would otherwise act as natural breakwaters and weld onto the shore over time (Hayes and Michel 2008, p. 85; Morton 2003, p. 6). Further, removing these sand features can cause or worsen localized erosion by altering depth contours and changing wave refraction (Hayes and Michel 2008, p. 85), potentially degrading other nearby red knot habitats indirectly because inlet dynamics exert a strong influence on the adjacent shorelines. Studying barrier islands in Virginia and North Carolina, Fenster and Dolan (1996, p. 294) found that inlet influences extend 3.4 to 8.1 mi (5.4 to 13.0 km), and that inlets dominate shoreline changes for up to 2.7 mi (4.3 km). Changing the location of dominant channels at inlets can create profound alterations to the adjacent shoreline (Nordstrom 2000, p. 57).

Shoreline Stabilization and Coastal Development—Existing Extent

Existing Extent—Atlantic Coast

The mid-Atlantic coast from New York to Virginia is the most urbanized shoreline in the country, except for parts of Florida and southern California. In New York and New Jersey, hard structures and beach nourishment programs cover much of the coastline. Farther south, there are more undeveloped and preserved sections of coast (Leatherman 1989, p. 2-15). Along the entire Atlantic, most of the ocean coast is fully or partly (intermediate) developed, less than 10 percent is in conservation, and about one-third is undeveloped and still available for new development (see table 17). Farther north, substantial habitat protection currently exists for most of the 43 red knot habitats documented in Maine. Fifteen of these Maine locations are fully or partially owned by a Federal, State, or private conservation organization, and three others are under permanent conservation easements (MDIFW 2013).

By area, more than 80 percent of the land below 3.3 ft (1 m) in Florida and north of Delaware is developed or intermediate. In contrast, only 45 percent of the land from Georgia to Delaware is developed or intermediate (Titus *et al*. 2009, p. 3). However, the 55 percent undeveloped coast in this southern region includes sparsely developed portions of the Chesapeake Bay, and the bay sides of Albermarle and Pamlico Sounds in North Carolina (Titus *et al*. 2009, p. 4), which do not typically support large numbers of red knots (eBird.org 2014).

Instead, red knots concentrate along the ocean coasts (eBird.org 2014), which are more heavily developed (Titus *et al*. 2009, p. 4). Conservation lands account for most of the Virginia ocean coast, and large parts of Massachusetts, North Carolina, and Georgia, including several key red knot stopover and wintering areas. The proportion of undeveloped land is generally greater at the lowest elevations, except along New Jersey's Atlantic coast (Titus *et al*. 2009, p. 3).

Table 17. Percent* of dry land within 3.3 ft (1 m) of high water by intensity of development along the United States Atlantic coast (Titus *et al.* 2009, p. 5)

	Developed	Intermediate	Undeveloped	Conservation
Massachusetts	26	29	22	23
Rhode island	36	11	48	5
Connecticut	80	8	7	5
New York	73	18	4	6
New Jersey	66	15	12	7
Pennsylvania	49	21	26	4
Delaware	27	26	23	24
Maryland	19	16	56	9
District of Columbia	82	5	14	0
Virginia	39	22	32	7
North Carolina	28	14	55	3
South Carolina	28	21	41	10
Georgia	27	16	23	34
Florida	65	10	12	13
Coastwide	42	15	33	9

* Percentages may not add up to 100 due to rounding.

New Jersey's Atlantic coast has the longest history of stabilized barrier island shoreline in North America. It also has the most developed coastal barriers and the highest degree of stabilization in the United States (Nordstrom 2000, p. 3). As measured by the amount of shoreline in the 90 to 100 percent stabilized category, New Jersey is 43 percent hard-stabilized (Pilkey and Wright 1988, p. 46). Of New Jersey's 130 mi (209 km) of coast, 98 mi (158 km) (75 percent) are developed (including 48 mi (77 km) with ongoing beach nourishment programs), 25 mi (40 km) are preserved (including several areas with existing hard structures), and 7 mi (11 km) are inlets (Gebert 2012, p. 32). Nearly 27 mi (43.5 km) are protected by shore-parallel structures (Nordstrom 2000, pp. 21–22), including 5.6 mi (9 km) of revetments and seawalls, and there are 24 inlet jetties, 368 groins, and 1 breakwater (Hafner 2012, p. 42). Only about 15 percent of the New Jersey coast lacks hard structures, and about 70 percent is nourished (USFWS 2014b, pp. 1–2).

Although much less developed than New Jersey's Atlantic coast, Delaware Bay does have many areas of bulkheads, groins, and jetties (Botton *et al.* in Shuster *et al.* 2003, p. 16). Beach stabilization structures such as bulkheads and riprap account for 4 percent of the Delaware shoreline and 5.6 percent of the New Jersey side. An additional 2.9 and 3.4 percent of the Delaware and New Jersey shorelines, respectively, also have some form of armoring in the back-beach. About 8 percent of the Delaware bayshore is subject to near-shore development. New development and redevelopment continues on the Delaware side of the bay, although some beaches in New Jersey and Delaware have had development removed (Niles *et al.* 2008, p. 40), and the State of Delaware has acquired, protects, and manages red knot foraging and roosting habitat (DDNREC 2013). Some optimal horseshoe crab spawning habitat is also the site of existing shoreline residential development (Niles *et al.* 2008, p. 97). Delaware has a standing nourishment program in the bay, and its beaches have been regularly nourished since 1962. Approximately 3 million cubic yards (yd^3; 2.3 million cubic meters (m^3)) of sand have been placed on Delaware Bay beaches in Delaware over the past 40 years (Smith *et al.* 2002a, p. 5). In 2010, the State of Delaware completed a 10-year management plan for Delaware Bay beaches, with ongoing nourishment recommended as the key measure to protect coastal development (DDNREC 2010, p. 4). Nourishment as a means of restoring New Jersey's Delaware Bay beaches began only recently, in response to Hurricane Sandy. About 30,000 yd^3 of sand were placed on five of New Jersey's bay beaches in spring 2013 (Niles *et al.* 2013a, entire), with more nourishment in 2014 and additional projects planned for the future (Niles *et al.* 2013b, entire).

Existing Extent—Southeast Atlantic and Gulf Coasts

The U.S. southeastern coast from North Carolina to Florida is the least urbanized along the Atlantic coast, although both coasts of Florida are urbanizing rapidly. Texas has the most extensive sandy coastline in the Gulf, and much of the area is sparsely developed (Leatherman 1989, p. 2-15). Table 18 gives the miles of developed and undeveloped beach from North Carolina to Texas. (Note the difference between tables 17 and 18. Table 17 gives all dry land within 3.3 ft (1 m) of high water, while table 18 is limited to sandy, oceanfront beaches.) As shown in table 18, about 40 percent of the southeast and Gulf coast is already developed.

However, not all of the remaining 60 percent in the "undeveloped" category is still available for development because about 43 percent (about 910 miles) of beaches across this region are considered preserved. Preserved beaches include those in public or nongovernmental conservation ownership and those under conservation easements. The 43 percent of preserved beaches generally overlap with the undeveloped beach category (1,264 miles or 60 percent, as shown in table 18), but may also include some developed areas such as recreational facilities or private inholdings within parks (USFWS 2012a, p. 15). To account for such recreational or inholding development, we rounded down the estimated preserved, undeveloped beaches to about 40 percent. Adding the preserved, undeveloped 40 percent estimate to the 40 percent that is already developed, we conclude that only about 20 percent of the beaches from North Carolina to Texas are still undeveloped and available for new development. Looking at differences in preservation rates across this region, Georgia and the Mississippi barrier islands have the highest percentages of preserved beaches (76 and 100 percent of shoreline miles, respectively), Alabama and the Mississippi mainland have the lowest percentages (24 and 25 percent of shoreline miles, respectively), and all other States have between 30 and 55 percent of their beach mileage in some form of preservation (USFWS 2012a, p. 15).

Table 18. The lengths and percentages of sandy, oceanfront beach that are developed and undeveloped along the Southeast Atlantic and Gulf coasts (T. Rice pers. comm. January 3, 2013; Rice 2012a, p. 6; USFWS 2012a, p. 15)

State	Miles of Shoreline	Miles and Percent of Developed Beach	Miles and Percent of Undeveloped Beach*
North Carolina	326	159 (49%)	167 (51%)
South Carolina	182	93 (51%)	89 (49%)
Georgia	90	15 (17%)	75 (83%)
Florida	809	459 (57%)	351 (43%)
Alabama	46	25 (55%)	21 (45%)
Mississippi barrier island	27	0 (0%)	27 (100%)
Mississippi mainland**	51	41 (80%)	10 (20%)
Louisiana	218	13 (6%)	205 (94%)
Texas	370	51 (14%)	319 (86%)
Coastwide	2,119	856 (40%)	1,264 (60%)

* Beaches classified as "undeveloped" occasionally include a few scattered structures.

** The mainland Mississippi coast along Mississippi Sound includes 51.3 mi of sandy beach as of 2010-2011, out of approximately 80.7 total shoreline miles (the remaining portion is nonsandy, either marsh or armored coastline with no sand).

Table 19 shows the extent of southeast and Gulf coast shoreline with shore-parallel structures, beach nourishment, or both. Although North Carolina's mileage of armored beach is unknown, this State has only a few permanent, hard stabilization structures along its beaches—two rock revetments, two sets of groins, and two terminal groins (K. Matthews pers. comm. May 2, 2014; Rice 2012a, p. 9). However, there are approximately 350 sandbag revetments (of unknown total length) along the North Carolina's sandy, oceanfront beaches. These sandbag structures are supposed to only be in place for 2 to 5 years, but most have been in place for much longer and their fate is uncertain (Rice 2012a, p. 9). New sandbags are typically allowed only when a structure is imminently threatened, at which point there is usually not much beach habitat left (K. Matthews pers. comm. May 2, 2014). North Carolina's coastal communities are also beginning to experiment with new methods of coastal stabilization, such as sand tube groins and sand tube bulkheads. However, permanent, hardened structures (except for four new terminal groins authorized in 2011) remain prohibited by North Carolina State law (K. Matthews pers. comm. May 2, 2014). As shown in table 19, beach nourishment is very prevalent in North Carolina, as is beach bulldozing (not shown in the table), in part because of the restrictions on hard structures. Most beaches are nourished at least every 3 years, some as often as every year (K. Matthews pers. comm. May 2, 2014). Even with State regulations to ensure sediment compatibility (see Factor D—United States—Coastal Management), such frequent nourishment can interfere with natural coastal processes and can result in cumulative effects to the prey species that support shorebirds. However, without nourishment, there would be little beachfront habitat in some parts of North Carolina (K. Matthews pers. comm. May 2, 2014).

Table 19. Approximate shoreline miles of sandy, oceanfront beach that have been modified by armoring, and by sand placement activities, North Carolina to Texas, as of December 2011 (Rice 2012a, p. 7; USFWS 2012a, p. 24)

	Known Approximate Miles of Armored Beach (Percent of Total Coastline)	Known Approximate Miles of Beach Receiving Sand Placement (Percent of Total Coastline)
North Carolina	Not available	91.3 (28%)
South Carolina	Not available	67.6 (37%)
Georgia	10.5 (12%)	5.5 (6%)
Florida	117.3*	379.6 (47%)
Alabama	4.7(10%)	7.5 (16%)
Mississippi barrier island	0 (0%)	1.1 (4%)
Mississippi mainland	45.4 (89%)	43.5 (85%)
Louisiana	15.9 (7%)	60.4 (28%)
Texas	36.6 (10%)	28.3 (8%)
Total*	230.4*	684.8 (32%)

*Partial data

Existing Extent—Inlets

Of the nation's top 50 ports active in foreign waterborne commerce, over 90 percent require regular dredging. Over 392 million yd^3 (300 million m^3) of dredged material are removed from coastal navigation channels each year. Most U.S. inlets and harbors used for commercial navigation are protected and stabilized by hard structures (USACE 2002, p. I-3-7). In New Jersey, many inlets that existed around 1885 and all inlets that formed since that time were artificially closed or kept from reopening after natural closure (Nordstrom 2000, p. 19). Of the 11 New Jersey inlets that now exist, 5 are stabilized by jetties or other hard structures on both sides, and 3 more are hardened on one side other inlets (Kisiel 2009, p. 65). Past channel dredging at some of New Jersey's less stabilized inlets changed the amount of sediment transferred across the inlets and the location of accretion and erosion on adjacent shorelines (K. Nordstrom pers. comm. March 12, 2014; Nordstrom 2000, p. 20). In addition to dredging for navigation, nearshore areas are routinely dredged ("mined") to obtain sand for beach nourishment. In some areas, coastal managers regularly dredge an inlet to keep a channel from migrating or filling in, and use the sand for beach nourishment (a practice sometimes called inlet management) (K. Matthews pers. comm. May 2, 2014). Depending on the prevalent coastal drift, this form of coastal management can be similar to backpassing, in that sediment is routinely moved from areas of accretion to areas of erosion. Table 20 gives the condition of inlets from North Carolina to Texas as of 2011. However, we expect this emerging practice of inlet management may expand as sediment supplies further constrain beach nourishment, as discussed below under Future Practices.

Table 20. Inlet condition along the Southeast Atlantic and Gulf coasts, December 2011 (Rice 2012b, p. 8)

	Existing Inlets								**Artificially closed**
	Number of Inlets	**Number of Modified Inlets**	**Habitat Modification Type**						
			Structures*	Dredged	Relocated	Mined	Artificially opened		
North Carolina	20	17 (85%)	7	16	3	4	2		11
South Carolina	47	21 (45%)	17	11	2	3	0		1
Georgia	23	6 (26%)	5	3	0	1	0		0
Florida east	21	19 (90%)	19	16	0	3	10		0
Florida west	48	24 (50%)	20	22	0	6	7		1
Alabama	4	4 (100%)	4	3	0	0	0		2
Mississippi	6	5 (67%)	0	4	0	0	0		0
Louisiana	34	10 (29%)	7	9	1	2	0		46
Texas	18	14 (78%)	10	13	2	1	11		3
Total	221	119 (54%)	89 (40%)	97 (44%)	8 (4%)	20 (9%)	30 (14%)		64

*Structures include jetties, terminal groins, groin fields, rock or sandbag revetments, seawalls, and offshore breakwaters.

Shoreline Stabilization and Coastal Development—Future Practices

As shown in tables 17 and 18 and explained above, much of the Atlantic and Gulf coasts are approaching "buildout," the condition that exists when all available land is either developed or preserved and no further development is possible. Table 17 shows that about one-third of dry land within 3.3 ft (1 m) of high tide on the Atlantic coast is still available for development (i.e., not already developed or preserved), but the percent of developable land in or near red knot habitats is probably lower because oceanfront beach areas are already more developed than other lands in this dataset (see Titus *et al.* 2009, p. 4). Focused on beach habitats, USFWS (2012a, p. 15) found that only about 20 percent of the coast from North Carolina to Texas is available for development. In light of sea level rise, it is unclear the extent to which these remaining lands will be developed over the next few decades. Several states already regulate or restrict new coastal development (Titus *et al.* 2009, p. 22; Higgins 2008, pp. 50–53).

However, development pressures continue, driven by tourism (Nordstrom 2000, p. 3; New Jersey Department of Environmental Protection (NJDEP) 2010, p. 1; Gebert 2012, pp. 14, 16), as well as high coastal population densities and rapid population growth. For example, 35 million people—1 of 8 people in the United States—live within 100 mi (161 km) of the New Jersey shore (Gebert 2012, p. 17). More than 50 percent of Americans—164 million people—live in coastal counties, with 1.2 million added each year, and more than 180 million tourists visit the U.S. coasts each year (Melillo *et al.* 2014, p. 581). Of the 25 most densely populated U.S. counties, 23 are along a coast (USEPA 2012). Population density along the coast is more than five times greater than in inland areas, and coastal populations are expected to grow another 9 percent by 2020 (NOAA 2012b). Coastal population density was greatest in the Northeast as of 2003, but population growth from 1980 to 2003 was greatest in the Southeast (Crossett *et al.* 2004, pp. 4–5).

Although the likely extent of future coastal development is highly uncertain, continued efforts to protect existing and any new developments is more certain, at least over the next 10 to 20 years. As shown in tables 17 and 18, about 40 percent of the coast within the U.S. range of the red knot is already developed, and much of this area is already protected by hard or soft means, or both. Shoreline stabilization over the near term is likely to come primarily through the maintenance of existing hard structures along with beach nourishment programs. As described below, it is unknown if these practices can be sustained in the longer term (CCSP 2009b, p. 87), but protection efforts seem likely to continue over shorter timeframes (Kana 2011, p. 34; Titus *et al.* 2009, pp. 2–3; Leatherman 1989, p. 2-27).

States have shown a commitment to beach nourishment that is likely to persist. Of the 18 Atlantic and Gulf coast States with federally approved Coastal Zone Management Programs, 16 have beach nourishment policies. Nine of these 18 States have a continuing funding program for beach nourishment, and 6 more fund projects on a case-by-case basis (Higgins 2008, p. 55). Annual State appropriations for beach nourishment are $25 million in New Jersey and $30 million in Florida (Gebert 2012, p. 18). Beach nourishment has become the default solution to beach erosion because oceanfront property values have risen many times faster than the cost of nourishment (Kana 2011, p. 34). The cost of sand delivery has risen about tenfold since 1950, while oceanfront property values rose about 1,000-fold over the same timeframe. As long as these trends persist, beach nourishment will remain more cost effective than property abandonment (Kana 2011, p. 34; Titus *et al.* 1991, p. 26). Over the next 50 years, Wakefield and Parsons (2002, pp. 5, 8) project that a retreat from the coast (i.e., relocation, abandonment of buildings and infrastructure, or both) in Delaware would cost three times more than a continued beach nourishment program, assuming no decline in cost due to technological advance and no increase in cost due to diminished availability of borrow sediment or accelerated sea level rise. The State of Delaware has committed to continued beach nourishment on both its Delaware Bay and Atlantic coast beaches, with retreat from the Atlantic considered an option only for the distant future (DDNREC undated; DDNREC 2010, p. 4; Daniel 2001, p. 87).

In attempting to infer the likely future quantity of red knot habitat, major sources of uncertainty are when and where the practice of routine beach nourishment may become unsustainable and how communities will respond. It is uncertain whether beach nourishment will be continued into the future due to economic constraints, as well as often limited supplies of suitable sand resources (CCSP 2009b, p. 49). Despite the current commitment to beach nourishment, it does seem likely that this practice will eventually become unsustainable. Given rising sea levels and increased intensity of storms predicted by climate change models, a steady increase in beach replenishment would be needed to maintain usable beaches and protect coastal development (NJDEP 2010, p. 3). For example, New Jersey has seen a steady increase in costs and volumes of sand since the 1970s (NJDEP 2010, p. 2). For the case where the rate of sea level rise continues to increase, as has been projected by several recent studies, perpetual nourishment becomes impossible since the time between successive nourishment episodes continues to decrease (Weggel 1986, p. 418).

Even if it remains physically possible for beach nourishment to keep pace with sea level rise, this option may be constrained by sand availability and cost (Pietrafesa 2012, entire; NJDEP 2010, p. 2; Titus *et al.* 1991, entire; Leatherman 1989, entire). In many locations, finding sufficient sand to artificially rebuild beaches and dunes will become increasingly difficult and expensive as present supplies near project sites are depleted ("high confidence") (Chapter 5 in IPCC 2014, p. 15). For example, there is a large deficit of readily available, nearshore sand in some coastal Florida counties (Florida Oceans and Coastal Council 2010, p. 15). To maintain Florida beaches in coming years, local governments will increasingly be forced to look for suitable sand in other regions of the State and from more expensive or nontraditional sources, such as deeper waters, inland sand mines, or the Bahamas. In Florida's Broward and Miami-Dade Counties, there is estimated to be a net deficit of 34 million yd^3 (26 million m^3) of sand over the next 50 years (Florida Oceans and Coastal Council 2010, p. 15).

The cost for transporting offshore sand for beach nourishment or coastal restoration is relatively expensive, so coastal planners first use resources in areas closest to shore (Bureau of Ocean Energy Management (BOEM) 2014a, p. 3-23). Despite the higher costs of transporting sand from farther offshore, States have increasingly utilized sand from the Outer Continental Shelf (OCS), which is located in Federal waters administered by BOEM. For over 20 years, BOEM has provided OCS sand resources to complete 42 projects and convey more than 77 million yd^3 (58.9 million m^3) of material to coastal communities. Beach nourishment projects using OCS sand have been completed in Louisiana, Florida, South Carolina, North Carolina, Virginia, and Maryland. Future projects are planned in New Jersey and many of the 12 other eastern States impacted by Hurricane Sandy. The demand for OCS sand by Federal, State, and local government varies; however, over the past several years, BOEM has experienced a significant increase in the number of requests for to use OCS sand resources. This trend is most likely due to a lack of available material in state waters, increased coastal erosion as a result of more frequent and intense storms, and sea level rise (BOEM 2014b, pp. 1–2). (See background section on Climate Change, and Factor A—Sea Level Rise.) Table 21 shows the estimated nationwide quantities of sand needed to maintain current beaches (including the Pacific and Hawaii, which constitute a small part of the total) through nourishment under various sea level rise scenarios. Tremendous quantities of good quality sand would be necessary to maintain the nation's beaches. These estimates are especially remarkable given that only about 562 million yd^3 (430 million m^3) of sand were placed from 1922 to 2003 (Peterson and Bishop 2005, p. 887). Almost all of this sand must be derived from offshore, but as of 1989 only enough sand had been identified to accommodate the two lowest sea level rise scenarios over the long term. In addition, available offshore sand is not distributed evenly along the U.S. coast, so some areas will run out of local (the least expensive) sand in a few decades. Costs of beach nourishment increase substantially if sand must be acquired from considerable distance from the beach requiring nourishment (Leatherman 1989, p. 2-21). Further, much more sand would be required to stabilize the shore if barrier island disintegration or segmentation occur (CCSP 2009b, p. 102) (see Factor A—Sea Level Rise).

Table 21. Cumulative nationwide estimates of sand quantities needed (in millions of cubic yards) to maintain current beaches through nourishment under various sea level rise scenarios (Leatherman 1989, p. 2-24)

Global Sea Level Rise by 2100 / Year	2.01 ft (0.6 m)	3.65 ft (1.1 m)	5.30 ft (1.6 m)	6.94 ft (2.1 m)
2020	405	531	654	778
2040	750	1,068	1,395	1,850
2100	2,424	4,345	6,768	9,071

For the Atlantic and Gulf coasts, Titus *et al.* (1991, p. 24) estimated the cumulative cost of beach nourishment in 2100 at \$14 billion to \$69 billion for a 1.6-ft (0.5-m) sea level rise; \$25 billion to \$119 billion for a 3.3-ft (1-m) rise; and \$56 to \$230 billion for a 6.6-ft (2-m) rise. At similar rates of sea level rise, projected costs reach at least \$4.1 billion to \$10.2 billion by 2040, not adjusted for inflation (Leatherman 1989, p. 2-24). As these cumulative cost projections were produced around 1990, we divided by 110 for Titus *et al.* (1991, p. 24) and by 50 for Leatherman (1989, p. 2-24) to infer a range of estimated annual costs of \$82 million to \$2.1 billion in 1990 dollars, or about \$135 million to \$3.5 billion in 2009 dollars (U.S. Bureau of Labor Statistics 2009). For comparison, Congressional appropriations for beach nourishment projects and studies around 2009 totaled about \$150 million per fiscal year (NOAA 2009), with the Federal share typically covering 65 percent of a beach nourishment project (NOAA 2000, p. 9), for a total public expenditure of about \$231 million. Thus, public spending around 2009 was above the minimum that is expected to be necessary to keep pace with 0.5-m sea level rise (\$135 million), but was far below the maximum estimated cost to maintain beaches under the 2-m rise scenario (\$3.5 billion). In recent years, Federal funding has not kept pace with some states' demands for beach nourishment (NJDEP 2010, p. 3).

Under current policies, protection of coastal development is standard practice. However, coastal communities were designed and built without recognition of rising sea levels. Most protection structures are designed for current sea level and may not accommodate a significant rise (CCSP 2009b, p. 100). Policymakers have not decided whether the practice of protecting development should continue as sea level rises, or be modified to avoid adverse environmental consequences and increased costs of protecting coastal development (CCSP 2009b, p. 87; Titus *et al.* 2009, entire). It is unclear at what point different areas may be forced by economics or sediment availability to move beyond beach nourishment (Leatherman 1989, p. 2-27). Due to lower costs and sand recycling, sediment backpassing may prolong the ability of communities to maintain artificial beaches in some areas. However, in those times and places that artificial beach maintenance is abandoned, the remaining alternatives would likely be limited to either a retreat from the coast or increased use of hard structures to protect development (CCSP 2009b, p. 87; Defeo *et al*. 2009, p. 7; Wakefield and Parsons 2002, p. 2). Retreat is more likely in areas of lower-density development, while in areas of higher-density development, the use of hard structures may expand substantially (Florida Oceans and Coastal Council 2010, p. 16; Titus *et al.* 2009, pp. 2–3; Defeo *et al.* 2009, p. 7; Wakefield and Parsons 2002, p. 2). The quantity of red knot habitat would be markedly decreased by a proliferation of hard structures. Red knot habitat would be significantly increased by retreat, but only where hard stabilization structures do not exist or where they get dismantled.

The response to Hurricane Sandy shows that societal retreat from coastal hazards is generally not being contemplated as an option along the highly developed Atlantic coasts of New York and New Jersey at this time (USACE 2014, p. ES-1; Nee 2014a; Nee 2014b; NJDEP 2014; T. Pover pers. comm. March 14, 2014; Samuelson 2014; Weaver 2014; USACE 2013a; USACE 2013b, pp. 2, 9–11; USACE 2013c, pp. 41, 44; Christie 2013; Dawsey 2013; Huba 2013; Spoto 2013; USACE 2012; Martin 2012; Regional Plan Association 2012, p. 1; Hurricane Sandy Rebuilding Task Force undated). The State of New Jersey has made funding available for voluntary buyouts in flood-prone areas, but—even post-Sandy—there has been little interest in buyouts along the Atlantic coast, probably due to high real estate values (Nee 2014b). Hurricane Sandy recovery projects also underscore the potential for a shortage of suitable sand, exacerbated by rising sea levels, that may hamper the continuation of beach nourishment as it has been practiced in recent decades (BOEM 2014b, p. 1; Dean 2012; NJDEP 2010; Titus *et al.* 1991; Leatherman 1989; Weggel 1986), as discussed above.

Shoreline Stabilization and Coastal Development—Summary

About 40 percent of the U.S. coastline within the range of the red knot is already developed, and much of this developed area is stabilized by a combination of existing hard structures and ongoing beach nourishment programs (Rice 2012a, p. 6; Titus *et al.* 2009, p. 5). In those portions of the range for which data are available (New Jersey and North Carolina to Texas), about 40 percent of inlets, a preferred red knot habitat, are hard-stabilized, dredged, or both (Rice 2012b, p. 8; Kisiel 2009, p. 65). Hard stabilization structures and dredging degrade and often eliminate existing intertidal habitats, and in many cases prevent the formation of new shorebird habitats (CCSP 2009b, pp. 99–100; Nordstrom 2000, pp. 20, 98–107). Beach nourishment may temporarily maintain suboptimal shorebird habitats where they would otherwise be lost as a result of hard structures or sea level rise (Nordstrom and Mauriello 2001, entire), but beach nourishment can also have adverse effects to red knots and their habitats

(Defeo *et al.* 2009, p. 4; Rice 2009, entire; Peterson *et al.* 2006, entire; Peterson and Bishop 2005, entire; Greene 2002, p. 5). Demographic and economic pressures remain strong to continue existing programs of shoreline stabilization, and to develop additional areas (Melillo *et al.* 2014, p. 581; Nordstrom 2000, p. 3), with an estimated 20 to 33 percent of the coast still available for development (Rice 2012a, p. 6; Titus *et al.* 2009, p. 5). However, we expect existing beach nourishment programs will likely face eventual constraints of budget and sediment availability as sea level continues to rise (BOEM 2014b, pp. 1–2; NJDEP 2010, p. 2; Titus *et al.* 1991, entire; Weggel 1986, p. 418). In those times and places that artificial beach maintenance is abandoned, the remaining alternatives would likely be limited to either a retreat from the coast or increased use of hard structures to protect development (CCSP 2009b, p. 87; Defeo *et al*. 2009, p. 7). The quantity of red knot habitat would be markedly decreased by a proliferation of hard structures. Red knot habitat would be significantly increased by retreat, but only where hard stabilization structures do not exist or where they get dismantled. The cumulative loss of habitat across the nonbreeding range could affect the ability of red knots to complete their annual cycles, possibly affecting fitness and survival, and is thereby likely to negatively influence the long-term survival of the rufa red knot.

Factor A—International Coastal Development

The red knot's breeding area is very sparsely developed, and development is not currently a threat in this part of the subspecies' range, but may emerge as a future threat. We have little information about coastal development in the red knot's non-U.S. migration and wintering areas, compared to U.S. nonbreeding areas. However, escalating pressures caused by the combined effects of population growth, demographic shifts, economic development, and global climate change pose unprecedented threats to sandy beach ecosystems worldwide (Defeo *et al.* 2009, p. 1; Schlacher *et al.* 2008a, p. 70).

International Development—Canada

We are not aware of any current development threats in the red knot's breeding range. However, climate change is opening the Arctic to development; exploration for minerals, oil, and gas; and likely new commercial shipping lanes (see Factor E—Oil Spills and Leaks—Canada) (Niles 2013; Astill 2012; Roach 2007). Reductions in Arctic sea ice will bring new challenges as navigation routes for commercial shipping open and marine access to the region for offshore oil and gas development, tourism, fishing and other activities (NRC 2013, p. 4). Lack of infrastructure is one of several nonclimate factors that currently limit arctic shipping (Smith and Stephenson 2013, p. 2), and likely mining as well. We conclude that a substantial increase in commercial vessel traffic through the red knot's breeding grounds is likely over coming decades (Astill 2012; Smith and Stephenson 2013, p. 1), as discussed further under Factor E—Oils Spills and Leaks. However, we have no data to evaluate the likelihood that infrastructure to support this shipping, and possibly other economic activities, will impact red knot breeding areas.

Cottage-building to support tourism and expansion of suburbs is taking place along coastal areas of the Bay of Fundy (Provinces of New Brunswick and Nova Scotia) (WHSRN 2012), an important staging area for red knots (Niles *et al.* 2008, p. 30). In addition, the Bay of Fundy supports North America's only tidal electric generating facility that uses the "head" created between the water levels at high and low tide to generate electricity (National Energy

Board 2006, p. 38). The 20-megawat (MW) Annapolis Tidal Power Plant in Nova Scotia Province is a tidal barrage design, involving a large dam across the river mouth (Nova Scotia Power 2013). Tidal energy helps reduce emissions of greenhouse gases. However, tidal barrage projects can be intrusive to the area surrounding the catch basins (the area into which water flows as the tide comes in), resulting in erosion and silt accumulation (National Energy Board 2006, pp. 39–40).

Although there is good potential for further tidal barrage development in Nova Scotia, with at least two more prospects in the northeast part of the Bay of Fundy, environmental and land use impacts would be carefully assessed. There are no current plans to develop these areas, but Nova Scotia and New Brunswick Provinces and some northeastern U.S. States are studying potential for power generation from tidal currents in the Maritime region (National Energy Board 2006, p. 40). Today, engineers are moving away from tidal barrage designs, in favor of new technologies like turbines that are anchored to the ocean floor. From 2009 to 2010, the Minas Passage in the Bay of Fundy supported a 1-MW in-stream tidal turbine. There is considerable interest in exploring the full potential of this resource (Nova Scotia Power 2013). The potential impacts to red knot habitat from in-stream generation designs are likely less than barrage designs. However, without careful siting and design, potential for habitat loss exists from the terrestrial development that would likely accompany such projects.

At another important red knot stopover, James Bay, barging has been proposed in connection with diamond mining developments near Attawapiskat on the west coast of the bay. Barging could affect river mouth habitats (COSEWIC 2007, p. 37), for example, through wake-induced erosion.

International Development—Central and South America

Moving from north to south, below is the limited information we have about development in the red knot’s Central and South American migration and wintering areas.

In the Costa del Este area of Panama City, Panama, prime roosting sites were lost to housing development in the mid-2000s (Niles *et al.* 2008, p. 73). Development pressure is high around Panama Bay, and legal protections for the bay and associated wetlands are in flux (WHSRN 2014b; WHSRN 2014c).

Along the coast of French Guiana, human infrastructures are very limited, located mainly in the cities and associated harbors of Cayenne, Rémire-Montjoly, and Kourou. Red knot habitats occur near Cayenne and Kourou (CSRPN 2013).

Due to the region’s remoteness, relatively little is known about threats to red knot habitat in Maranhão, Brazil. Among the key threats that can be identified to date are offshore petroleum exploration on the continental shelf (also see Factor E—Oil Spills and Leaks, and Environmental Contaminants, below), as well as iron ore and gold mining. These activities lead to loss and degradation of coastal habitat through the dumping of soil and urban spread along the coast. Mangrove clearing has also had a negative impact on red knot habitat by altering the deposition of sediments, which leads to a reduction in benthic prey (WHSRN 2012; Niles *et al.* 2008, p. 97; COSEWIC 2007, pp. 37). Threats to shorebird habitat also exist from salt extraction operations

(WHSRN 2012). In addition to industrial development, some areas with good access have potential for tourism; however, most areas are inaccessible (WHSRN 2012). Also see Factor A—Agriculture and Aquaculture for a discussion of shrimp farming in Brazil.

Goldfeder and Blanco in Boere *et al.* (2006, p. 193) cited unplanned development as a threat to the red knot in Argentina. Development is a threat to red knot stopover habitat along the Patagonian coast of Argentina. In the Bahía Samborombón reserve, Argentina's northernmost red knot stopover site, threats come from urban and agrosystem expansion and development (Niles *et al.* 2008, p. 98).

Further south, the beaches along Bahía San Antonio, Argentina, are a key red knot stopover (Niles *et al.* 2008, p. 19). The City of San Antonio Oeste has nearly 20,000 inhabitants and many more seasonal visitors (WHSRN 2012). Just one beach on Bahía San Antonio draws 300,000 tourists every summer, a number that has increased 20 percent per year over the past decade. New access points, buildings, and tourist amusement facilities are being constructed along the beach. Until recently, there was little planning for this rapid expansion. In 2005, the first urban management plan for the area advised restricted use of land close to key shorebird areas, which include extensive dune parks. Public land ownership includes the City's shoreline, beaches, and a regional port for shipping produce and soda ash (WHSRN 2012).

Habitat loss and deterioration are among the threats confronting the urban shorebird reserves at Río Gallegos, an important red knot site in Patagonia (Niles *et al.* 2008, p. 19). As the city of Río Gallegos grew toward the coast, ecologically productive tidal flats and marshes were filled for housing and used as urban solid waste dumps and disposal sites for untreated sewage, leading to the loss of roosting areas and the loss and modification of the feeding areas (WHSRN 2012; Niles *et al.* 2008, p. 98; Ferrari *et al.* 2002, p. 39), in part as a result of wind-blown trash from a nearby landfill being deposited in shorebird habitats (Niles *et al.* 2008, p. 98; Ferrari *et al.* 2002, p. 39) (see Factor E—Environmental Contaminants). Projects have been undertaken to abate wetland filling and inappropriate trash disposal Río Gallegos (Kubiak 2010, p. 10). While the creation of the reserve stopped most of these development practices, the lots that had been approved prior to the reserve's establishment have continued to be filled. In addition, a public works project to treat the previously dumped effluents is under construction, necessitating the use of heavy equipment and the crossing of several stretches of salt marshes and mud flats used by the shorebirds. Activities outside the shorebird reserve also have potential to impact red knots. While the tidal flat and salt marsh zones most important to shorebirds are located within the reserves, the land uses of adjacent areas include recreation, fishing, cattle ranching, urban development, and three ports. In an effort to address some of these concerns, local institutions and various nongovernmental organizations are working together to reassess the coastal environment and promote its management and conservation (WHSRN 2012).

Two of Argentina's Patagonian provinces (Río Negro that includes San Antonio Oeste, and Santa Cruz that includes Río Gallegos) have declared the conservation of migratory shorebirds to be "in the Provincial interest" and made it illegal to modify wetland habitat important for shorebirds (WHSRN 2011).

Ongoing development continues to encroach in parts of Argentinean Tierra del Fuego, an important red knot wintering area (Niles *et al.* 2008, p. 17). In the area called Pasos de las Cholgas, the land immediately behind the coast has been divided, and two homes are under construction. Over time, if no urban management plan is developed, development of this area could affect red knots and their habitat. South of Pasos de las Cholgas to the mouth of the Carmen Silva River (Chico), shorebirds have disappeared and trash is deposited by the wind from the city landfill. The municipality of Río Grande is working on relocating the landfill. Also nearby, a methanol and urea plant are under construction, with plans to build two seaports, one for the company and another for the public. Between Cape Domingo and Cape Peñas is the City of Río Grande, population 80,000. In the past 25 years, the city has increased its industrial economic growth and, in turn, its population. This rapid growth was not guided by an urban management plan. The coast shows signs of deterioration from industrial activities and effects from port construction, quarries, a concrete plant, trash dumps, plants and pipelines for wastewater treatment, and debris. Río Grande City is working closely with the Provincial government to reverse the coastal degradation. One of the projects under way is the construction of an interpretive trail along the coast that teaches visitors about the marine environment and wetlands, and the importance of migratory birds as indicators of healthy environments (WHSRN 2012).

International Development—Summary

Relative to the United States, we know little about development-related threats to the red knot's nonbreeding habitat in other countries. Development is not currently a threat within the red knot's breeding range, but may emerge as a future threat as arctic warming allows greater human access to this region (Niles 2013; Smith and Stephenson 2013, p. 1). Residential and recreational development is occurring along the Bay of Fundy in Canada (WHSRN 2012), a red knot stopover site. The Bay of Fundy also has considerable potential for the expansion of electric generation from tidal energy, but new power plant developments are likely to minimize environmental impacts relative to older designs (Nova Scotia Power 2013; National Energy Board 2006, p. 40). Industrial development is considered a threat to red knot habitat along the north coast of Brazil (WHSRN 2012; COSEWIC 2007, pp. 37), but relatively little is known about this region. Urban development is a localized threat to red knot habitats in Panama (WHSRN 2014b; WHSRN 2014c), along the Patagonian coast of Argentina, and in the Argentinean portion of Tierra del Fuego. Over the past decade, shorebird conservation efforts, including the establishment of shorebird reserves and the initiation of urban planning, have made important conservation progress in many of these areas (WHSRN 2012). However, human population and development continue to grow in many coastal areas (Defeo *et al.* 2009, p. 1; Schlacher *et al.* 2008a, p. 70). In some key wintering and stopover sites, development pressures are likely to exacerbate the habitat impacts caused by sea level rise (discussed previously).

Factor A—Beach Cleaning

On beaches that are heavily used for tourism, mechanical beach cleaning (also called beach grooming or raking) is a common practice to remove wrack (seaweed and other organic debris are deposited by the tides), trash, and other natural or manmade debris by raking or sieving the sand, often with heavy equipment (Defeo *et al.* 2009, p. 4). Beach raking became common practice in New Jersey in the late 1980s (Nordstrom and Mauriello 2001, p. 23) and is increasingly common in the Southeast, especially in Florida (M. Bimbi pers. comm. November 1, 2012). In Texas, wrack removal and beach raking occur on the Gulf beach side of the developed portion of South Padre Island in the Lower Laguna Madre (USFWS 2012a, p. 28), as well as on North Padre Island, Mustang Island, and Galveston Island (D. Newstead pers. comm. May 8, 2014), all known red knot areas. Along with beach nourishment, intensive beach grooming has probably reduced the capacity the southern edge of South Padre Island to support red knots (Newstead 2014a, p. 25). On the Southeast Atlantic and Gulf coasts, beach cleaning occurs on private beaches and on some municipal or county beaches that are used by red knots (M. Bimbi pers. comm. November 1, 2012). Most wrack removal on state and Federal lands is limited to post-storm cleanup and does not occur regularly (USFWS 2012a, p. 28).

Practiced routinely, beach cleaning can cause considerable physical changes to the beach ecosystem. In addition to removing humanmade debris, beach cleaning and raking machines remove accumulated wrack, topographic depressions, emergent foredunes and hummocks, and sparse vegetation (USFWS 2012a, p. 28; Defeo *et al.* 2009, p. 4; Nordstrom and Mauriello 2001, p. 23; Nordstrom 2000, p. 53), all of which can be important microhabitats for shorebirds and their prey. Many of these changes promote erosion. Grooming loosens the beach surface by breaking up surface crusts (salt and algae) and lag elements (shells or gravel), and roughens or “fluffs” the sand, all of which increase the erosive effects of wind (Cathcart and Melby 2009, p. 14; Defeo *et al.* 2009, p. 4; Nordstrom 2000, p. 53). Grooming can also result in abnormally broad unvegetated zones that are inhospitable to dune formation or plant colonization, thereby enhancing the likelihood of erosion (Defeo *et al.* 2009, p. 4). By removing vegetation and wrack, cleaning machines also reduce or eliminate natural sand-trapping features, further destabilizing the beach (USFWS 2012a, p. 28; Nordstrom *et al*. 2006b, p. 1266; Nordstrom 2000, p. 53). Further, the sand adhering to seaweed and trapped in the cracks and crevices of wrack is lost to the beach when the wrack is removed; although the amount of sand lost during a single sweeping activity is small, over a period of years this loss could be significant (USFWS 2012a, p. 28). Cathcart and Melby (2009, pp. i, 14) found that beach raking and grooming practices on mainland Mississippi beaches exacerbate the erosion process and shorten the time interval between beach nourishment projects (see discussion of shoreline stabilization, above). In addition to promoting erosion, raking also interferes with the natural cycles of dune growth and destruction on the beach (Nordstrom and Mauriello 2001, p. 23).

Wrack removal also has significant ecological consequences, especially in regions with high levels of marine macrophyte (e.g., seaweed) production. The community structure of sandy beach macroinvertebrates can be closely linked to wrack deposits, which provide both a food source and a microhabitat refuge against desiccation (drying out). Wrack-associated animals, such as amphipods, isopods, and insects, are significantly reduced in species richness, abundance, and biomass by beach grooming (Defeo *et al.* 2009, p. 4). Invertebrates in the wrack are a primary prey base for some shorebirds such as piping plovers (USFWS 2012a, p. 28), but generally make up only a secondary part of the red knot diet (see Wintering and Migration Food). Overall shorebird numbers are positively correlated with wrack cover and the biomass of their invertebrate prey that feed on wrack; therefore, grooming can lower bird numbers (USFWS 2012a, p. 28; Defeo *et al.* 2009, p. 4). Due to their specialization on benthic, intertidal mollusks, red knots may be less impacted by these effects than some other shorebird species. However, removal of wrack may cause more significant localized effects to red knots at those times and places where abundant mussel spat are attached to deposits of tide-cast material, or where red knots become more reliant on wrack-associated prey species such as amphipods, insects, and marine worms. In Delaware Bay, red knots preferentially feed in the wrack line because horseshoe crab eggs become concentrated there (Nordstrom *et al.* 2006a, p. 438; Karpanty *et al.* 2011, pp. 990, 992) (see Winter and Migration Habitat). However, removal of wrack material is not practiced along Delaware Bay beaches (K. Clark pers. comm. February 11, 2013; A. Dey and K. Kalasz pers. comm. February 8, 2013). (More substantial threats to the red knot's prey resources are discussed under Factor E, below.)

The heavy equipment used in beach grooming can cause disturbance to roosting and foraging red knots (see Factor E—Human Disturbance, below). Because beach cleaning generally occurs on beaches intensively used for human recreation, disturbance to red knots from these recreational activities may, on many beaches, be greater than the disturbance from the beach cleaning machines. However, beach cleaning may occur at times of day (e.g., early morning, evening) when few recreational activities are taking place, thus increasing total daily duration that knots are disturbed by human activities. Conversely, many raked beaches may have such high levels of human recreational use that red knots are precluded from using them entirely (see Factor E—Human Disturbance—Precluded Use of Preferred Habitats); in such cases there would be no incremental additional disturbance from the raking activities. Where it occurs, disturbance from beach grooming may be more problematic for roosting than foraging birds because roosting red knots are particularly vulnerable to disturbance (see Factor E—Disturbance), and because beach grooming is typically focused along or landward of the high tide line where red knots may roost but are unlikely to forage. On mid-Atlantic and northern Atlantic beaches, raking is most prevalent from Memorial Day to Labor Day. In the latter part of this period (late July and August), hundreds to thousands of red knots may occur at stopover habitats in this region (B. Harrington pers. comm. November 14, 2013; eBird.org 2014). We do not have information regarding the extent to which raking is practiced in fall stopover areas when red knots are present. Further south, raking may occur year-round.

In summary, the practice of intensive beach raking may cause physical changes to beaches that degrade their suitability as red knot habitat. Removal of wrack may also have an effect on the availability of red knot food resources, particularly in those times and places that birds are more reliant on wrack-associated prey items. Beach cleaning machines are likely to cause disturbance to nonbreeding red knots, particularly roosting birds. Mechanized beach cleaning is widespread within the red knot's U.S. range, particularly in developed areas. We anticipate beach grooming may expand in some areas that become more developed but may decrease in other areas due to increasing environmental regulations, such as restrictions on beach raking in piping plover nesting areas (e.g., Nordstrom and Mauriello 2001, p. 23).

Factor A—Invasive Vegetation

Defeo *et al.* (2009, p. 6) cited biological invasions of both plants and animals as global threats to sandy beaches, with the potential to alter food webs, nutrient cycling, and invertebrate assemblages. Although the extent of the threat is uncertain, this may be due to poor survey coverage more than an absence of invasions (USFWS 2012a, p. 27). The propensity of invasive species to spread, and their tenacity once established, make them a persistent problem that is only partially countered by increasing awareness and willingness of beach managers to undertake control efforts (USFWS 2012a, p. 27). Like most invasive species, exotic coastal plants tend to reproduce and spread quickly and exhibit dense growth habits, often outcompeting native plants (USFWS 2012a, p. 27; Bahamas National Trust 2010; True 2009, pp. 1,5; Invasive Plant Atlas of New England undated). If left uncontrolled, invasive plants can cause a habitat shift from open or sparsely vegetated sand to dense vegetation (USFWS 2012a, p. 27, True 2009, p. 6; City of Sanibel undated; Invasive Plant Atlas of New England undated). Many invasive species are either affecting or have the potential to affect coastal beaches (USFWS 2012a, p. 27), and thus red knot habitat. In nonbreeding habitats, *Calidris canutus* require sparse vegetation to avoid predation (Niles *et al.* 2008, p. 44; Piersma *et al.* 1993, pp. 338–339, 349).

Beach vitex (*Vitex rotundifolia*) is a woody vine introduced into the Southeast as a dune stabilization and ornamental plant that has spread from Virginia to Florida and west to Texas (Westbrooks and Madsen 2006, pp. 1–2). There are hundreds of beach vitex occurrences in North and South Carolina, and a small number of known locations in Georgia and Florida. Targeted beach vitex eradication efforts have been undertaken in the Carolinas (USFWS 2012a, p. 27). Crowfootgrass (*Dactyloctenium aegyptium*), which grows invasively along portions of the Florida coastline, forms thick bunches or mats that can change the vegetative structure of coastal plant communities and thus alter shorebird habitat (USFWS 2009, p. 37).

Japanese (or Asiatic) sand sedge (*Carex kobomugi*) is a 4- to 12-in (10- to 30-cm) tall perennial sedge adapted to coastal beaches and dunes (Plant Conservation Alliance 2005, p. 1; Invasive Plant Atlas of New England undated). The species occurs from Massachusetts to North Carolina (U.S. Department of Agriculture (USDA) 2013) and spreads primarily by vegetative means through production of underground rhizomes (horizontal stems) (Plant Conservation Alliance 2005, p. 2). Japanese sand sedge forms dense stands on coastal dunes, outcompeting native vegetation and increasing vulnerability to erosion (Plant Conservation Alliance 2005, p. 1; Invasive Plant Atlas of New England undated). In the 2000s, Wootton (2009) documented rapid (exponential) growth in the spread of Japanese sand sedge at two New Jersey sites that are known to support shorebirds.

Australian pine (*Casuarina equisetifolia*) is not a true pine, but is actually a flowering plant. Australian pine affects shorebirds by encroaching on foraging and roosting habitat and may also provide perches for avian predators (USFWS 2012a, p. 27; Bahamas National Trust 2010, p. 1). Native to Australia and southern Asia, Australian pine is now found in all tropical and many subtropical areas of the world. This species occurs on nearly all islands of the Bahamas (Bahamas National Trust 2010, p. 2), and is among the three worst invasive exotic trees damaging wildlife habitat throughout South Florida (City of Sanibel undated). Growing well in sandy soils and salt tolerant, Australian pine is most common along shorelines (Bahamas National Trust 2010, p. 2), where it grows in dense monocultures with thick mats of acidic needles (City of Sanibel undated). In the Bahamas, Australian pine often spreads to the edge of the intertidal zone, effectively usurping all shorebird roosting habitat (A. Hecht pers. comm. December 6, 2012). In addition to directly encroaching into shorebird habitats, Australian pine contributes to beach loss through physical alteration of the dune system (Stibolt 2011; Bahamas National Trust 2010, p. 2; City of Sanibel undated). The State of Florida prohibits the sale, transport, and planting of Australian pine (Stibolt 2011; City of Sanibel undated).

In summary, red knots require open habitats that allow them to see potential predators and that are away from tall perches used by avian predators. Invasive species, particularly woody species, degrade or eliminate the suitability of red knot roosting and foraging habitats by forming dense stands of vegetation. Although not a primary cause of habitat loss, invasive species can be a regionally important contributor to the overall loss and degradation of the red knot's nonbreeding habitat.

Factor A—Agriculture and Aquaculture

In some localized areas within the red knot's range, agricultural activities or aquaculture are impacting habitat quantity and quality. For example, on the Magdalen Islands, Canada (Province of Quebec), clam farming is a growing local business. The clam farming location overlaps with the feeding grounds of transient red knots, and foraging habitats are being affected. Clam farming involves extracting all the juvenile clams from an area and relocating them in a "nursery area" nearby. The top sand layer (upper 3.9 in (10 cm) of sand) is removed and filtered. Only the clams are kept, and the remaining fauna are rejected on the site. This disturbance of benthic fauna could affect foraging rates and weight gain in red knots by removing prey, disturbing birds, and altering habitat. This pilot clam farming project could expand into more demand for clam farming in other red knot feeding areas in Canada (USFWS 2011b, p. 23) (also see Factor E—Reduced Food Availability, below).

Luckenbach (2007, p. 15) found that aquaculture of clams (*Mercenaria mercenaria*) in the lower Chesapeake Bay occurs in close proximity to shorebird foraging areas. The current distribution of clam aquaculture in the very low intertidal zone minimizes the amount of direct overlap with shorebird foraging habitats, but if clam aquaculture expands farther into the intertidal zone, more shorebird impacts (e.g., habitat alteration) may occur. However, these Chesapeake Bay intertidal zones are not considered the primary habitat for red knots (Cohen *et al.* 2009, p. 940), and red knots were not among the shorebirds observed in this study (Luckenbach 2007, p. 11).

Oyster aquaculture is practiced in Delaware Bay (NJDEP 2011, pp. 1–10), and this practice, to date, has had minimial documented effects to red knots. However, as of fall 2014, we are aware of two proposed nearshore (intertidal) aquaculture projects on New Jersey's side of Delaware Bay, and three existing operations. Some of the exiting operations may wish to expand. Nearshore aquaculture could result in more substantial knot effects than offshore (subtidal) operations. For example, if aquaculture structures or activities are permitted in intertidal habitats during the spring stopover period, they would likely disturb red knots and could create a barrier to horseshoe crab movement. Federal and State agencies are working to minimize adverse effects to red knots from Delaware Bay aquaculture activities.

Shrimp (Family Penaeidae, mainly *Litopenaeus vannamei*) farming has expanded rapidly in Brazil in recent decades. Particularly since 1998, extensive areas of mangroves and salt flats, important shorebird habitats, have been converted to shrimp ponds (Carlos *et al.* 2010, p. 1). In addition to causing habitat conversion, shrimp farm development has caused deforestation of river margins (e.g., for pumping stations), pollution of coastal waters, and changes in estuarine and tidal flat water dynamics (Campos 2007, p. 23; Zitello 2007, p. 21). Ninety-seven percent of Brazil's shrimp production is in the Northeast region of the country (Zitello 2007, p. 4). Carlos *et al.* (2010, p. 48) evaluated aerial imagery from 1988 to 2008 along 435 mi (700 km) of Brazil’s northeast coastline in the States of Piauí, Ceará, and Rio Grande do Norte, covering 20 estuaries. Over this 20-year period, shrimp farms increased by 36,644 acres (ac) (14,829 hectares (ha)), while salt flats decreased by 34,842 ac (14,100 ha) and mangroves decreased by 2,876 ac (1,164 ha) (Carlos *et al.* 2010, pp. 54, 75).

In the region of Brazil with the most intensive shrimp farming (the Northeast), newer surveys from the 2000s have documented more red knots than were previously known to use this area from earlier surveys in the 1980s. However, considering the extensive loss of shorebird habitat over this period, the difference between these two surveys does not likely represent a population increase, but rather likely reflects differences in survey methodology, intensity, and coverage. In winter aerial surveys of Northeast Brazil in 1983, Morrison and Ross (1989, Vol. 2, pp. 149, 183) documented only 15 red knots in the States of Ceará, Piauí, and eastern Maranhão. However, ground surveys in the State of Ceará in December 2007 documented an average peak count of 481 ± 31 wintering red knots at just one site, Cajuais Bank (Carlos *et al.* 2010, pp. 10–11), which is located immediately adjacent to the 1983 survey area (figure 5). Cajuais Bank also supports considerable numbers of red knots during migration, with an average peak count of 434 ± 95 in September 2007 (Carlos *et al.* 2010, pp. 10–11). Over this 1-year study, red knots were the most numerous shorebird at Cajuais Bank, accounting for nearly 25 percent of observations (Carlos *et al.* 2010, p. 9). Red knots that utilize Northeast Brazil were likely affected by recent habitat losses and degradation from the expansion of shrimp farming.

Farther west along the North-Central coast of Brazil, the western part of Maranhão and extending into the State of Pará is considered an important red knot concentration area during both winter and migration (D. Mizrahi pers. comm. November 17, 2012; Niles *et al.* 2008, p. 48; Baker *et al.* 2005, p. 12; Morrison and Ross 1989, Vol. 2, pp. 149, 183). Shrimp farm development has been far less extensive in Maranhão and Pará than in Brazil’s Northeast region (Campos 2007, pp. 3–4). However, rapid or unregulated expansion of shrimp farming in Maranhão and Pará could pose an important threat to this key red knot wintering and stopover area (WHSRN 2012). In addition to aquaculture, some fishing is practiced in Maranhão, but the area is fairly protected from conversion to land-based agriculture by its high salinity and inaccessibility (WHSRN 2012). Fishing activities could potentially cause disturbance or alter habitat conditions.

On the east coast of Brazil, Lagoa do Peixe serves as an important migration stopover for red knots. The abundance and availability of the red knot’s food supply (snails and other invertebrates) are dependent on the lagoon’s water levels. The lagoon’s natural fluctuations, and the coastal processes that allow for an annual connection of the lagoon with the sea, are altered by farmers draining water from farm fields into the lagoon. The hydrology of the lagoon is also affected by upland pine (*Pinus* spp.) plantations that cause siltation and lower the water table (Niles *et al*. 2008, pp. 97–98). These coastal habitats are also degraded by extensive upland cattle grazing, farming of food crops, and commercial shrimp farming. Fishermen also harvest from the lagoon and the sea, with trawlers setting nets along the coast (WHSRN 2012). Fishing activities could potentially cause disturbance or alter habitat conditions.

The red knot wintering and stopover area of Río Gallegos is located on the south coast of Argentina, just north of Tierra del Fuego. The lands surrounding the estuary have historically been used for raising cattle. During the past few years significant areas of brush land (that had served as a buffer) next to the shorebird reserve have been cleared and designated for agricultural use and the establishment of small farms. This loss of buffer areas may cause an increase in disturbance of the shorebirds (WHSRN 2012) because agricultural activities within visual distance of roosting or foraging shorebirds, including red knots, may cause the birds to flush. Regarding aquaculture, Goldfeder and Blanco in Boere *et al.* (2006, p. 193) cited sea farming projects as a potential threat to the red knot in Argentina. Likewise, aquaculture and seaweed farming could alter prey composition for *Calidris canutus* on Chiloè Island on the Pacific coast of Chile (B. Andres pers. comm. July 21, 2014).

Grazing of the upland buffer is also a problem at Bahía Lomas in Chilean Tierra del Fuego. The government owns all intertidal land and an upland buffer extending 262 ft (80 m) above the highest high tide, but ranchers graze sheep into the intertidal vegetation. Landowners have indicated willingness to relocate fencing to exclude sheep from the intertidal area and the upland buffer, but as of 2011, funding was needed to implement this work (L. Niles pers. comm. March 2, 2011). Grazing in the intertidal zone could potentially displace roosting and foraging red knots, as well as degrade the quality of habitat through trampling, grazing, and feces.

In summary, moderate numbers of red knots that winter or stopover in Northeast Brazil are likely impacted by past and ongoing habitat loss and degradation due to the rapid expansion of shrimp farming. Expansion of shrimp farming in North-Central Brazil, if it occurs, would affect far more red knots. Localized clam farming in Canada could degrade habitat quality and prey availability for transient red knots, and aquaculture may be impacting red knot habitats in Argentina and on Chiloé Island, Chile. Farming practices around Lagoa do Peixe are degrading habitats at this red knot stopover site. Agriculture is contributing to shorebird habitat loss and degradation at Río Gallegos in Argentina, and probably at other localized areas within the range of the red knot. However, clam farming in the Chesapeake Bay and Delaware Bay aquaculture do not appear to be impacting red knots at this time. Agriculture and aquaculture activities are a minor but locally important contributor to overall loss and degradation of the red knot's nonbreeding habitat.

Factor A—Breeding Habitat Loss and Ecosystem Change from Arctic Warming

For several decades, surface air temperatures in the Arctic have warmed at approximately twice the global rate, a phenomenon known as "polar amplification" or "Arctic amplification" (IPCC 2013a, p. 398; NASA 2013) (see Climate Change—Background). Since 1875, areas above 60 degrees (º) north latitude (around the middle of Hudson Bay) have warmed at a rate of 2.45 degrees Fahrenheit (ºF) (1.36 degrees Celsius (ºC)) per century, and since 1979, arctic land surface has warmed at an even higher rate of 0.9ºF (0.5°C) per decade (IPCC 2013a, p. 398). From 1954 to 2003, mean annual temperatures across most of Arctic Canada increased by as much as 3.6 to 5.4 ºF (2 to 3 ºC), and warming in this region has been pronounced since 1966 (Arctic Climate Impact Assessment (ACIA) 2005, p. 1101).

It is "very likely" that anthropogenic warming over the Arctic in winter will be greater than the mean global warming over the period 2016 to 2035 (IPCC 2013a, pp. 955–956).

Increased atmospheric concentrations of GHGs are "very likely" to have a larger effect on climate in the Arctic than anywhere else on the globe (ACIA 2005, p. 100). (The ACIA (2005, pp. 607) report uses likelihood terminology similar, but not identical, to that used by the IPCC; see table 14). Under two mid-range emissions scenarios, models predict a mean global temperature increase of 4.5 to 6.3 ºF (2.5 to 3.5 ºC) by 2100, while the predicted increase in the Arctic is 9 to 12.6 ºF (5 to 7 ºC). Under both emission scenarios, arctic temperatures are predicted to rise 4.5 ºF (2.5 ºC) by mid-century. Under the lower of these two emissions scenarios, some of the highest temperature increases in the Arctic (9 ºF; 5 ºC) in 2100 are predicted to occur in the Canadian Archipelago (ACIA 2005, p. 100), where the red knot breeds.

Arctic Warming—Eco-Regional Changes

To evaluate predicted changes in breeding habitat resulting from climate change, we note the eco-regional classification of the red knot's current breeding range. Most of the red knot's current breeding range (see figure 1, and Niles *et al.* 2008, p. 16) is classified as High Arctic, although some known and potential nesting areas are at the northern limits of the Low Arctic zone (CAFF 2010, p. 11). Based on mapping by the World Wildlife Fund (WWF) (2012) and modeling by Kaplan *et al.* (2003, p. 6), the red knot breeding range appears to correspond with the hemiarctic (i.e., "middle Arctic") zone described by ACIA (2005, p. 258). The region of known and potential breeding habitat is classified by the Canada Map Office (1989; 1993) as sparsely vegetated tundra, and most of the breeding range is classified by the WWF as Middle Arctic Tundra. Mapping by ACIA (2005, p. 5), based on Kaplan *et al.* (2003, entire), classifies almost all of the red knot breeding range as tundra, with only some small areas of potential breeding habitat on Melville and Bathurst Islands classified as polar desert. Kaplan *et al.* (2003, p. 6) mapped nearly all of the red knot breeding range as "prostrate dwarf-shrub tundra," which is defined as discontinuous shrubland of prostrate (low-growing) deciduous shrubs, 0 to 0.8 in (0 to 2 cm) tall, typically vegetated with willow (*Salix* spp.), avens (*Dryas* spp.), *Pedicularis*, Asteraceae, Caryophyllaceae, grasses, sedges, and true moss species (Kaplan *et al.* 2003, p. 3). The average annual temperature for the High Arctic is about -5°F (-15°C), and much of the rest of the Arctic is close to melting point for much of the year. Thus, even modest warming can therefore have a dramatic effect on the region's ecosystems (Astill 2012) (e.g., if temperatures are pushed above the melting point).

Ecological changes are already occurring in the Canadian Arctic, and are likely to impact the red knot's breeding habitat in coming decades. Rapid climate changes in the polar regions are already causing observable ecological impacts of various types and degrees of severity at all ecosystem levels. Even larger changes and more significant impacts are anticipated. As species respond to changing environments over time, their interactions with the physical world and other organisms can also change. This chain of interactions can trigger cascades of impacts throughout entire ecosystems. Evaluating the interrelated physical, chemical, biological, and societal components of polar ecosystems is key to understanding their vulnerability and resilience to climate change (NRC 2011, p. 1). In the future, trends in polar regions of populations of marine mammals, fish, and birds will be a complex response to multiple stressors and indirect effects ("high confidence") (Chapter 28 in IPCC 2014, p. 3).

High uncertainty exists about when and how changing interactions among vegetation, predators, competitors, prey, parasites, and pathogens may affect the red knot, but the extent of ecosystem change is likely profound (Fraser *et al.* 2013, entire; Gauthier *et al.* 2013, p. 10; Olofsson *et al.* 2013, entire; Schmidt *et al.* 2012, p. 4421; NRC 2011, pp. 1, 44; Meltofte *et al.* 2007, p. 35; Ims and Fuglei 2005, entire; Lindström and Agrell, entire; see Factor A and Cumulative Effects). Further, certainty is high that ecosystem changes are already underway and will continue. The following changes are already observed in the Arctic, attributed to climate change with varying degrees of confidence (Summary for Policymakers in IPCC 2014, p. 32):

- decreasing sea ice cover in summer (“high confidence”);
- decreasing snow cover extent (“medium confidence”);
- widespread permafrost degradation especially in the southern Arctic (“high confidence”);
- increased shrub cover in North American tundra (“high confidence”);
- advance of the arctic tree line in latitude and altitude (“medium confidence”);
- changed breeding area and population size of sub-Arctic birds due to snowbed reduction, tundra shrub encroachment, or both (“medium confidence”);
- loss of snowbed ecosystems and tussock tundra (a typically Low Arctic type of tundra) (“high confidence”); and
- impacts on tundra animals from increased ice layers in snow pack, following rain-on-snow events (“medium confidence”).

The IPCC notes there are early warning signs that arctic ecosystems are already experiencing irreversible regime shifts (Summary for Policymakers in IPCC 2014, p. 12). Because of their simplicity, arctic food webs may be especially prone to exhibit nonlinear dynamics in response to climate warming and to show abrupt changes due to threshold effects and feedback processes (Gauthier *et al*. 2013, p. 10). The rapid rate at which climate is changing in the polar regions will impact natural systems (“high confidence”) and may exceed the rate at which some of their components can successfully adapt (“low” to “medium confidence”). Projections suggest that arctic ecosystems could change more in the next 100 years than they did over the last 6,000 years (Kaplan *et al.* 2003, pp. 1–2), which is longer than the rufa red knot is thought to have existed as a subspecies (Buehler *et al.* 2006, p. 485; Buehler and Baker 2005, p. 505), suggesting that these ecosystem changes may exceed the knot’s adaptive capacity.

Arctic Warming—Effects to Red Knots and their Habitats

Tundra Migration and Contraction

Arctic plants, animals, and microorganisms have adapted to climate change in the geologic past primarily by relocation, and their main response to future climate change is also likely to be through relocation. In many areas of the Arctic, however, relocation possibilities will likely be limited by regional and geographical barriers (ACIA 2005, p. 997). The Canadian High Arctic is characterized by land fragmentation within the archipelago and by large glaciated areas that can constrain species’ movement and establishment (ACIA 2005, p. 1012). Even if red knots are physically capable of relocating, some important elements of their breeding habitat (e.g., vegetative elements, prey species) may not have such capacity, and thus red knots may not be ecologically capable of relocation.

Where not prevented by geographic barriers, vegetation zones are generally expected to migrate north in response to warming conditions. Warming is “very likely” to lead to slow northward displacement of tundra by forests, while tundra will in turn displace High Arctic polar desert (Feng *et al.* 2012, pp. 1359, 1366; Meltofte *et al.* 2007, p. 35; ACIA 2005, pp. 991, 998). Geographic shifting of biomes can lead to shifts in bird migration patterns (NRC 2011, p. 34). If red knots are able to shift their breeding range (i.e., poleward) in response to warming, they would likely face increased energetic and temporal demands from a longer migration. However, we lack information regarding the probable effects of a breeding range shift on the survival of individual red knots or the subspecies as a whole.

Tundra is projected to decrease to its smallest extent in the last 21,000 years, shrinking by a predicted 33 to 44 percent by 2100 (Feng *et al.* 2012, pp. 1359, 1366; Meltofte *et al.* 2007, p. 35; ACIA 2005, pp. 991, 998). Thus, with arctic warming, the zone of red knot breeding habitat is expected to contract as well as shift. In the long term, loss of tundra breeding habitat is a serious threat to shorebird species. The preferred habitats of shorebird populations that breed in the High Arctic are predicted to decrease or disappear as vegetation zones move northward (Meltofte *et al.* 2007, p. 34; Lindström and Agrell 1999, p. 145). High Arctic shorebirds such as the red knot seem to be particularly at risk, because the High Arctic already constitutes a relatively limited area “squeezed in” between the extensive Low Arctic biome and the Arctic Ocean (Meltofte *et al.* 2007, p. 35). In a circumpolar assessment of climate change impacts on Arctic-breeding waterbirds, Zöckler and Lysenko (2000, pp. 5, 13) concluded that most of the Calidrid shorebirds (*Calidris* and related species) will not be able to adapt to shrubby or treelike habitats, although they note that habitat area may not be the most important factor limiting population size or breeding success.

In a climate vulnerability assessment for North American shorebirds using a mid-range emissions scenario, Galbraith *et al.* (2014) concluded (with “medium” confidence) that tundra habitat will be reduced in extent as the tree line moves poleward. Areas that persist as tundra will become less dominated by graminoids (e.g., grasses) and other low–growth species, and will become increasingly dominated by more shrubby species, reducing the habitat value for breeding shorebirds. Also, the boreal forest will extend its range northward as it replaces tundra. Although new areas of bare ground are likely to be created by ice cap and glacial recession in high tundra areas, these authors do not believe that this will result in more habitat for most breeding shorebirds since it will persist as gravel or boulder moraine for a long period until vegetated and soil-forming processes can occur (Galbraith *et al.* 2014, p. 2).

It is unlikely that any major changes in the extent of *Calidris canutus* breeding habitat have occurred to date, but long-term changes in breeding habitat resulting from climate change are likely to negatively affect this species in the future (COSEWIC 2007, p. 16). Using two early-generation climate models and two different climate scenarios (temperature increases of 3 and 9 ºF (1.7 and 5ºC)), Zöckler and Lysenko (2000, pp. iii, 8) predicted 16 to 33 percent loss of breeding habitat across all *Calidris canutus* subspecies by 2070 to 2099. Some authors (Meltofte *et al.* 2007, p. 36; Piersma and Lindström 2004, p. 66) have suggested that the 16 to 33 percent projection is low, in part because it does not reflect ecological changes beyond outright loss of tundra. In 2007, COSEWIC concluded that, as the High Arctic zone is expected to shift north, *C. canutus* is likely to be among the species most affected. This would be the case particularly for populations breeding toward the southern part of the High Arctic zone, such as the *rufa* subspecies breeding in the central Canadian Arctic (COSEWIC 2007, p. 40), as such areas would be the first converted from tundra vegetation to shrubs and trees.

Boreal forest is generally projected by models to move northward under a warming climate, displacing between 11 and 50 percent of the tundra within 100 years (Chapter 28 in IPCC 2014, p. 27). Using multiple, recent-generation climate models and three emissions scenarios, Feng *et al*. (2012, p. 1366) found that tundra in northern Canada would be pushed poleward to the coast of the Arctic Ocean and adjacent islands and would be replaced by boreal forests and shrubs by 2040 to 2059. By 2080 to 2099, the tundra would be restricted to the islands of the Arctic Ocean, with total loss of tundra in some current red knot breeding areas (e.g., Southampton Island) (Feng *et al*. 2012, p. 1366). The findings of Feng *et al*. (2012, p. 1366) support previous mapping by ACIA (2005, p. 991) that shows the treeline migrating north to overlap with the southern end of the red knot breeding range, including Southampton Island, by 2100.

The response of arctic biomes to warming goes beyond the outright replacement of tundra by forest to also include the northward migration of vegetative subtypes within the remaining tundra zone. While predictions show forest establishment limited to the southern end of the red knot's current breeding range by 2100, migration of tundra subtypes will likely be widespread across the breeding range. A simulation by Kaplan *et al.* (2003, p. 10) showed that the current vegetative community (prostrate dwarf-shrub tundra) would be replaced by taller, denser vegetative communities throughout the entire known and potential breeding range by 2090 to 2100. The prostrate dwarf-shrub tundra would migrate north beyond the current breeding range of *Calidris canutus rufa* into the range of *C.c. islandica*, where it would replace the current community of cushion forb, lichen, and moss tundra (Kaplan *et al.* 2003, p. 10). This simulation was not intended as a realistic forward projection and did not include the potentially significant feedbacks between land surface and atmosphere. Instead, the simulation was meant to show one possible course of vegetative change and illustrate the sensitivity of arctic ecosystems to climate change (Kaplan *et al.* 2003, p. 2).

Newer modeling by Pearson *et al.* (2013) produced results consistent with Kaplan *et al.* (2003), but showed widespread vegetative changes much sooner, by mid-century. These authors projected that at least half of vegetated Arctic areas, including much of the red knot's breeding range, will shift to a different physiognomic class, and woody cover will increase by as much as 52 percent, by the 2050s. Such widespread redistribution of Arctic vegetation would have impacts that reverberate through higher trophic levels. These impacts would extend far beyond the Arctic region due to effects on species that seasonally migrate from lower latitudes and rely on finding particular polar habitats, such as open space for ground-nesting birds (Pearson *et al.* 2013, pp. 673–674) including the red knot. As with Kaplan *et al.* (2003), the modeling by Pearson *et al.* (2013, p. 675) does not account for all feedbacks associated with vegetative shifts, which may cause underestimation of woody vegetation expansion. Thus, fine-scale geographic results from this study should not be interpreted as precise predictions. In general, however, these authors found that the distributions of lower-lying vegetation classes with sparse plant cover, such as those used by nesting red knots, are predicted to contract as larger shrubs and trees expand their ranges (Pearson *et al.* 2013, p. 675).

Although biome migrations may take decades to unfold across the red knot's breeding grounds (Feng *et al*. 2012, p. 1366; Meltofte *et al.* 2007, p. 36; Kaplan *et al*. 2003, p. 10), the possibility of abrupt change also exists. In the arctic tundra, a series of positive feedback mechanisms may trigger a relatively sudden, domino-like chain of events that result in conversion from low tundra vegetation to shrubland, initiated by a relatively slight increase in temperature (CCSP 2009a, pp. 1–2). Warmer temperatures result in reduced snow cover duration, reducing surface reflectivity and increasing absorption of solar energy. The net effect is local warming, which, in turn, further accelerates the loss of snow cover. This amplified, positive feedback effect quickly leads to warmer conditions that foster the invasion of shrubs into the tundra. The new shrubs themselves then further reduce reflectivity and add to the local warming (CCSP 2009a, pp. 1–2). In the polar regions, there is considerable risk of passing thresholds and tipping points caused by the rapid response of the cryosphere (those portions of Earth's surface where water is in solid form) (NRC 2011, p. 9).

Indicators of Change

Arctic communities are "very likely" to respond strongly and rapidly to high-latitude temperature change (ACIA 2005, p. 257). The likely initial response of arctic communities to warming is an increase in the diversity of plants, animals, and microbes, but reduced dominance of currently widespread species (ACIA 2005, p. 263). Species that are important community dominants are likely to have a particularly rapid and strong effect on ecosystem processes where regional warming occurs. Hemiarctic plant species (those that occur throughout the Arctic, but most frequently in the middle Arctic) include several community dominants, such as grass, sedge, moss, and *Dryas* species (ACIA 2005, pp. 257–258), primary vegetative components of red knot nesting habitat (Niles *et al.* 2008, p. 27). Due to the current widespread distribution of these hemiarctic plants, their initial responses to climatic warming are likely to be increased productivity and abundance, probably followed by northward extension of their ranges (ACIA 2005, p. 257) (i.e., the first stage in a poleward migration, as discussed in the preceding section). Even during this initial phase of vegetative response (i.e., before any substantial changes in species composition), we expect breeding red knots to be adversely affected by increased plant height, cover, or both, based on their preference to nest in low, sparse tundra vegetation (Niles *et al.* in Baker 2001, p. 14; Canada Map Office 1989; 1993).

Studies have already documented changes in arctic vegetation consistent with these expected initial responses to warming. The "greenness" of North American tundra vegetation has increased during the period of satellite observations, 1982 to 2010 (Walker *et al.* in Richter-Menge *et al.* 2011, p. 89). Over the 29-year record, North America saw an increase in the maximum Normalized Difference Vegetation Index (NDVI, a measure of vegetation photosynthetic capacity) but no significant shift in timing of peak greenness and no significant trend toward a longer growing season. However, whole-continent data can mask changes along latitudinal gradients and in different regions. For example, looking only at the Low Arctic (from 1982 to 2003), maximum NDVI showed about a 1-week shift in the initiation of "green-up," and a somewhat higher NDVI late in the growing season. The Canadian High Arctic did not show earlier initiation of greenness, but did show a roughly 1- to 2-week shift toward earlier maximum NDVI (Walker *et al.* in Richter-Menge *et al.* 2011, pp. 91–92).

Several studies have also found increases in plant biomass linked to warming arctic temperatures (Epstein *et al.* 2012, p. 1; Hill and Henry 2011, p. 276; Hudson and Henry 2009, p. 2657). The North American High Arctic is among the Arctic regions showing the greatest increases in NDVI (Chapter 28 in IPCC 2014, p. 12). Observations from near the Lewis Glacier, Baffin Island, Canada, documented rapid vegetation changes along the margins of large retreating glaciers, and these changes may be partly responsible for large NDVI changes observed in northern Canada and Greenland (Bhatt *et al.* 2010, p. 2). Such ongoing changes to plant productivity will affect many aspects of arctic systems, including changes to active-layer depths, permafrost, and biodiversity (Bhatt *et al.* 2010, p. 2).

Importantly for red knot habitat, several studies have already found increased shrub abundance, biomass ("high confidence"), ranges, and cover; increased plant canopy heights; and decreased prevalence of bare ground (Summary for Policymakers in IPCC 2014, p. 32; Chapter 28 in IPCC 2014, p. 12; Liebezeit *et al.* 2014, p. 2; Gauthier *et al.* 2013, pp. 1, 8; Jeffries *et al.* 2013, p. 38; Elmendorf *et al.* 2012a, p. 1; Elmendorf *et al.* 2012b; Myers-Smith *et al.* 2011, p. 2; Walker *et al.* in Richter-Menge *et al.* 2011, p. 93). Ongoing increases in shrub cover in North American tundra, and advancing of the arctic tree line in latitude, are attributed to climate change with "high confidence" and "medium confidence," respectively (Summary for Policymakers in IPCC 2014, p. 32).

Community Composition

Temperature is not the only factor that currently prevents some plant species from occurring in the Arctic. Latitude is also important, as life cycles depend not only on temperature but on the light regime as well. It is very likely that arctic species will tolerate warmer summers, whereas long day lengths will initially restrict the distribution of some sub-Arctic species. This scenario will "very likely" cause new plant communities to arise with a novel species composition and structure, unlike any that exist now (ACIA 2005, p. 259). In addition, climate change will increase the vulnerability of the Arctic's terrestrial ecosystems to invasions by nonindigenous species, the majority likely to arrive through direct human assistance ("high confidence") (Chapter 28 in IPCC 2014, p. 3).

Studies have shown that during the transition from the last Ice Age to the present interglacial period, nearly all species moved north, as expected. During a significant portion of the transition period, however, the distribution of pollen types (representing plant species) provided no analogue (similar species associations) to today's vegetation communities (Root *et al.* 2013, pp. 82–83). This global warming in the geologic past caused individual species distributions to change along environmental gradients in different directions, at different rates, and over different periods. That is, whereas all species moved, they did so individual by individual, not as linked groups of species. Consequently, the groupings of species during the transition period were often dissimilar to those present today. In light of the paleoecological record, which reveals that nonanalogue climates produce nonanalogue assemblages of species, individualistic changes can be anticipated in the future. Even assuming there is enough time and space for such migrations, ecotypes will not necessarily move as a unit as the climate changes. Projections of future vegetative changes should not overlook transient states that may persist for decades. Further, past periods of warming in Earth's history should not be over-relied upon for future projects because the forecasted global average rate of temperature increase exceeds those rates typical of the last 120,000 years. If, as predicted, the rate of warming caused by the anthropogenic factors is greater than in past events, then species' individualistic responses may be even more profound (Root *et al.* 2013, pp. 82–83).

Thus, in addition to northward migration and contraction of the red knot's tundra breeding habitat, novel assemblages of species (of both plants and consumers at various trophic levels) are likely to emerge that have not previously existed. With available information, we cannot anticipate the red knot's adaptive capacity or likely response to such fundamental ecosystem changes across its breeding range.

Arctic Sea Ice

In addition to vegetative changes, the disappearance of dense ice cover on large parts of the Arctic Ocean may eliminate cooling effects on adjacent lands (Piersma and Lindström 2004, p. 66) and may cause the High Arctic climate to become more maritime-dominated, a habitat condition in which few shorebirds breed (Meltofte *et al.* 2007, p. 36). Indeed, Bhatt *et al.* (2010, pp. 1–2) used NDVI to document temporal relationships between near-coastal sea ice, summer tundra land surface temperatures, and vegetation productivity. These authors found that changes in sea ice conditions have the strongest effect on terrestrial ecosystems (e.g., accelerated warming, vegetation changes) immediately adjacent to the coast, but the effects of sea ice changes also extend far inland. Ecosystems that are currently adjacent to year-round sea ice are likely to experience the greatest changes (Bhatt *et al*. 2010, pp. 1–2). As red knots typically nest near (within about 30 mi (50 km) of) arctic coasts (Niles *et al.* 2008, p. 27; Niles *et al.* in Baker 2001, p. 14), their nesting habitats are vulnerable to accelerated temperature and vegetative changes and increasing maritime influence due to loss of sea ice.

Summer sea ice extent decreased by about 7 percent per decade from 1972 to 2002, the extent of multiyear sea ice has decreased, and ice thickness in the Arctic Basin has decreased by up to 40 percent since the 1950s and 1960s due to climate-related and other factors (ACIA 2005, p. 997). The estimated volume (reflecting both extent and thickness) of Arctic sea ice has been declining markedly since 2005, with a new record set in 2010 (World Meteorological Organization (WMO) 2013, p. 12). There is "medium confidence" that over the past three decades, Arctic summer sea ice retreat was unprecedented and sea surface temperatures were anomalously high in at least the last 1,450 years (IPCC 2013a, p. 9). Data since 2001 suggest that the rate of sea ice loss is accelerating (Stroeve *et al.* 2012, p. 1005). The five years with the lowest ever recorded sea ice extent in September were 2005, 2007, 2008, 2009 and 2010, until the previous (2007) record minimum extent (39 percent below the long-term average) was broken in 2012 (WMO 2013, p. 12). The extent of summer sea ice in 2012 was the smallest on record (during the satellite era) (NASA 2012). In September 2012 there was little sea ice in the interisland channels of Canada's Arctic Archipelago (Jeffries *et al.* 2013, p. 36), the red knot's breeding grounds. Ongoing decreases in sea ice cover in summer are attributed to climate change with "high confidence" (Summary for Policymakers in IPCC 2014, p. 32). Sea ice extent is "very likely" to continue to decrease, with predictive modeling results ranging from loss of several percent to complete loss (ACIA 2005, p. 997). Year-round reductions in arctic sea ice extent are projected by the end of the 21st century, with 43 to 94 percent reductions in September (IPCC 2013a, p. 24). A nearly ice-free Arctic Ocean in September before mid-century is "likely" (with "medium confidence") under a scenario of continued high GHG emissions (IPCC 2013a, p. 25).

Freshwater Wetlands

In addition to changes in plant communities and loss of sea ice, changes in freshwater hydrology of red knot breeding habitats are expected. Key foraging areas for red knot adults and chicks (Niles *et al.* 2008, p. 27), arctic freshwater systems are particularly sensitive to even small changes in climatic regimes. Hydrologic processes may change gradually but may also respond abruptly as environmental thresholds are exceeded (ACIA 2005, p. 1012). Loss of freshwater habitats may have more immediate effects on shorebird populations than the expansion of shrubs and trees (Meltofte *et al.* 2007, p. 35; ACIA 2005, p. 418). Rising global temperatures are expected to result in permafrost degradation, possible declines in precipitation, and lowering of water tables, leading to drying of marshes and ponds in the southern parts of the Arctic (Meltofte *et al.* 2007, p. 35; ACIA 2005, p. 418). Conversely, thawing permafrost and increasing precipitation are very likely to increase the occurrence and distribution of shallow wetlands (ACIA 2005, p. 418) in other areas. In a climate vulnerability assessment for North American shorebirds using a moderate emission scenario, Galbraith *et al.* (2014) concluded (with "medium" confidence) that changes in precipitation and evapotranspiration are likely, but the aggregate effects on tundra hydrology are difficult to predict. Drier overall conditions may be likely, and may reduce food availability during the breeding season. However, it is unclear how climate change will affect the water balance on tundra breeding habitats due to the complex interaction of several factors, including amounts and timing of precipitation events, timing and extent of spring thaw, depth of the active layer, and erosion events. While annual rainfall is predicted to increase throughout the breeding range, evapotranspiration is also expected to increase enough to more than offset the effect of increased precipitation. The result may be a loss of some wetland breeding habitat to dryer conditions, but this is unclear (Galbraith *et al.* 2014, pp. 2–3).

Due to the uncertainties discussed above, we cannot predict the likely net changes in wetland availability across the red knot's breeding range over coming decades. However, changes to freshwater systems are already underway. Climate change is already impacting terrestrial and freshwater ecosystems in some areas of the Arctic due to ecological effects resulting from reductions in the duration and extent of ice and snow cover and enhanced permafrost thaw ("very high confidence"), and through changes in the precipitation-evaporation balance ("medium confidence") (Chapter 28 in IPCC 2014, p. 2). It is "virtually certain" that near-surface permafrost extent at high northern latitudes will be reduced as global mean surface temperature increases. By the end of the 21st century, the area of permafrost near the surface is projected to decrease by between 37 to 81 percent (IPCC 2013a, p. 25).

Other Ecosystem Changes

Potential impacts to shorebirds from changing arctic ecosystems go well beyond the loss and degradation of tundra nesting and wetland foraging habitats (e.g., see Fraser *et al.* 2013, entire; Schmidt *et al.* 2012, p. 4421; Meltofte *et al.* 2007, p. 35; Ims and Fuglei 2005, entire; Lindström and Agrell, entire). One such ecosystem change involves colonization of the Arctic by a new suite of species. Animal terrestrial biodiversity is generally projected to increase in the

Arctic due to warming by immigration of new species from the south, vegetation changes, and indirectly by introduction of invasive species caused by increased human activities and increased survival of such species ("high confidence") (Chapter 28 in IPCC 2014, p. 28). A continuation of warm summers may lead to more and different parasites, pathogens, and predators (see Factor C—Predation—Breeding Areas). Northward expansion of Low Arctic and possibly sub-Arctic breeding shorebirds may lead to interspecific competition for an increasingly limited supply of suitable nesting habitat (Meltofte *et al.* 2007, p. 35). For example, ongoing changes in breeding areas and population sizes of sub-Arctic birds due to snowbed reduction, tundra shrub encroachment, or both, are attributed to climate change with "medium confidence" (Summary for Policymakers in IPCC 2014, p. 32). Invasion by new competitors is a threat faced by tundra species (Gauthier *et al.* 2013, p. 9).

Arctic Warming—Summary

Arctic regions are warming about twice as fast as the global average rate (IPCC 2013a, p. 398), and the Canadian Archipelago is predicted to experience some of the fastest warming in the Arctic (ACIA 2005, p. 100). Red knots currently breed in a region of sparse, low tundra vegetation within the southern part of the High Arctic and the northern limits of the Low Arctic (CAFF 2010, p. 11; Niles *et al.* 2008, p. 16; Niles *et al.* in Baker 2001, p. 14). Forests are expected to colonize the southern part of the red knot's current breeding range by 2100 (ACIA 2005, pp. 991, 998), and vegetation throughout the entire breeding range is likely to become taller and denser and with less bare ground, potentially making it unsuitable for red knot nesting, possibly as soon as mid-century (Galbraith *et al.* 2014, p. 2; Pearson *et al.* 2013, pp. 673–675; COSEWIC 2007, p. 40; Zöckler and Lysenko 2000, pp. 5, 13). Studies have already documented changes in arctic vegetation, including increases in peak "greenness" of North American tundra vegetation since 1982; increases in plant biomass linked to warming arctic temperatures; advancing of the arctic tree line; increased shrub abundance, biomass, and cover; increased plant canopy heights; and decreased prevalence of bare ground (Summary for Policymakers in IPCC 2014, p. 32; Chapter 28 in IPCC 2014, p. 12). Vegetation changes are likely accelerated near coastlines, where red knots breed, due to the loss of sea ice that currently cools the adjacent land (Bhatt *et al*. 2010, pp. 1–2). Loss of sea ice may also make the central Canadian island habitats more maritime-dominated and, therefore, less suitable for breeding shorebirds (Meltofte *et al.* 2007, p. 36). The red knot's breeding range is also experiencing changes in freshwater wetland foraging habitats (Meltofte *et al.* 2007, p. 35; ACIA 2005, p. 418), as well as unpredictable but profound ecosystem changes (e.g., changing interactions among predators, prey, and competitors) (Meltofte *et al.* 2007, p. 35). The IPCC notes early warning signs that arctic ecosystems are already experiencing irreversible regime shifts (Summary for Policymakers in IPCC 2014, p. 12). We conclude that ecosystem changes in the Arctic are already underway and likely to continue, and that arctic ecosystems likely face much greater future change that may be abrupt, irreversible, or both. The red knot's adaptive capacity to withstand these changes in place, or to shift its breeding range northward, is unknown (also see Factor B, and Cumulative Effects, below).

Factor A—Conservation Efforts

We are unaware of any broad-scale conservation measures to reduce the threat of destruction, modification, or curtailment of the red knot's habitat or range. Specifically, no conservation measures are specifically aimed at reducing sea level rise or warming conditions in the Arctic. As described in the sections above, shorebird reserves have been established at several key red knot sites in South America, and regional efforts are in progress to develop and implement urban development plans to help protect red knot habitats at some of these sites. For example, a shorebird conservation plan is being implemented for Chiloé Island on the Pacific coast of Chile (WHSRN 2014d). Work is underway to establish a new national park on the Joulter Cays, a group of small uninhabited islands and flats that support red knots in the Bahamas (National Audubon Society (NAS) 2014). In the United States, the Service is working with partners to minimize the effects of shoreline stabilization on shorebirds and other beach species (e.g., Rice 2009, entire), and there are efforts in Delaware Bay to maintain horseshoe crab spawning habitat (and, therefore, red knot foraging habitat) via beach nourishment (e.g., Niles *et al.* 2013a, entire; Niles *et al.* 2013b, entire; USACE 2012, entire; Kalasz 2008, entire). At some key U.S. stopovers, including the Atlantic coast of Virginia, Delaware Bay, and Cape Cod, considerable habitat is in public or private conservation ownership. Delaware has improved and increased red knot roosting habitat through impoundment management, and has conducted adaptive planning to increase impoundment resiliency to climate change and sea level rise (DDNREC 2013). In addition, local or regional efforts are ongoing to control several species of invasive beach vegetation. While additional best management practices could be implemented to address shoreline development and stabilization, beach cleaning, invasive species, agriculture, and aquaculture, we do not have any information that specific, large-scale actions are being taken to address these concerns such that those efforts would benefit red knot populations or the subspecies as a whole. See Factor D regarding regulatory mechanisms relevant to coastal development, shoreline stabilization, beach cleaning, and invasive species.

Factor A—Summary

Within the nonbreeding portion of the range, red knot habitat is primarily threatened by the highly interrelated effects of sea level rise, shoreline stabilization, and coastal development. The primary red knot foraging habitats, intertidal flats and sandy beaches, will likely be locally or regionally inundated as sea levels rise, but replacement habitats are likely to re-form along eroding shorelines in their new positions (CCSP 2009b, p. 186; Scavia *et al.* 2002, p. 152). However, if shorelines experience a decades-long period of rapid sea level rise, high instability, and landward migration, the formation rate of new foraging habitats may be slower than the rate at which existing habitats are lost (Iwamura *et al.* 2013, p. 6). In addition, low-lying and narrow islands (e.g., in the Caribbean, along the Gulf and Atlantic coasts) may disintegrate rather than migrate, representing a net loss of red knot habitat (Chapter 5 in IPCC 2014, p. 15; Titus 1990, p. 67).

Superimposed on changes from sea level rise are widespread human efforts to stabilize the shoreline, which are known to exacerbate losses of intertidal habitats by blocking their landward migration. About 40 percent of the U.S. coastline within the range of the red knot is already developed, and much of this developed area is stabilized by a combination of existing hard structures and ongoing beach nourishment programs (Rice 2012a, p. 6; Titus *et al.* 2009, p. 5). Hard stabilization structures and dredging degrade and often eliminate existing intertidal habitats, and in many cases prevent the formation of new shorebird habitats (CCSP 2009b, pp. 99–100; Nordstrom 2000, pp. 20, 98–107). Beach nourishment may temporarily maintain suboptimal shorebird habitats where they would otherwise be lost as a result of hard structures or sea level rise (Nordstrom and Mauriello 2001, entire), but beach nourishment can also have adverse effects to red knots and their habitats (Rice 2009, entire; Peterson *et al.* 2006, entire; Peterson and Bishop 2005, entire; Greene 2002, p. 5). In those times and places where artificial beach maintenance is abandoned (e.g., due to constraints on funding or sediment availability), the remaining alternatives available to coastal communities would likely be limited to either a retreat from the coast or increased use of hard structures to protect development (CCSP 2009b, p. 87; Defeo *et al*. 2009, p. 7). The quantity of red knot habitat would be markedly decreased by a proliferation of hard structures. Red knot habitat would be significantly increased by retreat, but only where hard stabilization structures do not exist or where they get dismantled. Relative to the United States, little is known about development-related threats to red knot nonbreeding habitat in other countries. However, in some key international wintering and stopover sites, development pressures are likely to exacerbate habitat impacts caused by sea level rise (CSRPN 2013; WHSRN 2012; Niles *et al.* 2008, pp. 17, 19, 73, 97–98; Ferrari *et al.* 2002, p. 39).

Lesser threats to nonbreeding habitat include beach cleaning, invasive vegetation, agriculture, and aquaculture. The practice of intensive beach raking may cause physical changes to beaches that degrade their suitability as red knot habitat (Defeo *et al.* 2009, p. 4; Nordstrom and Mauriello 2001). Although not a primary cause of habitat loss, invasive vegetation can be a regionally important contributor to the overall loss and degradation of the red knot's nonbreeding habitat (USFWS 2012a, p. 27; Defeo *et al.* 2009, p. 6). Agriculture and aquaculture are a minor but locally important contributor to overall loss and degradation of the red knot's nonbreeding habitat, particularly for moderate numbers of red knots that winter or stopover in Northeast Brazil where habitats were likely impacted by the rapid expansion of shrimp farming since 1998 (Carlos *et al.* 2010, entire).

Within the breeding portion of the range, the primary threat to red knot habitat is from climate change. With arctic warming, vegetation conditions on the breeding grounds are changing, which is expected to eventually cause the zone of nesting habitat to shift north and contract (Feng *et al.* 2012, pp. 1359, 1366; Meltofte *et al.* 2007, p. 35; ACIA 2005, pp. 991, 998). Studies have already documented changes in arctic vegetation (e.g, increases in peak "greenness" and plant biomass; advancing of the arctic tree line; increased shrub abundance, biomass, and cover; increased plant canopy heights; and decreased prevalence of bare ground (Summary for Policymakers in IPCC 2014, p. 32; Chapter 28 in IPCC 2014, p. 12)). Vegetation effects are likely exacerbated by loss of sea ice (Bhatt *et al*. 2010, pp. 1–2l; Meltofte *et al.* 2007, p. 36). Arctic freshwater systems, foraging areas for red knots during the nesting season, are particularly sensitive to climate change and are already being affected (ACIA 2005, p. 1012; Meltofte *et al.* 2007, p. 35). Unpredictable but profound ecosystem changes (e.g., changing interactions among predators, prey, and competitors) are also likely to occur. The IPCC acknowledges early warning signs that arctic ecosystems are already experiencing irreversible regime shifts (Summary for Policymakers in IPCC 2014, p. 12). We conclude that ecosystem changes in the Arctic are already underway and likely to continue, and that arctic ecosystems likely face much greater future change that may be abrupt and irreversible. Further, climate change is opening the Arctic to development such as oil and gas exploration, commercial shipping, tourism, fishing, and other activities (Niles 2013; NRC 2013, p. 4; Smith and Stephenson 2013, p. 2; Astill 2012; Roach 2007).

Threats to the red knot from habitat destruction and modification are occurring throughout the entire range of the subspecies. These threats include climate change, shoreline stabilization, and coastal development, exacerbated regionally or locally by lesser habitat-related threats such as beach cleaning, invasive vegetation, agriculture, and aquaculture. The subspecies-level impacts from these activities are expected to continue into the future.

FACTOR B. OVERUTILIZATION FOR COMMERCIAL, RECREATIONAL, SCIENTIFIC, OR EDUCATIONAL PURPOSES

In this section, we discuss historic shorebird hunting in the United States that caused a substantial red knot population decline, ongoing shorebird hunting in parts of the Caribbean and South America, and potential effects to red knots from scientific study.

Factor B—Hunting

Since the late 19th century, hunters concerned about the future of wildlife and the outdoor tradition have made countless contributions to conservation. In many cases, managed hunting is an important tool for wildlife management. However, unregulated or illegal hunting can cause population declines, as was documented in the 1800s for red knots in the United States. While no longer a concern in the United States, underregulated or illegal hunting of red knots and other shorebirds is ongoing in parts of the Caribbean and South America.

Hunting—United States (Historical)

Red knots were heavily hunted for both market and sport during the 19th and early 20th centuries (Harrington 2001, p. 22) in the Northeast and the mid-Atlantic. Red knot population declines were noted by several authors of the day, whose writings recorded a period of intensive hunting followed by the introduction of regulations and at least partial population recovery. As early as 1829, Wilson (1829, p. 140) described the red knot as a favorite among hunters and bringing a good market price. Giraud (1844, p. 225) described red knot hunting in the South Bay of Long Island. Noting confusion over species common names, Roosevelt (1866, pp. 91–96) reported that hunting of "bay snipe" (a name applied to several shorebird species including red knot) primarily occurred from Cape Cod to New Jersey, rarely south of Virginia. Specific to red knots, Roosevelt (1866, p. 151) noted they were "killed indiscriminately. . . with the other bay-birds." Hinting at shorebird population declines, Roosevelt (1866, pp. 95–96) found that "the sport [of bay snipe shooting] has greatly diminished of late . . . a few years ago . . . it was no unusual thing to expend twenty-five pounds of shot in a day, where now the sportsman that could use up five would be fortunate."

Mackay (1893, p. 29) described a practice on Cape Cod during the 1850s called "fire-lighting," involving night-time hand-harvest via lantern light. In just one instance, "six barrels" of red knots taken by fire-lighting were shipped to Boston (Mackay 1893, p. 29), leading to estimates of 25,000 knots killed in a single year (Harrington *et al.* 2010a, p. 188). Fire-lighting continued "several years" before it was banned (Mackay 1893, p. 29). Red knots continued to be taken "in large numbers on the Atlantic seaboard (Virginia)...one such place shipping to New York City in a single spring, from April 1 to June 3, upwards of six thousand Plover, a large share of which were Knots" (Mackay 1893, p. 30). Mackay (1893, p. 30) concluded that red knots were "in great danger of extinction."

Shriner (1897, p. 94) reported, "This bird was formerly very plentiful in migrations in New Jersey, but it has been killed off to a great extent, proving an easy prey for pothunters," and Eaton (1910, p. 94) described red knots as "much less common than formerly." Echoing Mackay (1893), Forbush (1912, pp. 262–266) cited numerous sources in describing a substantial coastwide decline in red knot numbers, and concluded, "The decrease is probably due…to shooting both spring and fall all along our coasts, and possibly to some extent in South America…its extirpation from the Atlantic coast of North America is [possible] in the near future."

By 1927, Bent (1927, p. 132) noted signs of red knot population recovery, "Excessive shooting, both in spring and fall reduced this species to a pitiful remnant of its former numbers; but spring shooting was stopped before it was too late and afterwards this bird was wisely taken off the list of game birds; it has increased slowly since then, but is far from abundant now." Urner and Storer (1949, pp. 192–193) reached the same conclusion, and documented population increases along New Jersey's Atlantic coast from 1931 to 1938. Based on his bird studies of Cape May, New Jersey, Stone (1937, p. 465) concluded that the red knot population decline had not been as sharp as previously thought, and that "since the abolishing of the shooting of shore birds it has steadily increased in abundance." It is unclear whether the red knot population fully recovered its historical numbers (Harrington 2001, p. 22) following the period of unregulated hunting, and it is possible this episode reduced the species' resilience to face other threats that emerged over the course of the 20th century. This era of intensive hunting has also been implicated as one factor, along with coastal development, that may have led to changing use of spring stopover areas along the Atlantic coast (Cohen *et al.* 2008). However, legal hunting of red knots is no longer allowed in the United States, and there is no indication of illegal hunting from any part of its mainland U.S. range.

Hunting—Caribbean and South America (Current)

Both legal and illegal sport and subsistence hunting of shorebirds takes place in several known red knot wintering and migration stopover areas. This analysis focuses on areas where both red knots and hunting are known to occur, although in many areas we lack specific information regarding levels of red knot mortality from hunting. Therefore, we document the activity and explain that red knots could be affected, but draw no conclusions about direct mortality unless specifically noted.

As described above (Breeding Chronology and Success; Migration—Fall Timing and Distribution), red knots are known to segregate by sex during migration. The effects of hunting would be far greater if mortality disproportionately affects adult females (D. Mizrahi pers. comm. October 16, 2011), which may predominate red knot aggregations at certain times of the year. In addition, we have little information regarding juvenile red knot wintering areas or survival rates (see Longevity and Survival; Wintering—Juveniles). To the extent that juveniles may occur in areas where shorebirds are hunted, it is possible that hunting could impact the rates of recruitment into the adult population, as juveniles could be more vulnerable to hunting pressure than adults (DDNREC 2013), (e.g., if juveniles spend a greater percentage of their annual cycle in regions where shorebirds are hunted, if juveniles are naïve to hunting, or both (K. Kalasz pers. comm. August 13, 2014)).

Moving from north to south, hunting is known from the Bahamas, including Andros, but it is not known if shorebirds specifically are hunted (B. Andres pers. comm. December 21, 2011); red knot hunting is prohibited by law (see Factor D). Likewise, hunting is considered a general threat to birds in Cuba but no specific information is available (B. Andres pers. comm. December 21, 2011). Regulated sport hunting occurs in Jamaica, but red knots are among the protected bird species for which hunting is prohibited in that country's wildlife law. Hunting occurs in Haiti, but information is not available specific to shorebirds (B. Andres pers. comm. December 21, 2011). United States laws including the Endangered Species Act (regulating take of listed species) and the Migratory Bird Treaty Act (MBTA) (regulating harvest of migratory birds) apply in Puerto Rico and the U.S. Virgin Islands. In Puerto Rico, hunting is strictly regulated and permitted only for certain species, but enforcement is lacking and nonlicensed hunters outnumber legal hunters. In the U.S. Virgin Islands, unregulated legal hunting, as well as poaching, has extirpated the West Indian whistling-duck (*Dendrocygna arborea*) (B. Andres pers. comm. December 21, 2011). General enforcement of hunting regulations is lacking in the U.S. Virgin Islands, but shorebird hunting is negligible (B. Andres pers. comm. February 5, 2013 and December 21, 2011).

Hunting birds is popular in Trinidad and Tobago. Seabird colonies are threatened by poachers who collect the adult birds for meat and presumably also take the eggs. In addition to seabirds, species at particular risk from hunting include several species of wading birds, fowl, and waterfowl (B. Andres pers. comm. December 21, 2011). Although hunters generally target larger waterbirds, harvest is a threat to shorebirds as well. There are about 750 hunters (on both Trinidad and Tobago), the season ranges from November to February, and there are no bag limits (USFWS 2011e, p. 4). Red knot hunting is prohibited by law in Belize and Uruguay. Reports suggest that 29 of 178 waterbirds may be threatened with hunting in Venezuela, but we lack any data specific to shorebird hunting (B. Andres pers. comm. December 21, 2011).

Current Hunting—Lesser Antilles Shooting Swamps

In parts of the Lesser Antilles, legal sport hunters target shorebirds in "shooting swamps." Most of the migratory shorebird species breeding in eastern North America and the Arctic pass through the Caribbean during late August and September on their way to wintering areas. When they encounter severe storms during migration, the birds use the islands as refuges before moving on to their final destinations. Hunting clubs take advantage of these events to shoot large numbers of shorebirds at one time (Nebel 2011, p. 217).

Lesser Antilles—Barbados

Barbados has a tradition of legal shorebird hunting that began with the colonists in the 17th and 18th centuries. The current shooting swamps were artificially created and can attract large numbers of migrant shorebirds during inclement weather. The open season for shorebirds is July 15 to October 15, and there is no daily bag limit (USFWS 2011e, p. 2). As of 1991, Hutt (pp. 77–78) estimated that fewer than 100 hunters killed 15,000 to 20,000 shorebirds per year at 7 major shooting swamps. Although conservation progress has been made, the number of shorebirds killed annually is still around 26,000. Hunters have an agreement with the conservation community to lower the annual shorebird harvest to 22,500 (McClain 2013; Eubanks 2011). The Barbados Wildfowlers Association has passed a series of self-imposed regulations, including annual and daily limits per swamp, and a limit of three active hunters per swamp (McClain 2013). Several species are protected, and hunters have voluntarily agreed to stop the harvest of red knots. Work is in progress to gather current mortality levels and develop a model of sustainable shorebird harvest. To date, half of the shooting swamps on Barbados have agreed to furnish harvest data (USFWS 2011e, p. 2).

Although hunting pressure on shorebirds remains high, red knots have not been documented in Barbados in large numbers. The red knot is a regular fall transient, usually occurring as single individuals and in small groups in late August and early September, and typically utilizing coastal swamps during adverse weather (Hutt and Hutt 1992, p. 70; Hutt 1991, p. 89). Detailed records from 1950 to 1965 show an average of about 20 red knots per year. Red knots may occur very exceptionally in flocks of up to a dozen birds; a record of 63 birds—brought in by a storm—were shot in 1 day in 1951 (Hutt and Hutt 1992, p. 70). From 1990 to 1992, seven shooting swamps were active, and red knot mortality was reported from two of the swamps; nine red knots were shot at Best Pond, and one was shot at Woodbourne. Due to its coastal location, Best Pond attracted more red knots than other shooting swamps, but it has been closed to hunting due to residential development (W. Burke pers. comm. October 12, 2011), and Woodbourne has been restored as a "no-shoot" shorebird refuge (BirdLife International 2009; Burke 2009, p. 287). The remaining shooting swamps in Barbados no longer target red knots, and only a few knots have been observed in recent years (W. Burke pers. comm. October 12, 2011).

Lesser Antilles—French West Indies

The French West Indies consist of Guadeloupe and its dependencies, Martinique, Saint Martin, and Saint Barthélemy. A 1976 account notes that the red knot was not well known by hunters in the French Antilles, and did not even have a creole name. Until 1950, red knot was catalogued as rare, if not exceptional in the West Indies. Since then, however, red knots have been noted to pass through the islands at least occasionally. By dozens (10s or 20s) some red knots have been killed in the wetlands of the French West Indies for approximately the last 15 years (G. Humbert pers. comm. November 29, 2013).

Like Barbados, legal sport hunting of shorebirds has a long tradition on the French territories of Guadeloupe and Martinique (USFWS 2011e, p. 3). Wetlands are not managed for shorebird hunting in Guadeloupe, but are sometimes on Martinique (USFWS 2011e, p. 3). However, Guadeloupe has several isolated mangrove swamps that serve to concentrate shorebirds for shooting (Nebel 2011, p. 217). Approximately 1,400 hunters on Martinique and 3,000 hunters on Guadeloupe harvest 14 to 15 shorebird species, which are typically eaten. The hunting season runs from July to January, and no daily bag limits are set. The shorebird hunting pressure in the French West Indies may be greater than on Barbados. There are no reliable estimates for the magnitude of the harvest; however, a single hunter has been known to harvest 500 to 1,000 shorebirds per season. Work is ongoing to more accurately determine the magnitude of the shorebird harvest in the French West Indies (USFWS 2011e, p. 3).

Although shorebird hunting had been previously documented on Guadeloupe (USFWS 2011e, p. 3), the issue gained notoriety in September 2011 when two whimbrels (*Numenius phaeopus*), fitted with satellite transmitters as part of a 4-year tracking study, were killed by hunters. The 2 birds were the first of 17 tracked whimbrels to stop on Guadeloupe; they were not migrating together, but both stopped on the island after encountering different storm systems. As both whimbrels were shot in a known shooting swamp within hours of arriving on Guadeloupe, the circumstances of these two documented mortalities suggest that shorebird hunting pressure may be very high (Smith *et al*. 2011b). Like other overseas territories, Guadeloupe is not covered by key European laws for biodiversity conservation (Nebel 2011, p. 217). Following the shooting of the tracked whimbrels, conservation groups launched an appeal for the protection of birds and their habitats in French overseas departments in the Caribbean and elsewhere (Nebel 2011, p. 217). The French Government has recently acted to impose new protective measures in Guadeloupe. The National Hunting and Wildlife Agency has implemented a bag limit of 20 birds per day and is working on a regulation that would stop hunting for 5 days following a tropical storm warning (McClain 2013; A. Levesque pers. comm. January 8, 2013; Niles 2012c). Significantly, the red knot was officially added to the list of protected species in Guadeloupe and Martinique on August 15, 2013. In addition, hunter education about red knots is in progress (G. Humbert pers. comm. November 29, 2013; A. Levesque pers. comm. January 8, 2013; Niles 2012c).

Although the red knot was until recently listed as a game bird, mortality from hunting was probably low because red knots occur only in small numbers. In Guadeloupe, the red knot is an uncommon but regular visitor during fall migration, typically in groups of 1 to 3 birds, but as many as 16 have been observed in 1 flock. Probably no more than a few dozen red knots were shot per year in Guadeloupe (A. Levesque pers. comm. October 11, 2011), prior to its protected designation. Likewise, while any mortality of red knots prior to their protection on Martinique is unknown, it is likely very low due to the very small numbers of knots reported there. Although not as regular in Martinique as in Guadeloupe, several recent sightings are known from there, including two fall eBird records (G. Humbert pers. comm. November 29, 2013). We have no records of red knots occurring on Saint Martin or Saint Barthélemy (eBird.org 2014).

Current Hunting—The Guianas

Recent evidence suggests at least 2,000 red knots pass through the Guianas (i.e., Suriname, Guyana, and French Guiana, see figure 5) during southbound migration (NJAS *et al.* 2013). Red knots are primarily passage migrants in the Guyanas, with many more birds documented in French Guiana (Niles 2012b) than in Suriname, where the habitat is not ideal for red knots (B. Harrington pers. comm. March 31, 2006; Spaans 1978, p. 72). However, there are sandy beach habitats in Suriname's Marowijne District, less than 31 mi (50 km) from the site where approximately 1,700 red knots were observed in French Guiana, and several hundred knots were observed using coastal mud flat habitat in the Commewijne district of Suriname during spring migration, April 2012 (NJAS *et al.* 2013).

Hunting is the most direct threat to shorebirds in this region (Ottema and Spaans 2008, p. 343). Band recoveries indicate that red knots are killed commonly for food in some regions of South America, especially in the Guianas. The overall take from these activities is unknown, but the number of band recoveries (about 17) in the Guianas hints that the take may be substantial

(Harrington 2001, p. 22). More recently two additional bands were recovered from red knots shot in French Guiana (D. Mizrahi pers. comm. October 16, 2011). One of these birds, shot in a rice field near Mana in May 2011, was banded in Delaware Bay in May 2005 and was subsequently resighted over 30 times in New Jersey, Delaware, and Florida (J. Parvin pers. comm. September 12, 2011).

Hunting is the primary threat to red knots in French Guiana, where hunting of all species of "peep" shorebirds occurs freely along the coast (CSRPN 2013). Hunting in French Guiana is mostly known from Mana, in the rice fields and on the beaches where massive shorebird (*Calidris* and *Tringa* spp.) flocks gather at high tides, immediately adjacent to the Amana Natural Reserve. Red knots are not particularly targeted by hunters, but may be killed along with all other shorebird species. In addition, disturbance due to the presence of hunters and noise of fired shots may be high (CSRPN 2013). Rice fields and other impoundments are prevalent in French Guiana as well as Guyana (USFWS 2011e, p. 3). In the rice fields near Mana, French Guiana, more than 1,700 red knots were observed in late August 2012 (CSRPN 2013; Niles 2012b). During the same timeframe, about 30 new shotgun shells per kilometer were collected along the dikes around the fields. This estimated density of spent shotgun shells is a minimum as some of the dikes were swept by the tides and most were overgrown with vegetation, limiting detectability. In addition to observing the indirect evidence of hunting, researchers saw two people with guns during 4 days in the field (Niles 2012b). Shorebirds are harvested legally in French Guiana and Guyana (USFWS 2011e, p. 3). As of 2013, shorebird hunting was unregulated in French Guiana (A. Levesque pers. comm. January 8, 2013; D. Mizrahi pers. comm. October 16, 2011). However, a ban on hunting all species of Charadriiformes has been proposed in French Guiana (CSRPN 2013), and the red knot was designated a protected species in October 2014(C. Carichiopulo and N. de Pracontal pers. comm. October 10, 2014). See Factor D.

Harvest of any shorebirds has been illegal in Suriname since 2002, but there is little enforcement (USFWS 2011e, p. 3; Ottema and Spaans 2008, p. 344). Law enforcement is hampered by limited resources (e.g., working boats, gasoline), and several tens of thousands of shorebirds are trapped and shot each year. A 2006 survey indicated that virtually all shorebird species occurring in Suriname were illegally hunted and trapped in some quantity, with the lesser yellowlegs (*Tringa flavipes*) and semipalmated sandpiper (*Calidris pusilla*) being the dominant species (USFWS 2011e, p. 3). The survey estimated that between 20,000 and 100,000 shorebirds are taken annually in Suriname (NJAS *et al.* 2013). The survey also documented an illegal food trade of shorebirds, including selling to local markets (USFWS 2011e, p. 3; Ottema and Spaans 2008, p. 344). Shorebirds are harvested by shooting, netting, and using choke wires. Many shorebirds are taken by Guyanese fishermen working in Suriname. The Suriname coast is mainly mudflats and much of the coast is legally protected. Three coastal areas in Suriname are designated as sites of hemispheric importance by WHSRN, and it is likely that hunting occurs in at least two of them. Education and awareness programs have begun along the coast of Suriname, and a hunter training program is being developed (USFWS 2011e, p. 3).

The magnitude of the harvest in French Guiana and Guyana is unknown (CSRPN 2013; USFWS 2011e, p. 3), but has been roughly estimated at a few tens of birds each year in French Guiana (CSRPN 2013). Based on work in Suriname and French Guiana since 2008, D. Mizrahi (pers. comm. October 16, 2011) suspects that red knot mortality from hunting in these countries may be an order of magnitude higher than in Guadeloupe, given the much larger stopover populations (i.e., hundreds of birds) that have been observed in the Guianas. While only moderate red knot mortality is documented, additional undocumented mortality is likely. The findings of Watts (2010, p. 39) suggest that even moderate (hundreds of birds) direct human-caused mortality may begin to have population-level effects on the red knot. We do not have reliable information to reasonably know if hunting mortality is or was previously at this level in the Guianas. We expect mortality of individual knots from hunting in the Guianas to continue into the future, but at stable or decreasing levels due to the recent international attention to shorebird hunting.

Current Hunting—Brazil

Hunting migratory shorebirds for food was previously common among local communities in Maranhão, Brazil. Shorebirds provided an alternative source of protein, and birds like the red knot with high subcutaneous fat content for long migratory flights were particularly valued. According to local people, red knot was among the most consumed species, although no data are available to document the number of birds taken. Local people say that, although some shorebirds are still hunted, this practice has greatly decreased over the past decade, and hunting is not thought to amount to a serious cause of mortality (Niles *et al.* 2008, p. 99). Nonetheless, hunting remains a threat in Maranhão (L. Niles pers. comm. November 19, 2013). Outside the State of Maranhão, hunting pressure on red knots has not been characterized. For some bird species, unregulated subsistence hunting in Brazil may be causing species declines (R. Huffines pers. comm. September 13, 2011).

Commercial and recreational hunting are prohibited in all Brazilian territory, except for the state of Rio Grande do Sul, which includes the Logoa do Peixe stopover site. The Rio Grande do Sul hunting law provides a list of animals that can be hunted, prohibits trapping, and bans commercialized hunting (B. Andres pers. comm. December 21, 2011). Poaching is known from waterbird colonies in Brazil (B. Andres pers. comm. December 21, 2011), but no information is available regarding any illegal shorebird harvest.

Factor B—Scientific Study

Aerial and ground surveys are conducted in many parts of the red knot's range. In addition, about 1,000 red knots per year are trapped for scientific study in Delaware Bay, and about 300 in South America (Niles *et al.* 2008, p. 100). In some years, additional birds are trapped in other parts of the range (e.g., Newstead *et al.* 2013, pp. 54–55; Schwarzer *et al.* 2012, p. 728; Baker *et al.* 2005, p. 13). In an effort to further understand the red knot's rates of weight gain, migratory movements, survival rates, and conservation needs, the trapped birds are weighed and measured, leg-banded, and fitted with individually numbered color-flags. In some years, coordinated tissue sampling (e.g., feathers, blood, mouth swabs) is conducted for various scientific studies (Niles *et al.* 2008, p. 100), such as contaminants testing, stable isotope analysis, or genetic research.

Temporary, localized disturbances occur during aerial surveys, as well as during capture via a "cannon" netting procedure in which a hidden net is propelled over a flock of foraging birds. Prolonged captivity or excessive handling during banding operations can cause *Calidris canutus* to rapidly lose weight, about 0.04 ounces (oz) (1 gram (g)) per hour (L. Niles and H. Sitters pers. comm. September 4, 2008; Davidson 1984, p. 1724). In rare circumstances, *C. canutus* held in captivity during banding, especially when temperatures are high, can develop muscle cramps that can be fatal or leave birds vulnerable to predators (Rogers *et al.* 2004, p. 157; Clark and Clark 2002, p. 49).

Through 2008, about 50 of the birds caught in Delaware Bay each year were the subject of radiotelemetry studies in which a 0.1-oz (2-g) radio tag was glued to the back of each bird (Niles *et al.* 2008, p. 100). Additional birds were recently radio-tracked in Texas (D. Newstead pers. comm. August 20, 2012). The tags are expected to drop off after 1 to 2 months through the natural replacement of skin. Resighting studies in subsequent years showed that the annual survival of radio-tagged birds was no different from that of birds that had only been banded (Niles *et al.* 2008, p. 100). In more recent years, tens of red knots have been fitted with geolocators. After 1 year, researchers found no significant differences in the resighting rates of birds carrying geolocators, suggesting that these devices did not affect survival (Niles *et al.* 2010a, p. 123).

Considerable care is taken to minimize disturbance caused to shorebirds from these research activities. Aerial surveys do flush foraging birds, which is necessary for species identification; however, this disturbance is very short-lived. Ground surveys are typically conducted slowly and from a distance (e.g., using spotting scopes) or behind a dune to minimize disturbance. Cannon netting is done by experienced personnel following a written manual, as well as state and Federal (e.g., Migratory Bird Treaty Act) scientific collecting permits for wildlife. Captured birds are processed (e.g., weighed, measured, banded) away from the capture site to avoid further disturbance to the uncaptured members of the flock. Standard practices limit the number of captures from any particular area, and numbers of birds per catch and total numbers caught over a season are limited. Careful handling protocols (e.g., adequate shade, proper holding cages) are followed, including a 3-hour limit on holding times (Niles *et al*. 2010a, p. 124; L. Niles and H. Sitters pers. comm. September 4, 2008; Niles *et al.* 2008). Despite these measures, hundreds of red knots are temporarily stressed during the course of annual research, and mortality, though rare, does occasionally occur (K. Clark pers. comm. January 21, 2013; Taylor 1981, p. 241). However, we conclude that these research activities are not a threat to the red knot because evaluations have shown no effects of these short-term stresses on red knot survival. Further, the rare, carefully documented, and properly permitted mortality of an individual bird in the course of well-founded research does not affect red knot populations or the overall subspecies.

Factor B—Conservation Efforts

As discussed above, a few countries where shorebird hunting is legal have implemented voluntary restrictions on red knot hunting, increased hunter education efforts, established "no-shoot" shorebird refuges, and are developing models of sustainable harvest. Hunting regulations have been adopted or proposed in several areas (see Factor D). Ongoing scientific research has benefitted red knot conservation in general and, through leg-band recoveries, has provided documentation of hunting-related mortality. Research activities adhere to best practices for the careful capture and handling of red knots.

Factor B—Summary

Legal and illegal sport and market hunting in the mid-Atlantic and Northeast United States substantially reduced red knot populations in the 1800s, and we do not know if the subspecies ever fully recovered its former abundance or distribution (Karpanty *et al.* 2014, p. 2; Cohen *et al.* 2008; Harrington 2001, p. 22). Neither legal nor illegal hunting are currently a threat to red knots in the United States, but both occur in the Caribbean and parts of South America (Harrington 2001, p. 22). Hunting pressure on shorebirds in the Lesser Antilles (e.g., Barbados, Guadeloupe) is very high (USFWS 2011e, pp. 2–3), but only small numbers of red knots have been documented on these islands, so past mortality may not have exceeded tens of birds per year (G. Humbert pers. comm. November 29, 2013). Red knots are no longer being targeted in Barbados or Guadeloupe, and other measures to regulate shorebird hunting on these islands are being negotiated (G. Humbert pers. comm. November 29, 2013; McClain 2013; USFWS 2011e, p. 2). Much larger numbers (thousands) of red knots occur in the Guianas, where legal and illegal subsistence shorebird hunting is common (CSRPN 2013; Niles 2012b; Ottema and Spaans 2008, p. 343). About 20 red knot mortalities have been documented in the Guianas (D. Mizrahi pers. comm. October 16, 2011; Harrington 2001, p. 22), but total red knot hunting mortality in this region cannot be surmised. As of 2013, shorebird hunting was unregulated in French Guiana (A. Levesque pers. comm. January 8, 2013; D. Mizrahi pers. comm. October 16, 2011). However, a ban on hunting all shorebird species has been proposed in French Guiana (CSRPN 2013) and the red knot was designated a protected species in October 2014 (C. Carichiopulo and N. de Pracontal pers. comm. October 10, 2014). Subsistence shorebird hunting was also common in northern Brazil, but has decreased in recent decades (Niles *et al.* 2008, p. 99).

We have no evidence that hunting was a driving factor in red knot population declines in the 2000s, or that hunting pressure is increasing. While only low to moderate red knot mortality is documented, additional undocumented mortality is likely. The findings of Watts (2010, p. 39) suggest that even moderate (hundreds of birds) direct human-caused mortality may begin to have population-level effects on the red knot. We do not have reliable information to reasonably know if hunting mortality is or was previously at this level in the Guianas, though we conclude it was likely much lower (tens of birds) in the Caribbean islands. In contrast, catch limits, handling protocols, and studies on the effects of research activities on survival all indicate that overutilization for scientific purposes is not a threat to the red knot (Niles *et al*. 2010a, p. 124; L. Niles and H. Sitters pers. comm. September 4, 2008; Niles *et al.* 2008, p. 100).

Threats to the red knot from overutilization for commercial, recreational, scientific, or educational purposes exist in parts of the Caribbean and South America. Specifically, legal and illegal hunting do occur. We expect mortality of individual knots from hunting to continue into the future, but at stable or decreasing levels due to the recent international attention to shorebird hunting, and due to new voluntary and regulatory hunting restrictions in some areas (as discussed above and under Factor D) .

FACTOR C. DISEASE OR PREDATION

Red knots are exposed to several diseases and experience variable rates of predation from avian and mammalian predators throughout their range. In this section, we discuss known parasites and viruses, and the direct and indirect effects of predation in the red knot's breeding, wintering, and migration areas.

Factor C—Disease

Red knots are exposed to parasites and disease throughout their annual cycle. Susceptibility to disease may be higher when the energy demands of migration have weakened the immune system. Studying red knots in Delaware Bay in 2007, Buehler *et al.* (2010, p. 394) found that several indices of immune function were lower in birds recovering protein after migration than in birds storing fat to fuel the next leg of the migration. These authors hypothesized that fueling birds may have an increased rate of infection or may be bolstering immune defense, or recovering birds may be immuno-compromised because of the physical strain of migratory flight or as a result of adaptive energy tradeoffs between immune function and migration, or both (Buehler *et al.* 2010, p. 394). A number of known parasites and viruses are described below, but we have no evidence that disease is a current threat to the red knot.

Disease—Parasites

An epizootic disease (epidemic simultaneously affecting many animals) that caused illness or death of about 150 red knots on the west coast of Florida in December 1973 and November 1974 was caused by a protozoan (single-celled organism) parasite, most likely an undescribed sporozoan (reproducing by spores) species (USFWS 2003, p. 22; Harrington 2001, p. 21; Woodward *et al.* 1977, p. 338).

On April 7, 1997, 26 red knots, 10 white-rumped sandpipers (*Calidris fuscicollis*), and 3 sanderlings were found dead or dying along 6.2 mi (10 km) of beach at Lagoa do Peixe in southern Brazil. The following day, another 13 dead or sick red knots were found along 21.7 mi (35 km) of nearby beach (Niles *et al.* 2008, p. 101; Baker *et al.* 1998, p. 74). All 35 red knots were heavily infected with hookworms (Phylum Acanthocephala), which punctured the birds' intestines. Although hookworms can cause sudden deaths in birds, the lungs of some birds were discolored, suggesting there may have been an additional factor in their mortality. Three white-rumped sandpipers and three sanderlings were also examined, and none appeared to be infected with hookworms, again suggesting another cause of death. Bacterial agents, environmental contaminants, and red tide (see Factor E—Harmful Algal Blooms) were not ruled out (Baker *et al.* 1998, p. 75), but Harrington (2001, p. 21) attributed the deaths to the hookworms. Smaller mortalities of spring migrants with similar symptoms were also reported from Uruguay in the 2000s (Niles *et al*. 2008, p. 101).

Blood parasites represent a complex, spatially heterogeneous host-parasite system having ecological and evolutionary impacts on host populations. Three closely related genera, (*Plasmodium*, *Haemoproteus* and *Leucocytozoon*) are commonly found in wild birds, and infections in highly susceptible species or age classes may result in death (D'Amico *et al*. 2008, p. 195). Reported red knot mortalities in Florida in 1981 were attributed to the blood parasite

Plasmodium hermani (Niles *et al.* 2008, p. 101; Harrington 2001, p. 21). However, no blood parasites (*Plasmodium*, *Haemoproteus* or *Leucocytozoon* spp.) were found in red knots sampled in 2004 and 2005 in Tierra del Fuego (181 samples), Maranhão, Brazil (52 samples), or Delaware Bay (140 samples), and this finding is consistent with the generally low incidence of blood parasite vectors along marine shores (D'Amico *et al*. 2008, pp. 193, 197). No blood parasites (*Plasmodium* or *Haemoproteus* spp.) were detected in 156 red knots sampled at 2 sites in Argentina (Río Grande and San Antonio Oeste) in 2005 and 2006 (D'Amico *et al.* 2007, p. 794).

In 2008, Escudero *et al.* (2012, pp. 362–363) observed a high prevalence of a Digenea parasitic flatworm (*Bartolius pierrei*) in clams (*Darina solenoids*), a major prey item of red knots foraging at Río Grande in Argentinean Tierra del Fuego. Clams near the surface of the sediment were the most highly infected by the flatworm, and were preferentially eaten by red knots, probably due to their larger size. While digenean worm parasites may be part of the natural intestinal fauna of red knots, parasites are detrimental by definition. It is likely that the adult stage of this parasite living in the intestines and stomach causes either damage or an immunological response, adversely affecting the condition of the host birds (Escudero *et al.* 2012, p. 363). Farther north, at Fracasso Beach, Península Valdés, Argentina, Cremonte (2004, p. 1591) found that *B. pierrei* uses the same clam *D. solenoides* as its intermediate host. The red knot and a gull species (Family Laridae) act as definitive hosts, with 92 percent of red knots infected. *Bartolius pierrei* did not parasitize other invertebrates that share the intertidal habitat with *D. solenoides*, suggesting this parasite may be adapted to target red knot prey species. *Bartolius pierrei* is an endemic parasite of the Magellan region, distributed where its intermediate clam host is present, from San José Gulf in Península Valdés to the southern tip of South America (Cremonte 2004, p. 1591). To date, the impacts of flatworm infection on red knot health or fitness have not been investigated.

Ectoparasites, which live on the surface of the body, can affect birds by directly hindering their success in obtaining food and by acting as vectors and invertebrate hosts to microorganisms. For example, lice and mites infest skin and feathers leaving their hosts susceptible to secondary infections (D'Amico *et al*. 2008, p. 195). Individual red knots examined in 1968 (New York) and 1980 (Massachusetts) were infested with bird lice (Mallophaga (Amblycera): Menoponidae), which live in the feather shafts. Based on the bird examined in 1980, the lice likely caused that red knot to molt some primary feathers, known as an adventitious molt. Other than the molt, this red knot appeared healthy (Taylor 1981, p. 241). In the course of ongoing field studies in Maranhão, Brazil, all 38 knots caught and sampled in February 2005 were found to be heavily infected with ectoparasites. The birds were also extremely lightweight, less than the usual fat-free mass of red knots (Baker *et al*. 2005, p. 15). Fieldworkers have also noticed ectoparasites on a substantial number of red knots caught in Delaware Bay (Niles *et al.* 2008, p. 101).

D'Amico *et al.* (2008, pp. 193, 197) examined red knots for ectoparasites at three sites in 2004 and 2005. All ectoparasites observed during this study were feather lice (Phthiraptera: Mallophaga (Amblycera)). Only 5 of 113 (4 percent) of red knots examined on Tierra del Fuego in Río Grande, Argentina, had ectoparasites, while all 36 knots (100 percent) examined in Maranhão, Brazil, were infected. Almost 40 percent of the Brazilian birds had very high parasite loads. Of 256 red knots examined in Delaware Bay, 174 (68 percent) had ectoparasites. Using feather isotopes from the Delaware Bay birds, D’Amico *et al.* (2008, p. 197) identified 90 of the 256 birds as coming from northern wintering areas (e.g., Brazil, the Southeast) and 66 from southern wintering areas (e.g., Tierra del Fuego) (the wintering region of the remaining 100 birds was unknown). The proportions of parasitized birds captured at Delaware Bay from the different wintering regions were not significantly different (50 percent from northern areas infected versus 40 percent from southern areas). However, the northern-wintering red knots tended to have higher loads of ectoparasites (i.e., more parasites per bird). These data suggest that many southern birds may be infected during a short stopover during the northward migration or by direct contact in Delaware Bay (D’Amico *et al.* 2008, pp. 193, 197). To date, the impacts of ectoparasite infection on red knot health or fitness have not been investigated.

Associating characteristics of breeding and wintering habitats, chick energetics, and apparent immunocompetence (the ability of the body to produce a normal immune response following exposure to disease), Piersma (1997, p. 623) suggested that shorebird species make tradeoffs of immune system function versus growth and sustained exercise. This author suggested that these tradeoffs determine the use of particular habitat types by long-distance migrating shorebirds. Some species appear restricted to parasite-poor habitats such as the Arctic tundra and exposed seashores, where small investments in the immune system may suffice and even allow for high chick growth rates. However, such habitats are few and far between, necessitating long and demanding migratory flights and often high energy expenditures while in residence (e.g., to deal with cold temperatures) (Piersma 1997, p. 623). Increased adult survival afforded by inhabiting areas of low parasite loads may offset the energetic and other costs of breeding in the climatically marginal, but parasite-low, Arctic (USFWS 2003, p. 22). Piersma’s (1997) parasite hypothesis predicts that red knots should evolve migrations to low-parasite marine wintering sites to reduce the fitness consequences of high ectoparasite loads in tropical Brazil, but there is likely a tradeoff with increased mortality for long-distance migration to cold-temperate Tierra del Fuego (D’Amico *et al.* 2008, p. 193).

Species adapted to parasite-poor habitats may be particularly susceptible to parasites and pathogens (USFWS 2003, p. 22; Piersma 1997, p. 623). For example, captive *Calidris canutus* are susceptible to common avian pathogens (e.g., the avian pox virus, bacterial infections, feather lice), and reconstructing a marine environment (i.e., flushing the cages with seawater) helps to reduce at least the external signs of infections (Piersma 1997, pp. 624–625).

In summary, three localized red knot die-off events have been attributed to parasites, but these kinds of parasites (sporozoans, hookworms) have not been documented elsewhere or implicated in further red knot mortality. Blood parasites have caused red knot deaths, but blood parasite infections were not detected by testing that took place across the knot’s geographic range in the 2000s. In contrast, flatworm infection is widespread in Argentina, and bird lice infection is widespread in tropical and temperate portions of the red knot’s range. However, impacts of these infections on red knot health or fitness have not been documented. Red knots may be adapted to parasite-poor habitats, and may, therefore, be particularly susceptible to parasites and pathogens. However, we have no evidence that parasites have impacted red knot populations beyond causing normal, background levels of mortality, and we have no indications that parasite infection rates or fitness impacts are likely to increase. Therefore, we conclude parasites are not a threat to the red knot.

Disease—Viruses

Type A influenza viruses, also called avian influenza (AI), are categorized by two types of glycoproteins on their surface, abbreviated HA and NA (or H and N when given in various combinations to identify a unique type of AI virus). The AI viruses are also classified as high or low pathogenicity (HPAI and LPAI). The term HPAI (high pathogenicity avian influenza) has a specific meaning relating to the ability of the virus to cause disease in experimentally inoculated chickens, and does not necessarily reflect the capacity of these viruses to produce disease in other species (Food and Agriculture Organization of the United Nations (FAO) 2013). However, it is these more virulent (highly harmful or infective) HPAI viruses that cause outbreaks of sickness and death in humans and other species of mammals and birds (FAO 2013; Krauss *et al.* 2010, p. 3373). Some LPAI types can mutate into HPAI forms (FAO 2013).

Anseriformes (swans, geese, and ducks) and Charadriiformes (gulls and shorebirds) are the natural hosts of LPAI (FAO 2013; Maxted *et al.* 2012, p. 322; Krauss *et al.* 2010, p. 3373; Olsen *et al.* 2006, p. 384). All 16 HA and 9 NA subtypes discovered to date have been detected in various combinations in wild aquatic birds, mainly LP forms. In general, LPAI viruses do not have significant health effects on wild birds, typically causing only a short-lived subclinical intestinal infection (FAO 2013; Krauss *et al.* 2010, p. 3373; Olsen *et al.* 2006, p. 384). However, HPAI can also occur in wild birds. One form of HPAI (H5N1) has caused mortality in more than 60 wild bird species, with population-level impacts in a few of those species. Although numerous wild birds have become infected with H5N1, debate remains whether wild birds play a role in the geographic spread of the disease (Olsen *et al.* 2006, pp. 387–388).

Since 1985, AI surveillance has been conducted annually from mid-May to early June in shorebirds and gulls in Delaware Bay. Influenza viruses (LP forms) are consistently isolated from shorebirds (i.e., the shorebirds were found to be carrying AI viruses) in Delaware Bay at an overall rate (5.2 percent) that is about 17 times higher than the combined rate of isolation at all other surveillance sites worldwide (0.3 percent) (Krauss *et al.* 2010, p. 3373). The isolation rate was even higher, 6.3 percent, from 2003 to 2008. Across global studies to date, AI viruses were rarely isolated from shorebirds except at two locations, Delaware Bay and a site in Australia (Krauss *et al.* 2010, p. 3375). The convergence of host factors and environmental factors at Delaware Bay results in a unique ecological "hot spot" for AI viruses in shorebirds (Krauss *et al.* 2010, p. 3373). Among the Delaware Bay shorebird species, ruddy turnstones (*Arenaria interpres*) have the highest infection rates by far (Maxted *et al.* 2012, p. 323). Although overall AI rates in Delaware Bay shorebirds are very high, red knots are rarely infected (L. Niles and D. Stallknecht pers. comm. January 25, 2013; Maxted *et al.* 2012, p. 322). Declining antibody prevalence in red knots over the stopover period suggests that their exposure to AI viruses generally occurs prior to arrival at Delaware Bay, with limited infection taking place at this site (Maxted *et al.* 2012, p. 322).

In wild red knots in Delaware Bay, AI infection rates are low, and only LP forms have been detected (Maxted *et al.* 2012, pp. 322–323). There is no evidence that the LPAI documented in wild red knots causes any harm to the health of these birds (L. Niles and D.

Stallknecht pers. comm. January 25, 2013). However, susceptibility of *Calidris canutus* to HP forms of influenza has been shown in captivity. Five of 26 *C.c. islandica* experimentally infected with an HPAI (H5N1) developed neurological disease or died during an experiment from 2007 to 2009 (Reperant *et al.* 2011, pp. 1, 4, 8). The appearance of clinical signs in these birds was sudden and the affected birds did not behave significantly differently on the preceding days than birds that remained sub-clinically infected (Reperant *et al.* 2011, p. 4). See Cumulative Effects, below, for discussion of an unlikely but potentially high-impact interaction among AI, environmental contaminants, and climate change.

Newcastle disease is a contagious bird disease (an avian paramyxovirus), and one of the most important poultry diseases worldwide. While people in direct contact with infected birds can get swelling and reddening of tissues around the eyes (conjunctivitis), no human cases of Newcastle disease have occurred from eating poultry products (Iowa State University 2008, entire). Although Newcastle disease is the most economically important, other types of avian paramyxovirus have been isolated from domestic poultry, where they occasionally cause respiratory and reproductive disease (Coffee *et al.* 2010, p. 481). No information is available regarding health effects of avian paramyxovirus in shorebirds.

From 2000 to 2005, Coffee *et al.* (2010, p. 481) tested 9,128 shorebirds and gulls of 33 species captured in 10 U.S. States and 3 countries in the Caribbean and South America for various types of avian paramyxovirus, including Newcastle disease virus. Avian paramyxoviruses were isolated from 60 (0.7 percent) samples, with 58 of the isolates coming from shorebirds (only 2 from gulls). All of the 58 positive shorebirds were sampled at Delaware Bay, and 45 of these isolates came from ruddy turnstones. The higher prevalence of avian paramyxovirus in ruddy turnstones mirrors the results observed for avian influenza viruses in shorebirds and may suggest similar modes of transmission (Coffee *et al.* 2010, p. 481). Of the birds sampled, 1,723 were red knots from Delaware Bay and 921 were red knots from other locations (Coffee *et al.* 2010, p. 483). Of these 2,644 red knots, only 7 tested positive (0.4 percent), and all 7 were captured in Delaware Bay (Coffee *et al.* 2010, p. 484). Like avian influenza virus, avian paramyxovirus infections in red knots may be site dependent, and at Delaware Bay these viruses may be locally amplified (Coffee *et al.* 2010, p. 486).

Since 2002, migratory birds in Brazil have been tested for various viruses including West Nile and Newcastle. As of 2007, AI type H2 had been found in one red knot, equine encephalitis virus in another, and Mayaro virus in seven knots (Niles *et al*. 2008, p. 101). Evidence does not indicate that West Nile virus will affect red knot health, and shorebirds are generally not regarded as important avian hosts in West Nile virus epidemiology (D. Stallknecht pers. comm. January 25, 2013). In 2005 and 2006, 156 red knots were sampled at 2 sites in Argentina (Río Grande and San Antonio Oeste) and tested for Newcastle disease virus, AI virus, and antibodies to the St. Louis encephalitis virus; all test results were negative (D'Amico *et al.* 2007, p. 794). One red knot was among 165 shorebirds of 11 species from southern Patagonia, Argentina, that were tested for all AI subtypes in 2004 and 2005; no AI was detected (Escudero *et al.* 2008, pp. 494–495).

For the most prevalent viruses found in shorebirds within the red knot's geographic range, infection rates in red knots are low, and health effects are minimal. We conclude that viral infections documented to date do not cause significant mortality and are not currently a threat to the red knot. However, see Cumulative Effects, below, regarding an unlikely but potentially high-impact, synergistic effect among avian influenza, environmental contaminants, and climate change in Delaware Bay.

Factor C—Predation

Predation—Nonbreeding Areas

In wintering and migration areas, the most common predators of red knots are peregrine falcons (*Falco peregrinus*), harriers (*Circus* spp.), accipiters (Family Accipitridae), merlins (*F. columbarius*), shorteared owls (*Asio flammeus*), and greater black-backed gulls (*Larus marinus*) (Niles *et al.* 2008, p. 28). In addition to greater black-backed gulls, other large gulls (e.g., herring gulls (*Larus argentatus*)) are anecdotally known to prey on shorebirds (Breese 2010, p. 3). Predation by a great horned owl (*Bubo virginianus*) has been documented in Florida (A. Schwarzer pers. comm. June 17, 2013). Nearly all documented predation of wintering red knots in Florida has been by avian, not terrestrial, predators (A. Schwarzer pers. comm. June 17, 2013). However in migration areas like Delaware Bay, terrestrial predators such as red foxes (*Vulpes vulpes*) and feral cats (*Felis catus*) may be a threat to red knots by causing disturbance, though direct mortality from these ground predators may be low (Niles *et al*. 2008, p. 101).

Ellis *et al.* (2002, pp. 316–317) summarized the documented prey species taken by peregrine falcons in Patagonia and Tierra del Fuego, based on early 1980s field surveys. Shorebirds represented only 8 of 55 reported prey species (about 15 percent), but accounted for 44 of 138 individual birds preyed on (about 32 percent) (Ellis *et al.* 2002, pp. 316–317), suggesting that shorebirds may be a favored prey type. Red knots were not reported among the prey species, but these authors considered their list incomplete and believed many more prey species would be identified from further sampling (Ellis *et al.* 2002, pp. 317–318).

In French Guiana, the primary natural threat is probably predation by peregrine falcons, which are fairly common from October to March or April. However, most red knots pass through French Guiana in August and September, and again in April and May, so the period of overlap between red knots and peregrine falcons is relatively short (CSRPN 2013). In Suriname, peregrine falcon numbers have increased substantially during the last few decades, and the birds are now common along the entire coast. Predation of shorebirds by peregrine falcons probably has negligible direct effects on shorebird numbers, in part because the falcons mainly prey upon terns and egrets. However, falcons hunting above Suriname's tidal mudflats regularly force semipalmated sandpipers to seek cover among the mangroves, reducing the length of time that birds can forage (Ottema and Spaans 2008, p. 345); we have no information if similar disturbances are occurring to red knots in Suriname.

Peregrine falcons have been seen frequently along beaches in Texas, where dunes would provide good cover for peregrines preying on red knots foraging along the narrow beachfront (Niles *et al*. 2009, p. 2). Peregrines are known to hunt shorebirds in the red knot's Virginia and Delaware Bay stopover areas (Niles 2010a; Niles *et al.* 2008, p. 106), and peregrine predation on red knots has been observed in Florida (A. Schwarzer pers. comm. June 17, 2013).

Raptor predation has been shown to be an important mortality factor for shorebirds at several sites (Piersma *et al.* 1993, p. 349). However, Niles *et al.* (2008, p. 28) concluded that increased raptor populations have not been shown to affect the size of shorebird populations. Based on studies of other *Calidris canutus* subspecies in the Dutch Wadden Sea, Piersma *et al.* (1993, p. 349) concluded that the chance for an individual to be attacked and captured is small, as long as the shorebirds remain in the open and in large flocks so that approaching raptors are likely to be detected. Although direct mortality from predation is generally considered relatively low in nonbreeding areas, predators also impact red knots by affecting habitat use and migration strategies (Niles *et al.* 2008, p. 101; Pomeroy *et al.* 2006, p. 1041; Stillman *et al.* 2005, p. 215) and by causing disturbance, thereby potentially affecting red knots' rates of feeding and weight gain. Each of these factors is discussed below. In addition, predation pressure may induce sublethal physiological stress that can impact shorebird fitness. In Australia, Clark and Clark (2002, p. 49) observed two individual shorebirds (a great knot (*Calidris tenuirostris*) and a *C. canutus* of unspecified subspecies) exhibit leg cramps and become unable to stand. As these two birds had been within a short distance of a predation event on a third shorebird, Clark and Clark (2002, p. 49) conclude the cramping resulted from naturally occurring stress.

Red knots' selection of high-tide roosting areas on the coast appears to be strongly influenced by raptor predation, something well demonstrated in other shorebirds (Niles *et al.* 2008, p. 28). Red knots require roosting habitats away from vegetation and structures that could harbor predators (Niles *et al.* 2008, p. 63). Red knots' usage of foraging habitat can also be affected by the presence of predators, possibly affecting the birds' ability to prepare for their final flights to the arctic breeding grounds (Watts 2009b) (e.g., if the knots are pushed out of those areas with the highest prey density or quality). In 2010, horseshoe crab egg densities were very high in Mispillion Harbor, Delaware, but red knot use was low because peregrine falcons were regularly hunting shorebirds in that area (Niles 2010a). Growing numbers of peregrine falcons on the Delaware Bay and New Jersey's Atlantic coasts are decreasing the suitability of a number of important shorebird areas (Niles 2010a). Analyzing survey data from the Virginia stopover area, Watts (2009b) found the density of red knots far (greater than 3.7 mi (6 km)) from peregrine nests was nearly eight times higher than close (0 to 1.9 mi (0 to 3 km)) to peregrine nests. In addition, red knot density in Virginia was significantly higher close to peregrine nests during those years when peregrine territories were not active compared to years when they were (Watts 2009b). Similar results were found for other *Calidris canutus* subspecies in the Dutch Wadden Sea, where the spatial distribution of *C. canutus* was best explained by both food availability and avoidance of predators (Piersma *et al.* 1993, p. 331).

In addition to affecting habitat use, predation has been shown to affect migration strategies in Arctic-breeding shorebirds (Lank *et al.* 2003, p. 303). Although mortality rates from predation on migrating birds are often low, Ydenberg *et al.* (2007, pp. 527–528) concluded that the danger generated by raptors and their migrations has had profound and pervasive effects on the evolution of avian migration. Studying two other *Calidris* species, Hope *et al.* (2011, p. 522) found that both adults and juveniles shortened their stopover durations during the period of increased peregrine falcon abundance. Recovering raptor populations in North America appear to have led to changes in the migratory strategies of western sandpipers (*C. mauri*), including behavioral changes, lower numbers of shorebirds, reduced stopover length, and lower body mass at the more predation-prone sites (Niles *et al*. 2008, p. 101; Butler *et al*. 2003, p. 132; Ydenberg *et al.* 2004, p. 1263).

Red knots can also be affected by peregrines through repeated disturbance. Red knots in Virginia are frequently disturbed by peregrine falcons (Niles *et al.* 2008, p. 106). Peregrines flying near foraging shorebirds at Delaware Bay are known to cause severe disturbance, prompting the shorebirds to fly in evasive maneuvers and not return for prolonged time periods. It is not believed that disturbance by peregrines in Delaware Bay changed significantly over the time period that red knots declined (Breese 2010, pp. 3–4).

The vulnerability of red knots, and their reactivity to perceived predation danger, may be related to their field of vision. Studying other subspecies, Martin and Piersma (2009, p. 437) found that *Calidris canutus* did not show comprehensive panoramic vision as found in some other tactile-feeding shorebirds, but have a binocular field surrounding the bill and a substantial blind area behind the head. This visual system may be a tradeoff for switching to more visually guided foraging (i.e., insects) on the breeding grounds. However, this forward-focused visual field leaves *C. canutus* vulnerable to aerial predation, especially when using tactile foraging in nonbreeding locations where predation by falcons is an important selection factor (Martin and Piersma 2009, p. 437).

In the United States, most peregrine falcons in coastal areas rely on artificial nest sites (Niles *et al.* 2008, p. 101). In some areas, land managers have begun to remove peregrine nesting platforms in strategic locations where they were having the greatest impact on shorebirds (DDNREC 2013; Niles 2010a; Watts 2009b; Kalasz 2008, p. 39).

Peregrine falcon populations in the United States have increased substantially since the mid-1970s, when the bird was extirpated in the east and only 324 known nesting pairs remained in total (USFWS 2012b). Today there are from 2,000 to 3,000 breeding pairs of peregrine falcons in North America (USFWS 2012b). Other raptor populations also increased over this period due to stricter pesticide regulations and conservation efforts (Butler *et al.* 2003, p. 130). Such measures reduced the prevalence of DDT (dichloro-diphenyl-trichloroethane) in the environment, which had caused egg shell thinning and, therefore, poor nest productivity in peregrine falcons (USFWS 2012b). We expect that peregrine and other raptor populations will continue to grow over coming decades, but at a slower rate. We also expect that land managers will continue balancing the conservation needs of both raptors and shorebirds, so that the predation pressures in key red knot wintering and stopover areas are likely to remain the same or decrease slightly.

We conclude that, outside of the breeding grounds (which are discussed below), predation is not directly impacting red knot populations despite some mortality. At key stopover sites, however, localized predation pressures exacerbate other threats to red knot populations, such as habitat loss (Factor A), food shortages (Factor E), and asynchronies between the birds' stopover period and the occurrence of favorable food and weather conditions (Factor E). Predation pressures worsen these threats by pushing red knots out of otherwise suitable foraging and roosting habitats, causing disturbance, and possibly causing changes to stopover duration or other aspects of the migration strategy (see Cumulative Effects below).

Predation—Breeding Areas

Although little information is available from the breeding grounds, the long-tailed jaeger (*Stercorarius longicaudus*) is prominently mentioned as a predator of red knot chicks in most accounts. Other avian predators include parasitic jaeger (*S. parasiticus*), pomarine jaeger (*S. pomarinus*), herring gull, glaucous gull (*Larus hyperboreus*), gyrfalcon (*Falco rusticolus*), peregrine falcon, and snowy owl (*Bubo scandiacus*). Mammalian predators include arctic fox (*Alopex lagopus*) and sometimes arctic wolves (*Canis lupus arctos*) (Niles *et al.* 2008, p. 28; COSEWIC 2007, p. 19). On Southampton Island, a red knot breeding site, Smith and Wilson (2010, p. 615) found the most significant shorebird predators were parasitic jaegers and arctic foxes. These authors considered herring gulls, though abundant, a minor predator, and assumed other predators (e.g., peregrine falcons, glaucous gulls, common ravens (*Corvus corax*)) were unimportant due to their low abundance (Smith and Wilson 2010, p. 615).

Predation pressure on Arctic-nesting shorebird clutches varies widely regionally, interannually, and even within each nesting season, with nest losses to predators ranging from close to 0 percent to near 100 percent (Meltofte *et al.* 2007, p. 20), depending on ecological factors. Studying five shorebird species on Southampton Island, Smith and Wilson (2010, pp. 615, 621) found that predators were the proximate cause of 98 percent of observed nest failures. (Although this study did not include red knots, similar predation patterns likely apply to knots since they breed in this area.) These authors found that an index of predator activity showed no consistent patterns throughout the season that could account for observed temporal trends in nest survival. The lack of a relationship between the predator index and shorebird nest survival suggests that predators at these sites rely primarily on other food sources, such as waterfowl eggs and chicks. Thus, annual and seasonal variation in shorebird nest predation could reflect timing and availability of other food source (Smith and Wilson 2010, pp. 621–622).

Abundance of arctic rodents, such as lemmings, is often cyclical, although less so in North America than in Eurasia. In the Arctic, 3- to 4-year lemming cycles give rise to similar cycles in the predation of shorebird nests. When lemmings are abundant, predators concentrate on the lemmings, and shorebirds breed successfully. When lemmings are in short supply, predators switch to shorebird eggs and chicks (Niles *et al.* 2008, p. 101; COSEWIC 2007, p. 19; Meltofte *et al.* 2007, p. 21; USFWS 2003, p. 23; Blomqvist *et al*. 2002, p. 152; Summers and Underhill 1987, p. 169). Studying five shorebird species (not including red knot) on Southampton Island, Smith *et al.* (2007, p. 15) found that the factor with the greatest interannual influence on nest success was predation pressure, apparently related to the abundance of predators and lemmings. Blomqvist *et al.* (2002, p. 146) correlated predation pressure on *Calidris canutus canutus* on Siberian breeding grounds with numbers of juveniles in nonbreeding areas, following a 3-year cycle. These authors concluded that the reproductive output of *C.c. canutus* was limited by predation and that chick production was high when predation pressure was reduced by arctic foxes preying primarily on lemmings (Fraser *et al.* 2013, p. 13; Blomqvist *et al.* 2002, p. 146).

In addition to affecting reproductive output, these cyclic predation pressures have been shown to influence shorebird nesting chronology and distribution. Studying 12 shorebird species, including red knot, over 11 years at 4 sites in the eastern Canadian Arctic, Smith *et al.*

(2010a, pp. 292; 300) found that both snow conditions and predator abundance have significant effects on the chronology of breeding. Higher predator abundance resulted in earlier nesting than would be predicted by snow cover alone (Smith *et al.* 2010a, p. 292). Based on the adaptations of various species to deal with predators, Larson (1960, pp. 300–303) concluded that the distribution and abundance of *Calidris canutus* and other Arctic-breeding shorebirds were strongly influenced by arctic fox and rodent cycles, such that shorebirds were in low numbers or absent in areas without lemmings because foxes preyed predominately on shorebirds in those areas (as cited in Fraser *et al.* 2013, p. 14).

Years with few lemmings and many predators can be extremely unproductive for red knots, although predator cycles are usually not uniform across all breeding areas so that in most years there is generally some production of young (Niles *et al.* 2008, p. 63). Unsuccessful breeding seasons contributed to at least some of the observed reductions in the red knot population in the 2000s. Rodent-predator cycles have always affected the productivity of Arctic-breeding shorebirds and have generally caused only minor year-to-year changes in otherwise stable populations (Niles *et al.* 2008, pp. 64, 101). However, these cycles have diminished in several parts of the Arctic, potentially representing a long-term threat to the red knots successful reproduction.

In northern Europe, lemming cycles diminished after the early 1990s but returned in the early 2000s (Fraser *et al.* 2013, p. 16; Brommer *et al*. 2010, p. 577; Kausrud *et al.* 2008, p. 93). Changes in temperature and humidity seemed to markedly affect rodent dynamics by altering conditions in the spaces below the snow where lemming prefer to live. These observations led Kausrud *et al.* (2008, p. 93) to conclude that the pattern of less regular rodent peaks, and corresponding ecosystem changes mediated by predators, seem likely to prevail over a growing geographic area under projected climate change. Likewise, Ims *et al.* (2008, p. 79) found several lines of evidence pointing to anthropogenic climate change as the general underlying cause for the fading of population cycles at northern latitudes. However, Brommer *et al.* (2010, p. 577) found that lemming cycles in Finland returned after about 5 years despite ongoing and rapid climate change, suggesting that climate change may not explain why the cycles were interrupted. Although factors other than climate change may also be important, the IPCC concluded that the documented collapse or dampening of population cycles of voles and lemmings over the last 20 to 30 years in parts of Fennoscandia and Greenland can be attributed to climate change with "high confidence" (Chapter 28 <u>in</u> IPCC 2014, p. 14). Herbivores with high-amplitude population cycles, such as voles and lemmings, form the heart of terrestrial food web dynamics. Thus, collapses of these cycles are also expected to imply collapses of important ecosystem functions (Ims *et al.* 2008, p. 79).

At two sites in northeast Greenland, lemming populations collapsed around 2000, both in terms of actual densities and periodicity (Schmidt *et al.* 2012, p. 4419). The observed change in Greenland lemming dynamics dramatically affected the predator guild, with the most pronounced response in two lemming-specialist predator species (Schmidt *et al.* 2012, p. 4421). Observed differences in predator responses between the two Greenland sites could arise from site-specific differences in lemming dynamics, interactions among predators, or subsidies from other resources (Schmidt *et al.* 2012, p. 4417) (e.g., shifting to other prey species, which could have implications for shorebirds). Ultimately, changing predator populations may cause cascading impacts on the entire tundra food web, with unknown consequences (Schmidt *et al.* 2012, p. 4421). Unlike the 1990s lemming cycle disruption in Europe, Schmidt *et al.* (2012, entire) did not report any signs of recovery of the Greenland lemming cycles, based on data through 2010.

At one localized site, Canada's Bylot Island (just north of the rufa red knot breeding range), Gauthier *et al.* (2013, p. 1) found little evidence for changes in the phenology, abundance, or productivity of lemmings, foxes, or avian predators over a 24-year period (1989 to 2013), despite a warming trend and a 4- to 7-day advancement of the date of spring snowmelt. The large inter-annual variability in lemming density due to cyclic population fluctuations makes the detection of trends difficult. Looking at the 3-year running mean of density to smooth out the variability suggested a possible decreasing trend in summer lemming density. Lemming populations were especially low from 2002 to 2009, but the most recent peak (2011) was relatively high. Although not statistically significant, nesting snowy owl abundance tended to decrease over time, which is consistent with the apparent decreasing trend in lemming density (Gauthier *et al.* 2013, p. 6). For cyclic populations such as small rodents and the predators that depend upon them, a large number of years may be needed to detect trends. Extreme changes such as a total collapse of cycles are probably easier to detect from relatively short time series, but were not observed in this study. In the case of predators such as arctic fox, future change in their populations may be more related to changes in the absolute abundance of their prey rather than on the seasonal timing of their availability. This study found that both the foxes' diet and annual productivity were strongly affected by cyclic fluctuations in lemming abundance, consistent with other studies (Gauthier *et al.* 2013, pp. 8–9).

Rodent population cycles typically show a geographic gradient in the return period (interval of time between peaks), with periods growing shorter moving from north to south. The period shortening is due to the increased strength of density dependence, which is attributed to an increased abundance of generalist predators towards the south, with the population cycle eventually collapsing into stable dynamics in the far south. Period shortening and eventual collapses of cycles can also occur along other geographic gradients, for example certain coast-inland or altitudinal gradients. However, for all types of such spatial gradients, the cycles appear to fade with shorter winters (Ims *et al.* 2008, p. 81). Thus, both the direct (e.g., shorter winters) and indirect (e.g., colonization by new predators from the south) effects of warming may impact rodent cycles within the red knot's breeding range. Ims *et al.* (2008, p. 85) interpreted available modeling and observation data to conclude that the geographical borders between cyclic and noncyclic populations are currently on the move following changing climatic gradients, and that the regions with cycles are shrinking.

Disruption of rodent-predator cycles may constitute a large-scale impact on predation pressure on arctic shorebird nests (Meltofte *et al.* 2007, p. 22). In the Siberian Arctic, lemmings are keystone species, and any climate effects on their abundance or population dynamics may indirectly affect shorebird populations through predation. The role of lemmings in the eastern Canadian Arctic is unclear, but large annual fluctuations in lemming or other rodent populations suggest that similar dynamics operate there (Meltofte *et al.* 2007, p. 34). Fraser *et al.* (2013, p. 13) investigated the relationship between the rodent cycle in Arctic Canada and numbers of red knots migrating through the United States. Shooting records from Cape Cod in the 1800s and red knot counts on Delaware Bay from 1986 to 1998 cycled with 4-year periods. Annual peaks in numbers of red knots stopping in the Delaware Bay from 1986 to 1998 occurred 2 years after arctic rodent peaks, with a correlation more often than expected at random. These results suggest that red knot reproductive output was linked to the rodent cycle before the red knot population decline (i.e., 1998 and earlier), but not after 1998. These findings are consistent with a hypothesis that an interruption of the rodent cycle in red knot breeding habitat could have been a driver in the red knot decline observed in the 2000s. However, additional studies would be needed to support this hypothesis (Fraser *et al.* 2013, p. 13).

McKinnon *et al.* (2010, p. 326) used artificial nests to measure predation risk along a 2,083-mi (3,350-km) south-north gradient in the Canadian Arctic and found that nest predation risk declined more than twofold along the latitudinal gradient. The study area included the entire latitudinal range of known and modeled red knot breeding habitat, extending both farther south (into the sub-Arctic) and farther north (to encompass the breeding range of *Calidris canutus islandica*). Nest predation risk was negatively correlated with latitude. For an increase in 1° of latitude, the relative risk of predation declined by 3.6 percent, equating to a 65 percent decrease in predation risk over the 29° latitudinal transect. These results provide evidence that birds migrating farther north may acquire reproductive benefits in the form of lower nest predation risk (McKinnon *et al.* 2010, p. 326). Predation pressure on red knots could increase if, due to climate change, a new suite of predators expands their ranges northward from the sub-Arctic into the knot's breeding range.

We conclude that cyclic predation in the Arctic results in years with extremely low reproductive output but does not threaten the red knot (Meltofte *et al.* 2007, p. 20). The cyclical nature of this predation on shorebirds is a situation that has probably occurred over many centuries, and under historical conditions likely had no lasting impact on red knot populations (Niles *et al.* 2008, pp. 64, 101). Where and when rodent-predator cycles are operating, we expect red knot reproductive success will also be cyclic. However, these cycles are being interrupted by climate change (Chapter 28 in IPCC 2014, p. 14). The geographic extent and duration of future interruptions to the cycles cannot be forecast, but are likely to intensify as the arctic climate continues to change (Ims *et al.* 2008, p. 85). Disruptions in the rodent-predator cycle pose a substantial threat to red knot populations, as they may result in prolonged periods of very low reproductive output (Meltofte *et al.* 2007, p. 22). Such disruptions have already impacted red knot populations and may increase due to climate change (Chapter 28 in IPCC 2014, p. 14; Fraser *et al.* 2013, pp. 13, 16; Brommer *et al*. 2010, p. 577; Ims *et al.* 2008, p. 79; Kausrud *et al.* 2008, p. 98). Superimposed on these potential cycle disruptions are warming temperatures and changing vegetative conditions in the Arctic, which are likely to bring about additional changes in the predation pressures faced by red knots on the breeding grounds, such as colonization by new predators from the south (see also Factor A—Breeding Habitat). We cannot forecast how such ecosystem changes are likely to unfold.

Factor C—Conservation Efforts

We are unaware of any conservation efforts to reduce disease in red knots. We are also unaware of any conservation efforts to reduce predation of the red knot in its breeding range. As discussed above, land managers in some areas of the United States have begun to remove peregrine nesting platforms in key locations where they are having the greatest impact on shorebirds.

Factor C—Summary

Red knots may be adapted to parasite-poor habitats and may, therefore, be susceptible to parasites when migrating or wintering in high-parasite regions (Piersma 1997, p. 623). However, we have no evidence that parasites have affected red knot populations beyond causing normal, background levels of mortality (D'Amico *et al*. 2008, pp. 193, 197; Harrington 2001, p. 21), and we have no indications that parasite infection rates or red knot fitness impacts are likely to increase. Therefore, we conclude that parasites are not a threat to the red knot. For the most prevalent viruses found in shorebirds within the red knot's geographic range (e.g., avian influenza, avian paramyxovirus), infection rates in red knots are low, and health effects are minimal or have not been documented (D. Stallknecht pers. comm. January 25, 2013; Maxted *et al.* 2012, pp. 322–323; Coffee *et al.* 2010, p. 484; Escudero *et al.* 2008, pp. 494–495; Niles *et al*. 2008, p. 101; D'Amico *et al.* 2007, p. 794). Therefore, we conclude that viral infections do not cause significant mortality and are not a threat to the red knot. However, see Cumulative Effects (below) regarding an unlikely but potentially high-impact, synergistic effect among avian influenza, environmental contaminants, and climate change in Delaware Bay.

Outside of the breeding grounds, predation is not directly effecting red knot populations despite some mortality (Niles *et al.* 2008, p. 28). At key stopover sites, however, localized predation pressures exacerbate other threats to red knot populations by pushing red knots out of otherwise suitable foraging and roosting habitats, causing disturbance, and possibly causing changes to stopover duration or other aspects of the migration strategy (Niles 2010a; Watts 2009b; Niles *et al.* 2008, pp. 101, 116; Lank *et al.* 2003, p. 303). In addition, predation pressure may induce sublethal physiological stress that can impact shorebird fitness (Clark and Clark 2002, p. 49). We expect the direct and indirect effects of predators to continue at the same level or decrease slightly over the next few decades.

Within the breeding range, normal 3- to 4-year cycles of high predation, mediated by rodent cycles, result in years with extremely low reproductive output but do not threaten the survival of the red knot at the subspecies level (Niles *et al.* 2008, pp. 64, 101; Meltofte *et al.* 2007, p. 20). It is believed shorebirds, such as red knots, have adapted to these cycles, therefore these natural cycles are not considered a threat to the red knot. What is a threat, however, is that these natural rodent/predator cycles are being disrupted by climate change, which may increase predation rates on shorebirds over the long term and have subspecies-level effects (Chapter 28 in IPCC 2014, p. 14; Fraser *et al.* 2013, pp. 13, 16; Brommer *et al*. 2010, p. 577; Ims *et al.* 2008, p. 79; Kausrud *et al.* 2008, p. 98). Disruptions in the rodent-predator cycle pose a substantial threat to the red knot, as they may result in prolonged periods of very low reproductive output (Meltofte *et al.* 2007, p. 22). Such disruptions have already occurred and may increase due to climate change (Chapter 28 in IPCC 2014, p. 14; Fraser *et al.* 2013, pp. 13, 16; Brommer *et al*. 2010, p. 577; Ims *et al.* 2008, p. 79; Kausrud *et al.* 2008, p. 98). The substantial impacts of elevated egg and chick predation on shorebird reproduction are well known (Smith and Wilson 2010, pp. 615, 621; Meltofte *et al.* 2007, p. 20), although the red knot's capacity to adapt to long-term changes in predation pressure is unknown (Meltofte *et al.* 2007, p. 34). The threat of persistent increases in predation in the Arctic may already be having subspecies-level effects

(Fraser *et al.* 2013, p. 13) and is anticipated to increase into the future. Further, warming temperatures and changing vegetative conditions in the Arctic are likely to bring additional changes in the predation pressures faced by red knots, such as colonization by new predators from the south, though we cannot forecast how such ecosystem changes are likely to unfold.

From our review of best available data, we conclude that disease is not likely a threat to red knot populations. Predation pressures likely exacerbate other threats in some nonbreeding areas, but likely contribute little direct mortality. Natural cycles of high predation rates on the breeding grounds are not a threat to red knot population, but disruption of these cycles from climate change, which may lead to prolonged periods of low productivity, is an emerging threat to the red knot.

FACTOR D. THE INADEQUACY OF EXISTING REGULATORY MECHANISMS

Under this factor, we examine the effects of existing regulatory mechanisms in relation to the threats to the red knot discussed under Factors A, B, C, and E. Section 4(b)(1)(A) of the Act requires the Service to take into account "those efforts, if any, being made by any State or foreign nation, or any political subdivision of a State or foreign nation, to protect such species." In relation to Factor D under the Act, we interpret this language to require the Service to consider relevant Federal, state, and tribal laws, regulations, and other such mechanisms that may reduce any of the threats we describe in our threat analyses under the other four factors. We give strongest weight to statutes and their implementing regulations and to management direction that stems from those laws and regulations. An example would be State governmental actions enforced under a State statute, or Federal actions under Federal statute. The following section includes a discussion of international, Federal, State, and local laws, regulations, and treaties that apply to the red knot. It includes legislation for Federal land management agencies and State and Federal regulatory authorities affecting identified threats to the red knot.

Factor D—Canadian Laws and Regulations

In 2012, the rufa red knot was determined to be endangered under the Canadian Species at Risk Act (SARA) (Species at Risk Public Registry 2012). The SARA makes it an offense to kill, harm, harass, capture, or take an individual of a listed species that is endangered or threatened; possess, collect, buy, sell, or trade an individual of a listed species that is extirpated, endangered, or threatened, or its part or derivative, or to damage or destroy the residence of one or more individuals of a listed endangered or threatened species or of a listed extirpated species if a recovery strategy has recommended its reintroduction. For many of the species listed under SARA, the prohibitions on harm to individuals and destruction of residences are limited to Federal lands, but this limitation does not apply to migratory birds protected under the Migratory Birds Convention Act (MBCA) (Statutes of Canada (S.C). c. 29, § 34), which includes the red knot (Environment Canada 2013). Hence, SARA protects red knots, where present, from harm and destruction of their residences, not only on Federal lands, but also on provincial and private lands. The MBCA (S.C. c. 22) is Canada's legislation, similar to the United States' Migratory Bird Treaty Act (MBTA), implementing the Migratory Bird Treaty among the United States, Canada, and Mexico (Environment Canada 2013). The MBCA and its implementing regulations prohibit the possession or sale of migratory birds or their carcasses, skins, nests, or eggs (C.R.C. c. 1035 §§ 6, 12). Birds in the Family Scolopacidae, including the red knot, are listed as a game species under international treaties with Canada and Mexico (U.S. Fish and Wildlife Service (USFWS) 2012c); thus these shorebirds are classified as game species under the MBCA (Environment Canada 2013). However, the only shorebirds that can be legally hunted in Canada are American woodcock (*Scolopax minor*) and snipe (*Gallinago delicata*); there has not been an open season for any other species of shorebirds in Canada since the passing of the original MBCA in 1917 (J. Bertrand pers. comm. May 16, 2013).

The following administrative divisions of Canada have enacted provincial or territorial laws for the protection of endangered species, but the red knot is not listed in these jurisdictions: Alberta, New Brunswick, Northwest Territories, Nunavut, Prince Edward Island, Quebec, and Saskatchewan. We consider British Columbia and Yukon to be outside the geographic range of the rufa red knot (see Subspecies Nonbreeding Ranges). The rufa red knot is listed as endangered by the remaining Provinces of Manitoba, Newfoundland and Labrador, Nova Scotia, and Ontario, which all provide some habitat protections and prohibit direct take of listed species.

In summary, SARA provides protections for the red knot and its habitat, both on and off of Federal lands. The red knot is afforded additional protections under the MBCA, and in the Provinces of Manitoba, Newfoundland and Labrador, Nova Scotia, and Ontario where it is listed as an endangered species at the provincial level.

Factor D—Caribbean and Latin American Laws and Regulations

Wildlife policy and legislation across Latin America and the Caribbean are heterogeneous. A 1996 review by the United Nations Food and Agriculture Organization (FAO) found that a few countries had not yet passed legislation on wildlife utilization, and that the wildlife rules and regulations in other countries were obsolete or incomplete. However, other countries (e.g., Brazil, Colombia, Panama, and Paraguay) had adopted protection-oriented policies, prohibiting almost all wildlife utilization, while still others (e.g., Argentina, Costa Rica, Mexico, Nicaragua, Peru, Suriname, and Venezuela) were trying to combine the protection of endangered species with the controlled utilization of numerically sufficient species. Wildlife policies have tended to change course suddenly in some countries. In many countries, legislation was updated after 1970; this new generation of laws (particularly in Argentina, Bolivia, Brazil, Colombia, Costa Rica, Ecuador, Peru, Suriname, and Venezuela) recognized wildlife as a public resource and went beyond mere game laws to include wildlife protection, promotion, and management (e.g., habitat protections, research, education) (FAO 1996).

A 1996 survey by the FAO found that nearly half (48 percent) of the experts consulted believed the legislation in force in their respective countries to be obsolete or unrealistic, 26 percent found it adequate, 22 percent found it satisfactory, and 4 percent indicated a lack of legislation in their countries. The Brazilian experts, for example, were in agreement with the legislation in force, whereas most of the Mexican experts felt that their hunting laws were obsolete (FAO 1996) (however, Mexico’s wildlife laws were subsequently updated around 2000, as discussed below). Wildlife legislation is national in scope throughout Latin America except for Argentina, which has Federal guidelines but, within which, each province enacts its own specific laws. Countries with modern legislation and where hunting is permitted have generally enacted regionalized timetables of open and closed seasons, bag limits, areas where hunting is permitted, and other regulations covering each specific game species (FAO 1996).

Where information is available, we discuss applicable laws in countries known to support red knots, generally moving from north to south. However, we largely lack information regarding the implementation (e.g., administration, compliance, and enforcement) and effects of these laws and, unless otherwise indicated, we are not aware of the extent to which these laws apply to the protection of red knots or their habitats (e.g., for many countries we do not have information indicating if the red knot is a protected species, a game species, or neither).

In the Bahamas, the Wild Birds Protection Act (Ch. 249) provides for the appointment of game wardens; prohibits killing or capturing certain wild birds during a closed season; establishes hunting licenses and harvest limits; establishes wild bird reserves; and restricts trade in wild birds, skins, feathers, and eggs. Red knots are included in Schedule 1, for which the closed season is the entire year.

In Cuba, Law 81 of the Environment provides for a National System of Protected Areas, and tasks the Ministries of Agriculture and Fishing Industry to regulate the use of wildlife, establish hunting and collection regulations, and protect threatened and endangered species.

In Jamaica, the Wildlife Protection Act (1945) regulates sport hunting, and has been enhanced by many regulations that attempt to address gaps, particularly in relation to protected animals. However, this act does not address habitat protection (B. Andres pers. comm. December 21, 2011). Red knots are among the protected bird species for which hunting is prohibited.

In the Dominican Republic, the Environment and Natural Resources General Law (No. 64-00) forbids the unauthorized destruction, degradation, disregard for or decrease of the natural ecosystems and of the species of wild flora and fauna and the collection of specimens of flora and fauna; establishes a list of species that are in danger of extinction, threatened, or protected, which shall be the object of rigorous control and of mechanisms of protection; tasks the State Secretariat of Environment and Natural Resources with managing protected areas and wildlife; provides for hunting regulations; and restricts the introduction of exotic species.

In the British Virgin Islands, the Wild Birds Protection Ordinance (Cap. 98, 1959 as amended 1980) provides for the establishment of bird sanctuaries, protects 24 species of birds, and designates game species with a closed season from February 1 to July 15. Procter and Fleming (1999, p. 51) concluded that this statute is in need of updating to address confusion regarding species' common names. The Protection of Endangered Animals, Plants and Articles (Removal and Possession) Ordinance 1981 seeks to prohibit the removal or possession, without a license from the Minister, of black coral or any article principally derived therefrom; provision is made for the addition of other species of plants, animals or articles requiring similar protection (Procter and Fleming 1999, pp. 51–52).

In recent years, the French government has acted to impose new protective measures in Guadeloupe. The National Hunting and Wildlife Agency has implemented a bag limit of 20 birds per day and is working on a regulation that would stop hunting for 5 days following a tropical storm warning (McClain 2013; A. Levesque pers. comm. January 8, 2013; Niles 2012c).

Significantly, the red knot was officially added to the list of protected species in Guadeloupe and Martinique on August 15, 2013. Maximum penalties for killing a red knot in Guadeloupe or Martinique include confiscation of the gun, a fine of up to 15,000 euros, up to 6 months in prison (A. Levesque pers. comm. September 5, 2013). In addition, hunter education about red knots is in progress (G. Humbert pers. comm. November 29, 2013; A. Levesque pers. comm. January 8, 2013; Niles 2012c). The red knot is therefore protected throughout the New World French territories (including in Saint-Pierre-et-Miquelon, off the south coast of Newfoundland, Canada, where knots have been protected since 1989)—French Guiana is discussed below (G. Humbert pers. comm. November 29, 2013).

In Barbados, the Wild Birds Protection Act (Chapter 398, 1985) prohibits killing of certain species, but not red knots. There has been voluntary agreement by hunters to stop the harvest of red knots (USFWS 2011e, p. 2), but we are unaware of any regulatory enforcement mechanism. The Barbados Coastal Zone Management Act (Chapter 394, 1998) restricts the removal of vegetation, sand, or stones from the beaches, and the fouling of a beach via waste disposal.

In Mexico, wildlife management prerogatives and regulatory powers reside in the Federal government with States playing only a minimal role (Valdez *et al.* 2006, p. 270). Mexico's 2001 revision of the General Law on Ecological Balance and Environmental Protection establishes Federal regulation of the sustainable use, protection, and preservation of wildlife and establishes natural areas. In 2000, Mexico enacted the General Wildlife Law, the most comprehensive Mexican wildlife legislation to date, which contains provisions on the sustainable use of wildlife; incentives for land owners; cooperation among Federal, State, and municipal governments and private individuals; wildlife diseases; ethical use of wildlife; restrictions on exotic species, wildlife research and rehabilitation centers; wildlife use by indigenous people; environmental education; species at risk and their critical habitat; reintroduction and translocation protocols; scientific collection permits; control of nuisance species; and law enforcement investigations and citations (Valdez *et al*. 2006, p. 274). Hunting is regulated by the Mexican government, and extensive dove hunting occurs in northern Mexico including the State of Tamaulipas (Valdez *et al.* 2006, pp. 275–276); however, we have no information on shorebird hunting. In a review of Mexico's wildlife conservation laws, Valdez *et al.* (2006, p. 270) concluded that the frequent shifting of the Federal agencies responsible for wildlife management and a concomitant lack of adequate funding and other obstacles have prevented the establishment of a robust wildlife program. These authors concluded that the present Federal wildlife management strategy is an initial positive effort because it promotes participatory wildlife conservation by key stakeholders (Valdez *et al.* 2006, p. 270).

The subspecies composition of *Calidris canutus* in several Central American countries is unknown, but we have data to suggest that at least some of these birds are rufa red knots (see Subspecies Nonbreeding Distributions); thus, we have assessed available information regarding applicable laws in this region. The Belize Wildlife Protection Act (Chapter 220) regulates hunting, and the hunting of most birds including red knots is prohibited. In El Salvador, the Law of Conservation of Wildlife (Decree No. 844) tasks the National Park Service and Wildlife Service with developing hunting and other regulations, developing and updating a list of threatened and endangered species, and conducting research. The World Future Council (2011, pp. 5–10) concluded that Costa Rica's 1998 Biodiversity Law was successfully meeting several environmental performance metrics by promoting the conservation and sustainable use of biodiversity and ensuring the fair and equitable sharing of benefits derived therefrom. This Costa Rican law establishes wild protected areas, provides for the conservation and sustainable use of ecosystems and species, provides for environmental impact assessment, and promotes education and research (World Future Council 2011, p. 5). The practice of sport hunting in Panama is governed by Law 24 of 1995 and Law 39 of 2005, which, among other regulations, determine the animal species for which hunting is allowed and the closure periods. Legal protections for shorebird habitats around Panama Bay, where rufa red knots are known to occur, are in flux (WHSRN 2014b; WHSRN 2014c)

As in Central America, best available data indicate that at least some of the *Calidris canutus* along South America's northwest coast are rufa red knots. In Colombia, Law 99 (1993) tasks the Ministry of the Environment with managing the collection, use, and trade of wildlife, and provides for the Regional Autonomous Corporations to establish hunting seasons. In Venezuela, the Law for the Protection of Wildlife (1970) establishes hunting regulations, as well as wildlife reserves, refuges, and sanctuaries. Trinidad and Tobago has three designated Environmentally Sensitive Areas and three designated Environmentally Sensitive Species, but the red knot is not included (Environmental Management Authority of Trinidad and Tobago 2011).

Current Guyanese legislation consists of the Wild Birds Protection Act of 1987 (B. Andres pers. comm. December 21, 2011), which does not include the red knot on the list of wild bird species protected seasonally or year round. In Suriname, the Nature Conservation Act (1954, last updated 1992) allows for the establishment of nature reserves. Shorebirds in Suriname have been protected since 2002 under multiple use management areas (except for South American snipe and whimbrel) (B. Andres pers. comm. December 21, 2011). French Guiana, an overseas department of France, had no hunting laws as of 2013, although commercial use of any wild bird was prohibited (CSRPN 2013). French Guiana established a list of protected birds in 1986 (CSRPN 2013)—the red knot was added to the list in October 2014, the first shorebird to be protected in French Guiana (C. Carichiopulo and N. de Pracontal pers. comm. October 10, 2014). In addition, the local Ministry of Environment has worked for several years with partners to produce a new ministerial order including a global hunting ban for every species of Charadriiformes in French Guiana (CSRPN 2013).

Brazil's Federal Constitution of 1988 includes protection of the country's fauna and flora, and establishes the legal standards for environmental protection. Article 225 of Brazil's Constitution confers jointly on the Federal government, States, and municipalities the authority and duty to protect the Brazilian fauna and flora. In 1998, the Environmental Crimes Law (law n. 9.605/98) was enacted to complement the Constitution and impose criminal liability on environmental crimes. The Environmental Crimes Law states that Brazilian wild fauna are considered public property that cannot be appropriated, and their use is subject to regulation by the Federal government. The Environmental Crimes Law includes criminal penalties in cases of actions or activities that may damage the environment and provides for the imposition of criminal liability on a person or legal entity that pollutes or degrades the environment. Crimes against wild fauna include killing, hounding, hunting, capturing, or using any fauna species without authorization or license, with penalties including detention of 6 months to 1 year, a fine, or both. The penalty is aggravated if the crime is committed against rare species or those considered endangered (even if only at the site of violation); in the period in which hunting is prohibited; during the night; by abusing a license; within a conservation unit; or by using methods or instruments capable of provoking mass destruction. There are exceptions including killing of animals to satisfy hunger or, via permit, to protect agriculture. Introductions of species into the country are prohibited without a license. Except for the state of Rio Grande do Sul, commercial, sport, and recreational hunting are prohibited in all Brazilian territory. The State of Rio Grande do Sul hunting law provides a list of animals that can be hunted, and prohibits trapping devices as well as commercialized hunting (Animal Legal and Historical Center 2011). The red knot is not listed as a species threatened with extinction in Brazil (Ministry of Environment 2013).

Uruguay has a variety of laws regarding wildlife, hunting, protected areas, biological diversity, environmental impact assessments, use of coastal and estuarine areas, and environmental contaminants (Vida Silvestre Uruguay 2013). The red knot is not listed as a game species in Uruguay (Uruguay Ministry of Livestock Agriculture and Fish 2013, p. 6). In Argentina, Law 22.421 regulates wildlife hunting, trade, and other uses. Rufa red knots were declared endangered in Argentina in 2010 (P. González pers. comm. November 29, 2013). With only Federal guidelines, each Argentinean province enacts its own specific laws (FAO 1996). Two of Argentina’s Patagonian provinces (Río Negro that includes San Antonio Oeste, and Santa Cruz that includes Río Gallegos) have declared the conservation of migratory shorebirds to be “in the Provincial interest” and made it illegal to modify wetland habitats important for shorebirds (WHSRN 2011). Chile has a variety of wildlife laws, including regulation of hunting and classification of rare, vulnerable, and endangered species (Chile Law 2013).

In summary, red knots are legally protected from direct take and hunting in several Caribbean and Latin American countries, but we lack information regarding the implementation or effectiveness of these measures. For many other countries, red knot hunting is unregulated, or we lack sufficient information to determine if red knot hunting is legal. We also lack information regarding the regulation of other activities, such as development, disturbance, oil spills, environmental contaminants, and wind energy development that threaten the red knot and its habitat.

Factor D—U.S. Laws and Regulations

United States—Wildlife Laws and Regulations

Prior to September 30, 2013, the Migratory Bird Treaty Act of 1918 (16 U.S.C. 703 *et seq*.) (MBTA) was the only Federal law in the United States providing specific protection for the red knot due to its status as a migratory bird. The MBTA prohibits the following actions, unless permitted by Federal regulation: to “pursue, hunt, take, capture, kill, attempt to take, capture or kill, possess, offer for sale, sell, offer to purchase, purchase, deliver for shipment, ship, cause to be shipped, deliver for transportation, transport, cause to be transported, carry, or cause to be carried by any means whatever, receive for shipment, transportation or carriage, or export, at any time, or in any manner, any migratory bird…or any part, nest, or egg of any such bird.” Through issuance of Migratory Bird Scientific Collecting permits, the Service ensures that best practices are implemented for the careful capture and handling of red knots during banding operations and other research activities (see Factor B—Scientific Study).

Birds in the Family Scolopacidae, including the red knot, are listed as a game species under international treaties with Canada and Mexico. The MBTA, which implements these treaties, grants the Service authority to establish hunting seasons for any listed game species. However, the Service has determined that hunting is appropriate only for those species for which there is a long tradition of hunting, and for which hunting is consistent with their population status and their long-term conservation. The Service would not consider legalizing the hunting of shorebird species, such as the red knot, whose populations were devastated by market hunting in the last decades of the 19th century (USFWS 2012c) (see Factor B—Hunting).

There are no provisions in the MBTA that prevent habitat destruction unless the activity causes direct mortality or the destruction of active nests, which would not apply since red knots do not breed in the United States. The MBTA does not address threats to the red knot from further population declines associated with habitat loss, insufficient food resources, climate change, or the other threats discussed in the proposed rule under Factors A, B, C, and E.

Among coastal States from Maine to Texas, all except Alabama have enacted some kind of endangered species legislation; however, the red knot is listed only in New Jersey (endangered), Delaware (endangered), and Georgia (rare, a category of protected species). The New Jersey Endangered and Nongame Species Conservation Act of 1973 (N.J.S.A. 23:2A *et seq*.) prohibits taking, possessing, transporting, exporting, processing, selling, or shipping listed species. "Take" is defined as harassing, hunting, capturing, or killing, or attempting to do so. As a State-listed species, the red knot is also afforded habitat protection under the New Jersey Coastal Zone Rules (N.J.A.C. 7:7E). Delaware's Endangered Species law prohibits the importation, transportation, possession, or sale of endangered animal species and their parts, as well as the sale or possession with the intent to sell of products made from endangered animals or their parts (7 Del.C. § 601 - 605). Under the Georgia Nongame and Endangered Species Conservation Act (Code 1976 § 50-15-10 – 90), red knots cannot be captured, killed, or sold, and their habitat is protected on public lands; however, Georgia law specifically states that rules and regulations related to the protection of State-protected species shall not affect rights in private property.

A total of 35 red knot occurrence areas in Maine (81 percent of all documented sites in that State) benefit from special regulatory protection under Maine State law. Thirty-four sites are classified as "high or moderate value shorebird feeding, roosting and staging areas" and designated as "Significant Wildlife Habitat" under Maine's Natural Resources Protection Act (Maine Revised Statutes and Annotations: Title 38, Sections 480-B.10 and 480-FF) and related regulations adopted under this law (Maine Department of Environmental Protection (MDEP) rules Chapter 335). Four red knot occurrences overlap with "Essential Habitat" designations (under the Maine Endangered Species Act) to protect nesting areas for piping plovers (Maine Revised Statutes and Annotations: Title 12, Sections 12804 and 12806 and related regulations; MDIFW rules Chapter 8.05). The MDIFW provides advisory input to permitting decisions in "Significant Wildlife Habitats" and is the lead review agency for "Essential Habitats" in Maine (MDIFW 2013).

United States—Federal Lands

Some red knot concentration areas occur on military bases. The Sikes Act (16 U.S.C. 670) authorizes the Secretary of Defense to develop cooperative plans with the Secretaries of Agriculture and the Interior for natural resources on public lands. The Sikes Act Improvement Act of 1997 requires Department of Defense installations to prepare Integrated Natural Resource Management Plans (INRMPs) that provide for the conservation and rehabilitation of natural resources on military lands consistent with the use of military installations to ensure the readiness of the Armed Forces. The INRMPs incorporate, to the maximum extent practicable, ecosystem management principles and provide the landscape necessary to sustain military land uses. While their implementation is subject to funding availability, INRMPs can be an added conservation tool in promoting the recovery of endangered and threatened species on military lands. We have identified one military base with an approved INRMP that explicitly addresses and benefits the red knot, Eglin Air Force Base in northwest Florida. However, INRMPs are not regulatory mechanisms and so are not considered further under Factor D.

Several red knot concentration areas occur in National Seashores or other units of the NPS, which must balance visitation and recreation with the protection of natural resources such as the red knot and its habitat. The National Park Service Organic Act of 1916, as amended (39 Stat. 535, 16 U.S.C. 1) (NPSOA), states that the NPS "shall promote and regulate the use of [NPS units]…to conserve the scenery and the national and historic objects and the wildlife therein and to provide for the enjoyment of the same in such manner and by such means as will leave them unimpaired for the enjoyment of future generations." By policy, NPS (2007) has interpreted the "impairment" standard and made the following findings. The fundamental purpose of all parks includes providing for the enjoyment of park resources and values by the people of the United States. "Enjoyment" means enjoyment both by people who directly experience parks and by those who appreciate them from afar, and includes more than recreation. When there is a conflict between conserving resources and values and providing for enjoyment of them, conservation is to be predominant. Impairment may occur from visitor activities; NPS activities in the course of managing a park; or activities undertaken by concessioners, contractors, and others operating in the park. The NPS has management discretion to allow impacts to park resources and values when necessary and appropriate to fulfill the purposes of a park, so long as the impact does not constitute impairment of the affected resources and values. In these situations, the NPS will ensure that the impacts are unavoidable and cannot be further mitigated. Rarely is there clear-cut evidence that impairment will occur. Superintendents and other NPS decision-makers apply their professional judgment to the facts of each case, taking into account technical and scientific studies and other information provided by subject matter experts (NPS 2007). In addition to the NPSOA, red knots may benefit from a 2010 nonregulatory Memorandum of Understanding (MOU) between the NPS and the Service regarding migratory birds that was executed pursuant to Executive Order 13186 (see Coastal Management, below); section F.4. of the MOU states that the NPS will identify and protect natural habitats of migratory bird species within park boundaries.

Several red knot concentration areas occur in NWRs, which are administered by the Service. The National Wildlife Refuge System Improvement Act of 1997 (16 U.S.C. 668dd *et seq.*) (NWRSIA) establishes the protection of biodiversity as the primary purpose of the NWR system; recreational and other uses of a NWR may only be approved if the Service finds such use to be compatible with the purposes of that individual NWR and the purposes of the NWR system. As the primary planning documents guiding management of NWRs, Comprehensive Conservation Plans typically set goals and list needed actions to protect and enhance populations of key wildlife species on refuge lands.

United States—Coastal Management

As discussed in the proposed rule under Factors A and E, shoreline stabilization has significant impacts on red knot habitats, and can also impact knots through disturbance and via impacts on prey resources. Federally funded shoreline stabilization is typically carried out by the USACE as authorized by a series of Water Resources Development Acts (WRDA), the most recent of which was passed in 2007 (P.L. 110–114). The 2007 WRDA continued Federal authorization for projects including flood damage reduction, stream bank protection, navigation, ecosystem restoration, shoreline protection (e.g., stabilization), and sediment removal (e.g., dredging).

In addition to its role in constructing shoreline stabilization projects, the USACE also administers a permitting program for certain projects in or near the intertidal habitats that support red knots. Many such projects require USACE permits under section 404 of the Clean Water Act (i.e., Federal Water Pollution Control Act) (33 U.S.C. 1251 *et seq*.), as amended, which establishes a program to regulate the discharge of dredged or fill material into waters of the United States, including wetlands. Activities regulated under section 404 include fill for development, water resource projects (such as dams and levees), infrastructure development (such as highways and airports), and mining. Section 404 requires a permit before dredged or fill material may be discharged into waters of the United States, unless the activity is exempt from section 404 regulation (e.g., certain farming and forestry activities). Under the section 404 program, no discharge of dredged or fill material may be permitted if a practicable alternative exists that is less damaging to the aquatic environment, or if the nation's waters would be significantly degraded. In addition to section 404 permits, some coastal projects require USACE permits under section 10 of the Rivers and Harbors Appropriation Act of 1899 (30 Stat. 1151, as amended; 33 U.S.C. 403 *et seq*.), which regulates the placement of structures in U.S. navigable waters. In addition to USACE permits, the U.S. Coast Guard (USCG) administers a permitting program under section 9 of the Rivers and Harbors Appropriation Act of 1899, which regulates the construction of bridges, causeways, and dams in navigable waters.

Federal funding or authorization for a project triggers several environmental requirements that may afford some protections to red knots or their habitats. The National Environmental Policy Act 42 U.S.C. §4321 *et seq*. (1969) (NEPA) requires Federal action agencies to assess the likely impacts from their proposed action as well as various alternative courses of action. However, NEPA does not mandate that Federal agencies include any specific environmental protections in the final project plans, and is therefore considered nonregulatory. Also nonregulatory in nature, Executive Order 13186, Responsibilities of Federal Agencies to Protect Migratory Birds, directs Federal agencies to evaluate the effects of their proposed actions and plans on migratory birds in the course of their NEPA analyses. Because NEPA and Executive Order 13186 are nonregulatory, they are not considered further under Factor D.

The Coastal Barrier Resources Act (P.L. 97-348) (96 Stat. 1653; 16 U.S.C. 3501 *et seq.*), as amended by the Coastal Barrier Improvement Act of 1990 (P.L. 101-591; 104 Stat. 2931) designated relatively undeveloped coastal barriers along the Atlantic and Gulf coasts as part of the John H. Chafee Coastal Barrier Resources System and made these areas ineligible for most new Federal expenditures and financial assistance, including Federal flood insurance that can promote development. The goal of these laws is to remove Federal incentives for the development of coastal barriers (e.g., barrier islands), because such development can lead to loss of natural resources, threats to human life and property, and imprudent expenditure of tax dollars. These restrictions on development likely benefit red knots in some areas.

The Coastal Zone Management Act of 1972 (P.L. 92-583) (86 Stat. 1280; 16 U.S.C. 1451-1464) (CZMA) provides Federal funding to implement the States' federally approved Coastal Zone Management Plans, which guide and regulate development and other activities within the designated coastal zone of each State. To be federally approved, a State plan must identify areas needed to protect, maintain, or replenish coastal lands or resources including coastal flood plains, aquifers and their recharge areas, estuaries, sand dunes, reefs, beaches, offshore sand deposits, and mangrove stands; include a definition of the term "beach" and a planning process for the protection of, and access to, public beaches and other public coastal areas of environmental, recreational, historical, esthetic, ecological, or cultural value; provide for the management of those land and water uses having a direct and significant impact on coastal waters and those geographic areas that are likely to be affected by or vulnerable to sea level rise; and assure the appropriate protection of those significant resources and areas, such as wetlands, beaches, dunes, and barrier islands, that make that State's coastal zone a unique, vulnerable, or valuable area (15 CFR Part 923). All eligible States in the red knot's U.S. range (including the Great Lakes) have approved Coastal Zone Management Plans (NOAA 2012c, p. 2). In those States with approved plans, the CZMA requires Federal action agencies to ensure that the activities they fund or authorize are consistent, to the maximum extent practicable, with the enforceable policies of that State's federally approved coastal management program; this provision of CZMA is known as Federal consistency (NOAA 2012c, p. 2).

Titus (2000, p. 743) reviewed the shoreline armoring and beach nourishment policies of all 18 coastal States from Maine to Texas. The States of Maine, Maryland, North Carolina, South Carolina, Mississippi, and Texas had policies to prohibit the armoring of ocean beaches, while the remaining 12 states allowed for at least some oceanfront armoring. In 2011, the State of North Carolina authorized an exception to its 1985 ban on new oceanfront structures, to allow for the construction of up to four new terminal groins (Rice 2012a, p. 7). Thus, 72 percent of Atlantic or Gulf coast States (13 of 18) allow for some new hard structures along the oceanfront beach. Titus (2000, p. 743) found that only Maine and Massachusetts had policies to prohibit the armoring of bays and sounds, with the other 16 States allowing these practices. Every State from Maine to Texas allowed oceanfront beach nourishment, although beach nourishment of bays and sounds was permitted in only 7 of these 18 States (Titus 2000, p. 743). Some States regulate the manner in which beach nourishment is carried out. For example, North Carolina's Technical Standards for Beach Fill (15A NCAC 07H .0312) address sediment compatibility of material proposed to be placed on beaches. (Ensuring material compatibility is important to protecting shorebird foraging habitat.) Due to the Federal consistency provision of CZMA, the policies of each State are generally also followed by Federal agencies in determining if coastal projects may be federally funded or authorized under the statutes discussed above (e.g., WRDA, section 404 of the Clean Water Act, Rivers and Harbors Act). However, North Carolina's Technical Standards are an exception—because the Standards were not submitted for Coastal Consistency, federally funded USACE projects do not have to abide by them (K. Matthews pers. comm. May 2, 2014).

State policies regarding beach grooming and sand removal or transfers are varied, and we do not have comprehensive information for each State. Above the high tide line, these activities are typically not regulated by the USACE, and thus fall under State and local jurisdictions. In those jurisdictions for which information is available, beach grooming is generally permitted in red knot habitat, including while the birds are present. Maine's Coastal Sand Dune Rules (Chapter 355) state that no review or permit is required for removal of debris from a beach, provided that little or no sand is removed with the debris, or for the removal of seaweed from the beach by hand or mechanical means provided the seaweed is not removed from the coastal sand dune system and does not disturb dune vegetation. The Massachusetts Wetlands Protection regulations (310 CMR 10.27) state that vegetative debris along the drift line is vital for resident and migratory shorebirds, which feed largely on invertebrates that eat the vegetation, and below the drift line in the lower intertidal zone are infauna (invertebrates such as mollusks and crustacea), which are also eaten by shorebirds; however, these regulations do not prohibit mechanical beach grooming that would remove such vegetative debris. New Jersey's Coastal Zone rules (N.J.A.C. 7:7E) seasonally restrict beach raking in nesting areas for piping plovers and State-listed beach species, and limit mechanical sifting and beach raking to recreational beach areas within 300 ft (91 m) of a staffed lifeguard stand. Florida regulates mechanized beach cleaning under its Beach and Shore Preservation Act (Florida Statute 161) (Florida Shorebird Alliance 2012); special conditions must be followed for beach cleaning to occur during the sea turtle nesting season. The City of South Padre Island, Texas strives to rake the beaches only when there is a significant amount of seaweed present; any seaweed removed is separated from nonnatural material and then placed at the toe of the dunes for possible future use in dune restoration (City of South Padre Island 2013).

United States—Invasive Species Control

Several Federal laws and policies relate to the control of invasive species. Invasive vegetation can affect red knot habitat (Factor A), while nonnative marine species can threaten red knot food supplies and facilitate the spread of harmful algal species (Factor E). Under Executive Order 13112, Invasive Species, Federal agencies may not authorize, fund, or carry out actions that are likely to cause or promote the introduction or spread of invasive species in the United States or elsewhere unless, pursuant to guidelines that it has prescribed, the agency has determined and made public its determination that the benefits of such actions clearly outweigh the potential harm caused by invasive species, and that all feasible and prudent measures to minimize risk of harm will be taken in conjunction with the actions.

The Plant Protection Act of 2000 (P.L. 106-224) regulates the movement of noxious weeds, which are defined as any plant or plant product that can directly or indirectly injure or cause damage to crops or other interests of agriculture, navigation, the natural resources of the United States, the public health, or the environment. The USDA publishes, by regulation, a list of noxious weeds that are prohibited or restricted from entering the United States or that are subject to restrictions on interstate movement within the United States. Of the invasive plant species discussed in the proposed rule under Factor A, none are on the Federal list, but *Carex kobomugi* is listed as a noxious weed by the States of Massachusetts and Connecticut, and *Casuarina* species are considered noxious weeds by the State of Florida. By policy, the USDA considers a plant species invasive only when it occurs on the Federal or a State-specific noxious weed list or a similar State list.

Regarding invasive marine species, the Aquatic Nuisance Species Task Force is an intergovernmental organization dedicated to preventing and controlling aquatic nuisance species and to implementing the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990 (P.L. 101-646), which was expanded with the passage of the National Invasive Species Act (NISA) (P.L. 104-332) in 1996. Under NISA, the USCG established both regulations and guidelines to prevent the introduction of aquatic nuisance species via ship ballast water. The USCG's final ballast water rule was published in the Federal Register on March 23, 2012, and became effective on June 21, 2012 (77 FR 17254). The USCG amended its regulations on ballast water management by establishing a standard for the allowable concentration of living organisms in ballast water discharged from ships in waters of the United States. The USCG also amended its regulations for engineering equipment by establishing an approval process for ballast water management systems (USCG 2013). Although the Aquatic Nuisance Species Task Force is still actively working to reduce the risk of new introductions, and the harmful effects of existing aquatic nuisance species, several funding provisions of NISA expired in 2002 and have not been reauthorized or replaced.

Under the Lacey Act (18 U.S.C. 42; 50 CFR 16), species listed as injurious may not be imported into the United States or transported between the continental United States, the District of Columbia, Hawaii, the Commonwealth of Puerto Rico, or any territory or possession of the United States by any means without a permit issued by the Service. The Service implements the injurious wildlife provisions of the Lacey Act through regulations at 50 CFR part 16. Species are added to the list of injurious wildlife to prevent their introduction or establishment through human movement in the United States. Regulation of transport or use within a State is the responsibility of each State. Possession of a species within State boundaries is also the responsibility of each State and is not regulated by an injurious wildlife listing. Injurious wildlife are defined as vertebrates, crustaceans, mollusks, and their offspring that are injurious to the interests of human beings, agriculture, horticulture, forestry, or the wildlife resources of the United States (USFWS 2010, p. 1). Because Asian horseshoe crabs are not among the taxa eligible for listing as injurious, the Service currently lacks the regulatory authority to restrict their importation (USFWS 2013, pp. 1–2), although Congress is deliberating legislation to expand that authority to include nonnative horseshoe crabs (National Environmental Coalition on Invasive Species (NECIS) 2014; USFWS 2013, pp. 1–2). Absent Federal authority to regulate these species, individual States may restrict the import of nonnative horseshoe crabs. For example, imports of Asian horseshoe crabs have been prohibited in Delaware and South Carolina, and restrictions are being formulated in Rhode Island and Maryland (M. Hawk pers. comm. June 2, 2014; DDNREC 2013). (See Factor E—Reduced Food Availability—Horseshoe Crab Harvest.)

In addition to their introduction via ballast water (regulated by USCG) and deliberate import (regulated by the Service), nonnative marine and estuarine species can also be introduced into red knot habitats via aquaculture, and can involve pathogens as well as marine invaders and harmful algal species. In addition, red knot habitats can be directly converted to aquaculture facilities (see Factor A—Agriculture and Aquaculture). Aquaculture in the United States is regulated at both the Federal and State level. At the Federal level, the primary agencies include the Food and Drug Administration (FDA), the USDA, and the USEPA. The USEPA is responsible for wastewater permitting across all industries, while the FDA covers food safety regulations and drug approvals. Several other Federal agencies and programs, including NOAA and the Service, are involved indirectly in aquaculture activities. A coordinating body, the Joint Subcommittee on Aquaculture, was created by enactment of the National Aquaculture Act of 1980 (P.L. 96-362), as amended, which promotes aquaculture and is nonregulatory in nature. Federal regulatory statutes rarely address aquaculture directly, but collectively these laws (e.g., Federal Water Pollution Control Act; Food, Drug and Cosmetic Act; Animal Drug Availability Act; Magnuson-Stevens Fisheries Conservation Act; Lacey Act; Coastal Zone Management Act; and Virus Serum Toxin Act) provide the statutory framework for regulating food safety, veterinary medicines, fisheries, coastal zone management, and other activities related to aquaculture (National Agricultural Law Center undated).

State and local governments generally regulate activities that are permitted or licensed at the community level. In aquaculture, the majority of operations fall into this regulatory scheme. Generally, permits deal with zoning, building, water use, waste discharge, species certification related to wildlife management, marketing or processing, and trade. Often, regulations differ based on the position of the operation—inland, wetland, coastal, or offshore. Due mainly to environmental concerns, requirements for each type of operation are varied, with each State administering permits based on its own specific rules. There are no consistent or universal laws and regulations of aquaculture among the several States; thus, regulations can vary considerably between geographic locations (National Agricultural Law Center undated).

United States—Regulation of Other Threats

Reduced food availability at the Delaware Bay stopover site due to commercial harvest of the horseshoe crab is considered a primary causal factor in the decline of the rufa red knot in the 2000s (see Factor E—Reduced Food Availability). The Atlantic Coastal Fisheries Cooperative Management Act of 1993 set forth the current role of the ASMFC, which had been established under an interstate compact among all States from Maine to Florida and previously approved by Congress (P.L. 77-539 and 81-721). Under the 1993 law, the ASMFC develops coastal fishery management plans and monitors each State’s compliance with the plans. If a State fails to implement and enforce a fishery plan, NOAA declares a moratorium in the fishery in question within the waters of the noncomplying State. The ASMFC adopted a horseshoe crab management plan in 1998, with different provisions for the bait industry versus the biomedical industry. In 2012, the ASMFC adopted Addendum VII to the plan, which utilizes an Adaptive Resource Management (ARM) framework to manage the bait fishery in the Delaware Bay Region (New Jersey, Delaware, and parts of Maryland and Virginia) (ASMFC 2012a, p. 2). Under the ARM, bait harvest levels are tied to red knot populations via scientific modeling. There have been no instances of State noncompliance with the horseshoe crab management plan. In 2008, New Jersey enacted a law (N.J.S.A. 23.2b.21) extending an earlier (2006) Statewide moratorium on the bait harvest until specific red knot recovery targets are achieved. Thus, New Jersey does not use its bait harvest quota as allocated by the ASMFC. Regulation of the horseshoe crab harvest is discussed further under Factor E.

On December 15, 2009, the USEPA published in the Federal Register (74 FR 66496) a rule titled, "Endangerment and Cause or Contribute Findings for Greenhouse Gases Under Section 202(a) of the Clean Air Act." In this rule, the USEPA Administrator found that the current and projected concentrations of the six long-lived and directly emitted GHGs—carbon dioxide, methane, nitrous oxide, hydrofluorocarbons, perfluorocarbons, and sulfur hexafluoride—in the atmosphere threaten the public health and welfare of current and future generations, and that the combined emissions of these GHGs from new motor vehicles and new motor vehicle engines contribute to the GHG pollution that threatens public health and welfare (74 FR 66496). In effect, the USEPA has concluded that the GHGs linked to climate change are pollutants, whose emissions are subject to the Clean Air Act (42 U.S.C. 7401 *et seq.*) (see 74 FR 66496). The USEPA subsequently has developed several initiatives related to GHGs (USEPA 2014). These include adoption in 2011 of GHG standards and fuel efficiency standards for medium- and heavy-duty engines and vehicles (76 FR 57106), and adoption in 2012 of regulations related to oil and gas development and processing (77 FR 49489). More recently, USEPA published a proposed rule for regulating carbon pollution from new power plants (January 8, 2014; 79 FR 1430), and a proposed rule for regulating carbon pollution from existing power plants, the largest current source of greenhouse gas emissions in the United States (June 18, 2014; 79 FR 34829). However, because these proposed regulations are not final, we cannot consider them as existing regulatory mechanisms. In addition, although adoption of USEPA's proposed rules would contribute to efforts to reduce GHGs, even when combined with reductions resulting from the rules adopted in 2011 and 2012, it is clear that substantially greater reductions in GHGs would still be needed through measures at multiple scales (global to sub-national) in order to reduce the magnitude of likely changes in climate and related effects projected over the next several decades (Pachauri *et al.* 2014, pp. 58–133). Thus, we have no basis to conclude that implementation of the Clean Air Act now or over the next several decades will reduce the ongoing or likely future climate change effects on the rufa red knot.

Recreational activities, including off-road vehicle (ORV) use, can impact red knots through disturbance, and through effects on prey resources. The MBTA prohibits direct take of migratory birds including red knots on both Federal and non-Federal lands. However, recreational activities occurring within the red knot's U.S. range seldom cause direct mortality; rather, recreational activities typically cause disturbance of and other impacts to (including indirect take of) migratory birds. On Federal lands, the development and implementation of recreation management regulations and policies are subject to several of the statutes, orders, and policies discussed above, including the MBTA, the Sikes Act, NPSOA, NWRSIA, NEPA, and Executive Order 13186 (Responsibilities of Federal Agencies to Protect Migratory Birds); collectively, these laws and policies strongly encourage Federal land managers to consider the effects of red knot disturbance and prey availability as a result of recreational disturbance. On non-Federal lands, recreation is managed by a patchwork of State and local laws, many of which are contingent upon the land ownership of thousands of individual parcels along the coasts. We lack information regarding most of the existing non-Federal recreation management policies and their effects on the red knot. However, we are aware of only a few locations (e.g., portions of the Delaware bayshore) in which beaches are closed, regulated, or patrolled to protect nonbreeding shorebirds.

The Harmful Algal Bloom and Hypoxia Amendments Act of 2004 (Public Law 108-456) authorizes funding for research on harmful algal blooms (HABs) and hypoxia to advance scientific understanding and our ability to detect, assess, predict, control, and mitigate events. However, this law is nonregulatory. To the extent that HABs may be caused or intensified by poor water quality, section 402 (the National Pollutant Discharge Elimination System) and other provisions of the Clean Water Act likely reduce these effects, through discharge requirements and by seeking to achieve minimum surface water quality standards. Regulatory provisions relevant to the spread of harmful algal species (e.g., USCG ballast water regulations, the aquaculture regulatory framework) are discussed above.

The Oil Pollution Act of 1990 (P.L. 101-380) (104 Stat. 484; 33 U.S.C. 2701 *et seq.*) (OPA) expanded the ability of Federal agencies to respond to oil spills. The OPA also created the national Oil Spill Liability Trust Fund, which is available to provide up to one billion dollars per spill incident. In addition, the OPA provided new requirements for contingency planning by both government and industry in a three-tiered approach: the Federal government is required to direct all public and private response efforts for certain types of spill events; Area Committees (composed of Federal, State, and local officials) must develop detailed, location-specific Area Contingency Plans; and owners or operators of vessels and certain facilities that pose serious threats to the environment must prepare their own Facility Response Plans. The USEPA has published regulations for aboveground storage facilities, and the USCG has done so for oil tankers. The OPA also increased penalties for regulatory noncompliance, broadened the response and enforcement authorities of the Federal government, and preserved State authority to establish laws governing oil spill prevention and response (USEPA 2011). All oil and gas operations on the Outer Continental Shelf (OCS) (e.g., exploration, extraction) are governed by laws and regulations to ensure safe operations and preservation of the environment (50 CFR 203-291). The Bureau of Safety and Environmental Enforcement (BSEE) within the Department of the Interior (DOI) enforces these regulations and periodically updates rules to reflect changes in technology and new information.

The USEPA and the States register or license pesticides for use in the United States. The USEPA receives its authority to register pesticides under the Federal Insecticide, Fungicide, and Rodenticide Act (7 U.S.C. §136 *et seq.*) (FIFRA). States are authorized to regulate pesticides under FIFRA and under State pesticide laws. States may place more restrictive requirements on pesticides than USEPA. Pesticides must be registered both by USEPA and the State before distribution.

The construction and operation of terrestrial wind turbines are potentially subject to various Federal regulations. The MBTA applies to all activities (both Federal and non-Federal) that result in the “take” of migratory birds, including the construction and operation of wind turbines. To help both Federal and non-Federal project proponents minimize the risk of take under the MBTA, the Service has produced voluntary Land-Based Wind Energy Guidelines, to provide a structured, scientific process for addressing wildlife conservation concerns at all stages of land-based wind energy development (USFWS 2012d, p. vi). Because the MBTA does not allow for the authorization of take that is incidental to an otherwise lawful activity ("incidental take"), the Service cannot authorize the take of red knots or other migratory birds caused by collisions with wind turbines. The Service makes decisions whether to refer for prosecution any alleged take of migratory birds at wind energy facilities, and takes into account the adherence of the developer or operator with the voluntary guidelines (USFWS 2012d, p. 6).

In addition to MBTA, other Federal regulatory mechanisms may apply to terrestrial wind energy development, depending on the role (if any) of a Federal agency in turbine construction and operation (i.e., the nature of the Federal nexus, if any). For wholly non-Federal projects, section 10 of the Endangered Species Act (incidental take permits for listed species upon completion of a Habitat Conservation Plan (HCP)) would apply and can provide protection for nonlisted species, but only if the section 10 permittee chooses to include the nonlisted species as a species covered by the HCP. For wind energy projects that are federally funded (e.g., by the USDA's Rural Energy for America Program) or authorized (e.g., if a section 404 wetland permit is required) or located on Federal land, several of the regulations and policies described above would apply, such as NEPA and Executive Order 13186; however, as these measures are nonregulatory, we do not consider these further here. Additional Federal regulations and policies (e.g., NPSOA, NWRSIA) apply to any wind energy development on Federal land. However, many terrestrial wind energy development projects lack any Federal nexus (e.g., Newstead 2014a, p. 26) and are thus subject only to State and local approvals.

Regarding offshore wind energy development, section 388 of the Energy Policy Act of 2005 granted the DOI discretionary authority to issue leases, easements, or rights-of-way for activities on the OCS that produce or support production, transportation, or transmission of energy from sources other than oil and gas, and that are not otherwise authorized by other applicable law. The DOI has delegated this authority to BOEM, which has jurisdiction over projects on the OCS including but not limited to offshore wind energy, wave energy, ocean current energy, offshore solar energy, and hydrogen generation, as well as other projects that make alternate use of existing oil and natural gas platforms in Federal waters on the OCS. Under NEPA, the BOEM has prepared a Programmatic Environmental Impact Statement setting forth policies and best management practices, and has promulgated regulations and guidelines (Department of Energy (DOE) and Bureau of Ocean Energy Management, Regulation, and Enforcement (BOEMRE) 2011, p. iii).

United States—Summary

The MBTA and state wildlife laws protect the red knot from direct take resulting from scientific study and hunting. The Sikes Act, NPSOA, and NWRSIA provide protection for the red knot from habitat loss and inappropriate management on many Federal lands. Although shorebirds are not their focus, some laws do regulate shoreline stabilization and coastal development, including section 404 of the Clean Water Act, the Rivers and Harbors Act, the Coastal Barrier Resources Act, and the Coastal Zone Management Act as implemented by Federal and State regulations. We have limited information regarding State and local regulations regarding beach cleaning or recreational disturbance. Several Federal and state policies are in effect to stem the introductions and effects of invasive species, but collectively these do not provide complete protection for the red knot from impacts to its habitats or food supplies resulting from beach or marine invaders or the spread of harmful algal species. Although threats to the horseshoe crab egg resource remain (see Factor E—Reduced Food Supplies), regulatory management of the horseshoe crab fishery under the ARM is adequate to address threats to the knot's Delaware Bay food supply from direct harvest. Although we lack information regarding the overall effect of recreation management policies on the red knot, we are aware of only a few locations in which beaches are closed, regulated, or monitored to protect nonbreeding shorebirds. Relatively strong Federal laws likely reduce risks to red knots from oil spills, but cannot fully abate the risks of oil spills and leaks (see Factor E—Oil Spills and Leaks). Similarly, existing Federal laws and policies are likely to reduce the red knot's collision risks from new wind turbine development, but some level of mortality is expected upon buildout of the Nation's wind energy infrastructure.

Factor D—International Laws and Regulations

Internationally, there are different laws among nations that affect aquaculture. However, the United Nations (UN) has played a significant role in the development of international law for seas and fisheries, directly impacting coastal or open ocean aquacultural operations. The 1982 UN Conference on the Law of the Sea (UNCLOS) set offshore territorial boundaries that establish zones of exclusive economic and fisheries rights for coastal nations. While some nations have not ratified this convention, it is the de facto set of guidelines for the world's oceans. Furthermore, the UN has developed a Code of Conduct for Responsible Fisheries, based on international laws including UNCLOS (National Agricultural Law Center undated).

International regulations can also slow or halt the spread of diseases that could potentially affect red knots. For example, many countries have applied trade restrictions on the import of birds and their products from countries affected by the H5N1 avian influenza virus, by invoking the Agreement on the Application of Sanitary and Phytosanitary Measures of the World Trade Organization (Fidler 2004). The World Health Organization (WHO) has protocols for the containment of diseases such as pandemic influenza (WHO 2007, entire).

The International Maritime Organization (IMO) is the UN specialized agency with responsibility for the safety and security of shipping and the prevention of marine pollution by ships (IMO 2013). The 1969 International Convention Relating to Intervention on the High Seas in Cases of Oil Pollution Casualties affirms the right of a coastal nation to take such measures on the high seas as may be necessary to prevent, mitigate or eliminate danger to its coastline or related interests from pollution by oil or the threat thereof, following upon a maritime casualty; the coastal nation is, however, empowered to take only such action as is necessary, and after due consultations with appropriate interests. The 1973 International Convention for the Prevention of Pollution from Ships (MARPOL) is the main international convention covering prevention of pollution of the marine environment by ships from operational or accidental causes. The MARPOL has been updated by amendments through the years. Since 1992, it has been mandatory for new oil tankers to have double hulls, and a schedule has been implemented for existing tankers to be retrofitted with double hulls. Other MARPOL provisions prohibit the discharge of noxious substances within 12 mi (19 km) of the nearest land (IMO 2013).

In summary, the existing international regulatory mechanisms are likely reducing threats to the red knot from oil spills, other contaminants, and disease incidental to the pursuit of other goals.

Factor D—Summary

In Canada, SARA provides protections for the red knot and its habitat, both on and off of Federal lands. The red knot is afforded additional protections under the MBCA and by provincial law in four of the Provinces. Red knots are legally protected from direct take and hunting in several Caribbean and Latin American countries, but we lack information regarding the implementation or effectiveness of these measures. For many other countries, red knot hunting is unregulated, or we lack sufficient information to determine if red knot hunting is legal. We also lack information for countries outside the United States regarding the protection or management of red knot habitat, and regarding the regulation of other activities that threaten the red knot such as development, disturbance, oil spills, environmental contaminants, and wind energy development.

The MBTA and state wildlife laws protect the red knot from direct take resulting from scientific study and hunting. The Sikes Act, NPSOA, and NWRSIA provide protection for the red knot from habitat loss and inappropriate management on Federal lands. Section 404 of the Clean Water Act, the Rivers and Harbors Act, the Coastal Barrier Resources Act, the Coastal Zone Management Act, and State mechanisms regulate shoreline stabilization and development. State and local regulations provide varying levels of protection from impacts associated with beach grooming. Several Federal and State policies are in effect to stem the introductions and effects of invasive species, but collectively do not provide complete protection to the red knot from impacts to its habitats or food supplies resulting from beach or marine invaders or the spread of harmful algal species. Although threats to the horseshoe crab egg resource remain (see Factor E—Reduced Food Supplies), regulatory management of the horseshoe crab fishery under the ARM is adequate to address threats to the knot's Delaware Bay food supply from direct harvest. Regarding climate change, the USEPA has developed several initiatives related to GHGs. However, some of the USEPA's proposed GHG regulations are not yet final and, even when final, substantially greater reductions in GHGs would still be needed at multiple scales to reduce the magnitude of likely climate changes over the next several decades. Although we lack information regarding the overall effect of recreation management policies on the red knot, we are aware of a few locations in which beaches are closed, regulated, or monitored to protect nonbreeding shorebirds. Relatively strong Federal laws likely reduce risks to red knots from oil spills, but cannot fully abate the risk of oil spills and leaks. Similarly, Federal law and policy reduce the red knot's collision risks from new wind turbine development, but some level of mortality is expected upon buildout of the Nation's wind energy infrastructure.

FACTOR E. OTHER NATURAL OR MANMADE FACTORS AFFECTING ITS CONTINUED EXISTENCE

In this section, we present and assess the best available information regarding a range of other ongoing and emerging threats to the red knot, including reduced food availability, asynchronies ("mismatches") between the timing of the red knot's annual cycle and the windows of optimal food and weather conditions on which it depends, human disturbance, oil spills, environmental contaminants, and wind energy development.

Factor E—Reduced Food Availability

Declining food resources can have major implications for the survival and reproduction of long-distance migrant shorebirds (IWSG 2003, p. 10). The life history of long-distance, "jump" migrant shorebirds indicates that the availability of abundant food resources at temperate stopovers is critical for completing their annual cycle (USFWS 2003, p. 4). Commercial harvest of horseshoe crabs has been implicated as a causal factor in the decline of the rufa red knot, by decreasing the availability of horseshoe crab eggs in the Delaware Bay stopover (Niles *et al.* 2008, pp. 1–2). Notwithstanding the importance of the horseshoe crab and Delaware Bay, other data indicate that the rufa red knot also faces threats to its food resources throughout its range.

Documented changes to prey species are already underway but not always fully understood. In Australia's Roebuck Bay, for example, a sharp decline has been documented in a clam species that had been a key prey resource for *Calidris canutus*. The causes for this decline, its likely duration, and impacts to *C. canutus* are unknown (Vivian 2013a). Specific to the rufa red knot, additional examples of threats to food resources, from known and unknown causes, are discussed below. However, red knots in localized areas have shown some ability to switch prey when the preferred prey species became reduced (Escudero *et al.* 2012, pp. 359, 362; Musmeci *et al.* 2011, entire), suggesting some adaptive capacity for the species to cope with this threat under certain circumstances.

The following discussion addresses known or likely threats to the abundance or quality of red knot prey, while potential food shortages caused by asynchronies (timing "mismatches") in the red knot's annual cycle are discussed in the next section (Factor E—Asynchronies). In addition to the factors affecting red knot prey discussed in this section, see Harmful Algal Blooms and Oil Spills and Leaks, below, for consideration of these additional threats to red knot prey species.

Food Availability—Marine Harvest

See Horseshoe Crab Harvest, below, for a full discussion of the role this fishery is thought to have played in the decline of rufa red knots in the 2000s. Also see Factor A—Agriculture and Aquaculture regarding clam farming in Canada, aquaculture in Argentina, and seaweed farming on Chiloè Island, Chile.

Brian Harrington (pers. comm. November 14, 2013) noted that red knots formerly had major stopover locations on the Massachusetts coast (e.g., Scituate, Plymouth) where the birds were attracted by vast beds of mussels and their spat (see Population Surveys and Estimates—Fall Stopover Areas). Commercial harvesting of mussels began in the early 1980s, around the same time that numbers of knots using these stopovers began to decline (B. Harrington pers. comm. November 14, 2013). It is unknown if the mussel harvest was a factor in the declining red knot use of this area, which may have also been influenced by coastal erosion (see Other Aspects of Climate Change, below) (B. Harrington pers. comm. November 14, 2013), overall declines in the population size of southern-wintering red knots (Harrington *et al.* 2010b, p. 361), or both.

In other *Calidris canutus* subspecies, commercial shellfish harvests have been linked to local decreases in recruitment and possibly emigration in a wintering area in England (Atkinson *et al.* 2003a, p. 127); increased gizzard sizes (possibly to grind lower quality, i.e., thicker shelled, prey) and decreases in local survival in a wintering area in the Dutch Wadden Sea (van Gils *et al.* 2006, p. 2399); and prey switching and reduced habitat use in a wintering and stopover area in the Dutch Wadden Sea (Piersma *et al.* 1993, pp. 343, 354). Harvest activities have also been shown to impact prey availability for other *Calidris* species. For example, foraging efficiency of semipalmated sandpipers decreased nearly 70 percent after 1 year of baitworm harvesting in the Bay of Fundy, concurrent with habitat changes and a 39 percent decrease in the sandpiper's preferred amphipod prey (Shepherd and Boates 1999, p. 347). These examples show that species closely related to the red knot are vulnerable to reductions in food resources from marine harvest.

Food Availability—Ocean Acidification

Oceans become more acidic as carbon dioxide gas, the primary GHG, emitted into the atmosphere dissolves in the seawater ("very high confidence"), forming carbonic acid and changing the chemical composition of the ocean (IGBP *et al.* 2013, p. 14). The oceans have absorbed approximately 30 percent of all carbon dioxide released into the atmosphere by humans since the start of the industrial revolution, causing the pH (percent hydrogen, a measure of acidity or alkalinity) level of the oceans to decrease by approximately 0.1 pH units ("high confidence"), equivalent to a 26 percent increase in acidity (IGBP *et al.* 2013, p. 1; IPCC 2013a, pp. 12, 14). The oceans continue to acidify at a rate unprecedented in Earth's history ("high confidence"). The current rate of acidification may be faster than at any time in the last 300 million years, and is over 10 times faster than at any time in the past 55 million years (IGBP *et al.* 2013, pp. 2, 5, 14). The IPCC (2013a, p. 27) projects a global increase in ocean acidification under all GHG emissions scenarios, with a decrease in surface ocean pH of 0.06 to 0.30 by 2100, or as much as a 170 percent increase in acidity relative to preindustrial levels (IGBP *et al.* 2013, p. 5).

As ocean acidification increases, the availability of carbonate ions declines. Calcium carbonate is a key building block for the shells of many marine organisms, including bivalves and other mollusks (IGBP *et al.* 2013, p. 17; USEPA 2012; NRC 2010a, p. 286) that are the primary food for red knots during most of the year (see Migration and Wintering Food). Calcium carbonate has two main forms: calcite and aragonite. Aragonite, which is formed by the first larval stages of many mollusks and by some adult mollusks, is more soluble than calcite. Both aragonite and calcite dissolve when carbonate ion concentrations are low (known as "undersaturated conditions") unless the calcifying organisms have evolved mechanisms to prevent dissolution, such as protective layers or other means to isolate their carbonate structures from exposure to corrosive water (IGBP *et al.* 2013, p. 17). Widely varying responses of marine calcifiers to acidification reflect a number of physiological differences amongst these organisms (Ries *et al.* 2009, p. 1131). Species-specific impacts of ocean acidification have been seen in laboratory and field studies on organisms that inhabit areas from the poles to the tropics. Many organisms show adverse effects, such as reduced ability to form and maintain shells and skeletons, as well as reduced survival, growth, abundance, and larval development (IGBP *et al.*

2013, p. 2). Although ocean acidification is not abrupt (i.e., it occurs at a rate proportional to GHG emissions), acidification could represent an abrupt climate impact when thresholds are crossed below which organisms lose the ability to create their shells by calcification, or pH changes affect survival rates (NRC 2013, p. 64).

Mollusks in general are at risk from climate change-induced ocean acidification (Fabry *et al.* 2008, pp. 419–420), and are one of the groups most sensitive to acidification ("high confidence") (IGBP *et al.* 2013, p. 16). Adult and juvenile mollusks have shown reduced calcification, growth, and survival rates from acidification, and in some cases bivalve larvae or juveniles have even shown shell dissolution and mortality (IGBP *et al.* 2013, p. 9; NRC 2010b, pp. 68–69). About 60 percent of mollusks are considered sensitive to negative effects from acidification on current trajectories of GHG emissions, and some species may become locally extinct (IGBP *et al.* 2013, p. 9). Bivalves such as clams and mussels are very sensitive to changes in pH and carbonate chemistry (U.S. Ocean Carbon and Biogeochemistry 2013, p.). Vulnerability to ocean acidification has been shown in bivalve groups favored by red knots, such as mussels (Gaylord *et al.* 2011, p. 2586) and clams (Green *et al.* 2009, p. 1037; Ries *et al.* 2009, entire). For one known red knot prey species, the blue mussel, studies have shown adverse effects of acidification on immune response (Bibby *et al.* 2008, p. 67) and development of the planktonic larvae (Gazeau *et al.* 2010, p. 2051). However, Thomsen *et al.* (2013, p. 1017) found that the blue mussel's benthic stages can tolerate high levels of dissolved carbon dioxide (i.e., higher acidity) when its food supply is abundant, showing that ecological factors (e.g., habitat characteristics, species interactions, energy availability) need to be considered to predict species' vulnerability to ocean acidification (Gazeau *et al.* 2010, p. 2051). Ocean acidification may negatively influence populations of a key red knot prey species in Georgia, the dwarf surf clam (*Mulinia lateralis*), because this species has been found more sensitive to pH than other bivalves (GDNR 2013; Calabrese 1970, pp. 123–125).

Coastal regions that receive freshwater discharge (e.g., certain bays, estuaries, and lagoons that support red knots) may be particularly vulnerable to ocean acidification. Long-term pH decline could exceed the tolerance limits of marine species that live in coastal waters, even though they may have evolved strategies to deal with the short-term pH fluctuations that are typical of coastal environments (U.S. Ocean Carbon and Biogeochemistry 2013, p. 2). Reduced calcification rates and calcium metabolism are also expected to affect several mollusks and crustaceans that inhabit sandy beaches (Defeo *et al.* 2009, p. 8), a primary nonbreeding habitat for red knots.

Relevant to Tierra del Fuego-wintering knots, bivalves have also shown vulnerability to ocean acidification in Antarctic waters, which are predicted to be particularly affected due to naturally low carbonate saturation levels in cold waters (Cummings *et al.* 2011, p. 1). The high-latitude oceans are already becoming corrosive to some species ("medium confidence"). For example, the shells of pteropods, small free-swimming marine snails that are key species in the food web, are already dissolving in parts of the Southern Ocean, which surrounds Antarctica (IGBP *et al.* 2013, p. 16). The high latitudes will be the first ocean regions to become persistently undersaturated with respect to aragonite as a result of human induced acidification (NRC 2010b, p. 77), and conditions detrimental to high-latitude ecosystems could develop within decades (Orr *et al.* 2005, p. 681). If carbon dioxide emissions continue on the current trajectory, 60 percent of the Southern Ocean surface waters, as well as some of the waters off Tierra del Fuego, are expected to become corrosive to aragonite-shelled organisms, such as pteropods and other mollusks, by 2100 (IGBP *et al.* 2013, pp. 6–7).

Even in the temperate waters off the U.S. Pacific coast, Bednaršek *et al.* (2014, p. 1) found large areas are already corrosive to pteropods, with an average of 24 percent of offshore individuals and 53 percent of onshore individuals showing severe dissolution damage. These authors estimate that the incidence of severe pteropod shell dissolution owing to anthropogenic ocean acidification has doubled in near shore habitats since pre-industrial conditions across this region and is on track to triple by 2050 (Bednaršek *et al.* 2014, p. 1). Red knots do not eat pteropods, and natural upwellings result in a different ocean chemistry off the U.S. Pacific coast compared to the Gulf and Atlantic coasts where rufa red knots occur. Nonetheless, this study underscores the vulnerability of mollusks to acidification effects in near shore, temperate waters. Specific to bivalves, acidified waters along coasts of Washington and Vancouver, Canada have been linked to reproductive failure in wild and farmed oysters through impacts to larval shell formation (Waldbusser *et al.* 2013, p. 2171; Grossman 2011).

There are processes associated ocean acidification and ocean warming (discussed in the next section) that can combine to amplify the impact of each factor on ocean biology (Bijma *et al.* 2013, p. 495). For example, warming oceans favor a poleward migration of marine species, whereas ocean acidification would require an equatorward migration as colder waters acidify faster, with these opposing gradients increasing stress on marine communities (Bijma *et al.* 2013, pp. 495-496). In addition, the combination of ocean acidification and increased temperatures negatively affects many organisms (“high confidence”). Ocean acidification appears to narrow the thermal tolerance of some organisms, and others are more vulnerable to ocean acidification in warmer waters. The response to both changes together is often larger than the response to those changes taken separately (IGBP *et al.* 2013, p. 18; Hale *et al.* 2011, entire).

To study the effects of ocean acidification on marine invertebrates, Hale *et al*. (2011, p. 661) collected representative species, including mollusks, from the extreme low intertidal zone and exposed them in the laboratory to varying levels of pH and temperature. These authors found significant changes in community structure and lower diversity in response to reduced pH. At lower pH levels, warmer temperatures resulted in lower species abundances and diversity. The species losses responsible for these changes in community structure and diversity were not randomly distributed across the different phyla examined, with mollusks showing the greatest reduction in abundance and diversity in response to low pH and elevated temperature. This and other studies support the idea that acidification-induced changes in marine biodiversity will be driven by differential vulnerability within and between different taxonomic groups. This study also illustrates the importance of considering indirect effects that occur within multispecies assemblages when attempting to predict the consequences of ocean acidification and global warming on marine communities (Hale *et al*. 2011, p. 661; also see Thomsen *et al.* 2013, entire).

Species’ differing sensitivities and responses to acidification (e.g., Hale *et al*. 2011; Ries *et al.* 2009), combined with other related stressors such as warming, are likely to lead to changes in species composition, reorganizing food webs and thus changing food sources for predators (IGBP *et al.* 2013, p. 19; NRC 2010b, p. 77) like red knots. Indirectly, acidification may affect the productivity and composition of some coastal ecosystems by affecting the key species at the base of coastal food webs (NRC 2010b, p. 69). Acidification impacts on prey species could further exacerbate food web changes caused by other effects of climate change (e.g., range shifts and biological invasions brought on by warming ocean temperatures) (NRC 2010b, p. 78). The potential effects of acidification and warming are further exacerbated by other anthropogenic stresses (e.g., pollution, eutrophication, deoxygenation, overfishing) that have destabilized some ecosystems and limited the potential for adaptation by reducing the populations of many marine species (Bijma *et al.* 2013, p. 495). In some areas, the effects of ocean acidification on coastal ecosystems may be small relative to the effects of natural and human induced stresses. However, in some instances, acidification may act synergistically with other factors (NRC 2010b, p. 70), as multiple stressors compound the effects of ocean acidification ("high confidence") (IGBP *et al.* 2013, p. 2).

We conclude that, with climate change, interactions among ocean pH, ocean temperature, and other factors are likely to cause ecological changes to coastal environments that may affect the red knot's prey species at both wintering and migration stopover areas throughout the nonbreeding range.

Food Availability—Temperature Changes

In addition to being sensitive to acidification, mollusks and other marine invertebrates are sensitive to temperature changes. Global average air temperature is expected to warm at least twice as much in the next century as it has over the previous century, with an expected increase of 2 to 11.5 ºF (1.1 to 6.4 ºC) by 2100 (USEPA 2012). Coastal waters are "very likely" to continue to warm by as much as 4 to 8 ºF (2.2 to 4.4 ºC) in this century, both in summer and winter (USGCRP 2009, p. 151). In the mid-Atlantic, changes in water temperature (and quality) are expected to have mostly indirect effects on red knots and other shorebirds, primarily through changes in the distribution and abundance of food resources (Najjar *et al.* 2000, p. 227). Changes in sea temperatures can have major effects on marine populations, as witnessed during severe events such as El Niño (an occasional abnormal warming of tropical waters in the eastern Pacific from unknown causes), when the abundance of many invertebrate species plummeted on South American beaches (Rehfisch and Crick 2003, p. 88). Although the invertebrates recovered quickly when conditions returned to normal, this short-term change in sea temperature may give an indication of likely changes under projected global warming scenarios (Rehfisch and Crick 2003, p. 88).

Both salinity (discussed below) and temperature levels affect survival of a key red knot prey species in Georgia, the dwarf surf clam, especially the larval stages (Calabrese 1969, pp. 420–425). Based on this species' temperature sensitivities (Kennedy *et al.* 1974, p. 137), warming oceans may influence recruitment and survival of these clams in the future (GDNR 2013). Episodic massive die-offs of these clams have been documented, though the exact causes are not well understood (Cleveland *et al.* 2002, entire). The GDNR (2013) is concerned that a number of predicted changes associated with global climate change (e.g., ocean acidification, discussed above; warming) may negatively affect the most important prey item for red knots migrating through Georgia.

Asynchronies (“mismatches”) between the timing of the red knot’s annual cycle and the peak abundance periods of its prey are discussed in the next section. However, repeated asynchronies can also occur between a prey species’ own annual cycles and environmental conditions, leading to long-term declines of these invertebrate populations and thereby affecting the absolute quantity of red knot food supplies (in addition to the timing). For example, Philippart *et al.* (2003, p. 2171) found that rising water temperatures upset the timing of reproduction in the intertidal bivalve *Macoma balthica*, with the timing of the first vulnerable life stages thrown out of sync with respect to the most optimal environmental conditions (a phytoplankton bloom and the settlement of juvenile shrimps). These authors concluded that prolonged periods of lowered bivalve recruitment and stocks may lead to a reformulation of estuarine food webs and possibly a reduction of the resilience of the system to additional disturbances, such as shellfish harvest (Philippart *et al.* 2003, p. 2171).

Range shifts, typically poleward, are already underway among marine species, although considerable variation in species’ responses exists. Range shifts have been reported from 0.9 to 17.4 mi (1.4 to 28 km) per decade for marine communities (Burrows *et al.* 2011, p. 652). At the leading edges of marine species’ ranges, where the most rapid rates of migration occur, Poloczanska *et al.* (2013, p. 1) found a mean rate of expansion of 44.7 ± 8.4 mi (72.0 ± 13.5 km) per decade, about an order of magnitude faster than rates reported for predominately terrestrial species. In a literature review, Sorte *et al.* (2010, p. 303) identified 129 marine species experiencing range shifts. These authors found that 75 percent of the shifts were in the poleward direction, consistent with climate change scenarios, and that shifting species spread over an order of magnitude faster in marine than in terrestrial systems. Although data were limited, the effects of these range shifters on the recipient communities were largely negative and of similar magnitude as from species introduced by human activity. Just like introductions, range shifts have the potential to seriously affect biological systems. Among the species with documented range shifts were several bivalves and numerous gastropods (e.g., snails) (Sorte *et al.* 2010, p. 308), taxa that include important red knot prey species (see Migration and Wintering Food). Specific to benthic mollusks, Poloczanska *et al.* (2013, p. 4) found an average leading edge poleward shift of about 9.3 mi (15 km) per decade (sample size of 46). For some coastal species, a poleward shift in distribution may be limited by geography where organisms simply ‘‘run out’’ of coastline to migrate along and are faced with a major oceanic barrier to dispersal (Bijma *et al.* 2013, p. 496), or along east-west oriented shorelines (Burrows *et al.* 2011, p. 654). Thus, subregional and habitat-specific patterns in temperature may run counter to the simple expectation of poleward range shift (Burrows *et al.* 2011, p. 654; Richardson *et al.* 2012, p. 909). Looking at 360 marine taxa from 1968 to 2011, Pinsky *et al.* (2013, p. 1239) found that climate velocity (i.e., the rate and direction that climate shifts across the landscape) explained the magnitude and direction of shifts in latitude and depth much more effectively than did species characteristics. Range shifts in key prey species, or colonization of key nonbreeding areas by new marine species, would likely alter food availability for the red knot.

For example, blue mussel spat is an important prey item for red knots in Virginia (Karpanty *et al.* 2012, p. 1). The southern limit of adult blue mussels has contracted from North Carolina to Delaware since 1960 due to increasing air and water temperatures (Jones *et al.* 2010, pp. 2255–2256). Larvae have continued to recruit to southern locales (including Virginia) via currents, but those recruits die early in the summer due to water and air temperatures in excess of lethal physiological limits. Failure to recolonize southern regions will occur when reproducing populations at higher latitudes are beyond dispersal distance (Jones *et al.* 2010, pp. 2255–2256). Thus, this key prey resource may soon disappear from the red knot's Virginia spring stopover habitats (Karpanty *et al.* 2012, p. 1).

Food Availability—Other Aspects of Climate Change

Invertebrate prey species may also be affected by other physical aspects of climate change. In estuaries and lagoons, important red knot habitats, sea level rise and changing freshwater inputs will affect sediment redistribution, tidal prisms (the volume of water in an estuary between high and low tide), submergence periods, and salinity regimes, which could significantly alter the composition of estuarine communities. Primary production and biodiversity may be impacted by changes in water discharge and precipitation-evaporation balance. Altered riverine discharge and warming may lead to enhanced thermal stratification, salinity stratification, or both within estuaries and lagoons. Changes in precipitation extremes and freshwater supply may induce fluctuations in salinity with associated adverse impacts on biodiversity and ecosystem functions. Changes in storm events may also alter the sediment deposition-erosion balance of lagoons and estuaries. Finally, rising sea levels are expected to affect the physical shape (e.g., dimensions, configuration) of estuaries, changing their sediment compositions. In turn, these habitat changes would change invertebrate densities and community composition, thus affecting shorebirds (Chapter 5 in IPCC 2014, p. 21; Rehfisch and Crick 2003, p. 88; Najjar *et al.* 2000, p. 225), such as the red knot.

During fall migration in the 1980s, up to 10,000 red knots, primarily southern-wintering birds, used stopover habitats on western Cape Cod Bay in Massachusetts, where their foraging was concentrated primarily on mussel spat growing on peat banks, rather than spat growing on rocks. Since about 1990, these sites are no longer used by red knots, evidently because of a reduction in food resources brought about when oceanfront peat banks eroded and no longer offered a substrate from which knots could easily and rapidly remove mussel spat (B. Harrington pers. comm. November 14, 2013). We do not know the cause of this erosion, but coastal erosion along western Cape Cod Bay is well known, and is expected to increase to due accelerating rates of sea level rise (Carini 2014; Kleinfelder 2013, entire). In this and other red knot areas, similar effects to prey resources, mediated by habitat changes, may become more frequent as the rates of sea level rise increase. Under Factor A—Sea Level Rise, we concluded that, if shorelines experience a decades-long period of high instability and landward migration, the formation rate of new intertidal habitats may be slower than the inundation rate of existing habitats. This conclusion is supported by the prolonged duration (about 1990 to present) of prey and habitat loss on western Cape Cod Bay, brought about by coastal erosion. However, it is possible that declining red knot use of this area may have also been influenced by the advent of commercial mussel harvest (B. Harrington pers. comm. November 14, 2013), overall declines in the population size of southern-wintering red knots (Harrington *et al.* 2010b, p. 361), or both.

Food Availability—Disease, Parasites, Invasive Species, and Unknown Factors

Red knot prey species are also vulnerable to disease, parasites, invasive species, and unknown factors influencing their quality and quantity. For example, at the single largest wintering area, Bahía Lomas on Tierra del Fuego in Chile, Espoz *et al.* (2008, pp. 69, 74) found that most (91 percent) of the prey (the clam *Darina solenoides*) were much smaller and, therefore, probably less energetically profitable than the size classes of bivalves shown to be preferred by knots in many other locations. These authors suggest that food supply at Bahía Lomas may be a limiting factor for the knot population and might have contributed to population declines in the 2000s. However, no reasons for the small prey size are known (Espoz *et al.* 2008, p. 75), and it is unknown whether prey size in this area has decreased over time.

In Río Grande, Argentina, a key Tierra del Fuego wintering area, Escudero *et al*. (2012) sampled the area's two main red knot prey types (Mytilidae mussels and the clam *Darina solenoides*) in 1995, 2000, and 2008. Over the study period, significant decreases occurred in the sizes of available prey items and in the red knots' energy intake rates. Intake rates went from the highest known for red knots anywhere in the world in 2000 to among the lowest in 2008 (Escudero *et al.* 2012, pp. 359–362). These authors also found a substantial increase in the rate of red knots utilizing alternate prey species, and their findings imply that the birds incorporated other prey types into their diets to increase intake rates (Escudero *et al.* 2012, pp. 359, 362). No explanation is available for the decline in prey sizes. Escudero *et al.* (2012, p. 363) noted a high prevalence of a digenean parasite (*Bartolius pierrei*) on *D. solenoides* clams. These authors do not implicate the parasite in the declining sizes of available clams. The mussels, which were not subject to any noteworthy parasitism, also exhibited decreased sizes over the study period (Escudero *et al.* 2012, p. 359), suggesting that parasitism is not a likely explanation for declining sizes. However, disease and parasites of the red knots' mollusk prey may increase with climate change, with potential effects on both prey availability and the health of the birds exposed to these pathogens. Increases in mollusk diseases, apparently temperature-related, were detected in a review of scientific literature published from 1970 to 2001 (Ward and Lafferty 2004, p. 543).

Globally, coastal marine habitats are among the most heavily invaded systems, stemming in part from human-mediated transport of nonnative species in the ballast of ships and from intentional introductions for aquaculture and fisheries enhancement (Seebens *et al.* 2013, p. 782; Grosholz 2002, p. 22). For example, introduction of nonnative oysters (*Crassostrea* spp.) has been widespread within the range of the red knot (Ruesink *et al.* 2005, p. C-1). Worldwide, introduced oysters have been vectors for several invasive species of marine algae, invertebrates, and protozoa (Ruesink *et al.* 2005, pp. 669–670). Invasive species can cause disease in native mollusks, displace native invertebrates through competition or predation, alter ecosystems, and affect species at higher trophic levels such as shorebirds (Ruesink *et al.* 2005, pp. 671–674; Grosholz 2002, p. 23). Climate change can facilitate marine invasions, for example through changes in salinity and freshwater inputs (discussed above) in coastal waters (Winder *et al.* 2011, entire).

Food Availability—Sediment Placement

The quantity and quality of red knot prey may also be affected by the placement of sediment for beach nourishment or disposal of dredged material (see Factor A above for a discussion of the extent of these practices in the United States and their effects on red knot habitat). Invertebrates may be crushed or buried during project construction. Although some benthic species can burrow through a thin layer of additional sediment, thicker layers (over 35 in (90 cm)) smother the benthic fauna (Greene 2002, p. 24). By means of this vertical burrowing, recolonization from adjacent areas, or both, the benthic faunal communities typically recover.

Recovery can take as little as 2 weeks or as long as 2 years, but usually averages 2 to 7 months (Greene 2002, p. 25; Peterson and Manning 2001, p. 1). Although many studies have concluded that invertebrate communities recovered following sand placement, study methods have often been insufficient to detect even large changes (e.g., in abundance or species composition), due to high natural variability and small sample sizes (Peterson and Bishop 2005, p. 893). Therefore, uncertainty remains about the effects of sand placement on invertebrate communities, and how these impacts may affect red knots.

The invertebrate community structure and size class distribution following sediment placement may differ considerably from the original community (Zajac and Whitlatch 2003, p. 101; Peterson and Manning 2001, p. 1; Hurme and Pullen 1988, p. 127). Recovery may be slow or incomplete if placed sediments are a poor grain size match to the native beach substrate (Bricker 2012, pp. 31–33; Peterson *et al.* 2006, p. 219; Greene 2002, pp. 23–25; Peterson *et al.* 2000, p. 368; Hurme and Pullen 1988, p. 129), or if placement occurs during a seasonal low point in invertebrate abundance (Burlas 2001, p. 2-20). Recovery is also affected by the beach position and thickness of the deposited material (Schlacher *et al.* 2012, p. 411). If the profile of the nourished beach and the imported sediments do not match the original conditions, recovery of the benthos is unlikely (Defeo *et al.* 2009, p. 4). Reduced prey quantity and accessibility caused by a poor sediment size match have been shown to affect shorebirds, causing temporary but large (70 to 90 percent) declines in local shorebird abundance (Peterson *et al.* 2006, pp. 205, 219).

Beach nourishment is a regular practice in Delaware Bay and can affect spawning habitat for horseshoe crabs. Although beach nourishment generally preserves horseshoe habitat better than hard stabilization structures, nourishment can enhance, maintain, or decrease habitat value depending on beach geometry and sediment matrix (Smith *et al.* 2002a, p. 5). In a field study in 2001 and 2002, Smith *et al.* (2002a, p. 45) found a stable or increasing amount of spawning activity at beaches that were recently nourished while spawning activity at control beaches declined. These authors also found that beach characteristics affect horseshoe crab egg development and viability. Beach nourishment can alter both the beach foreshore (sediment size distribution, slope, and width) and low tide terrace (sediment size distribution, elevation, and width) (Smith *et al.* 2002b, p. 739). Avissar (2006, p. 427) modeled nourished versus control beaches and found that nourishment may compromise egg development and viability. Although nourishment is generally considered to be environmentally compatible, the effect of nourishment on horseshoe crab spawning, egg development, and survival of juveniles is understudied (Smith *et al.* 2002b, p. 739). Evaluating the impacts of beach nourishment projects on horseshoe crab populations has been identified as a high research priority by ASMFC (2013a, p. 31). Despite possible drawbacks, beach nourishment is often successfully used to restore and maintain horseshoe crab spawning habitat on both sides of Delaware Bay; see Factor A—Shoreline Stabilization—Sediment Transport—Beach Nourishment.

Food Availability—Recreational Activities

Recreational activities can likewise affect the availability of shorebird food resources by causing direct mortality of prey. Studies from the United States and other parts of the world have documented recreational impacts to beach invertebrates, primarily from the use of off-road vehicles (ORVs), but even heavy pedestrian traffic can have effects. Few studies have examined the potential link between these invertebrate impacts and shorebirds. However, several studies on the effects of recreation on invertebrates are considered the best available information, as they involve species and habitats similar to those preyed on by red knots.

Although pedestrians exert relatively low ground pressures, extremely heavy foot traffic can cause direct crushing of intertidal invertebrates. In South Africa, Moffett *et al.* (1998, p. 87) found the clam *Donax serra* was slightly affected at all trampling intensities, while *D. sordidus* and the isopod *Eurydice longicornis* were affected only at high trampling intensities. Few members of the macrofauna were damaged at low trampling intensities, but substantial damage occurred under intense trampling (Moffett *et al.* 1998, p. 87). At beach access points in Australia, Schlacher and Thompson (2012, pp. 123–124) found trampling impacts to benthic invertebrates on the lower part of the beach, including significant reductions in total abundance and species richness and a shift in community structure. Studies have found that macrobenthic populations and communities respond negatively to increased human activity, but not in all cases. In addition, it can be difficult to separate the effect of human trampling from habitat modifications because these often coincide in high-use areas. In general, evidence is sparse about how sensitive intertidal invertebrates might be to human trampling (Defeo *et al.* 2009, p. 3). We are not aware of any studies looking at potential links between trampling and shorebird prey availability, but red knots often occur in areas with high recreational use (see Human Disturbance, below).

In many areas, habitat for the piping plover overlaps considerably with red knot habitats. A preliminary review of ORV use at piping plover wintering locations (from North Carolina to Texas) suggests that ORV impacts may be most widespread in North Carolina and Texas (USFWS 2009, p. 46). Although red knots normally feed low on the beach, they may also utilize the wrack line (see "Migration and Wintering Habitat" under Species Information, and Factor A—Beach Cleaning). Kluft and Ginsberg (2009, p. vi) found that ORVs killed and displaced wrack invertebrates and lowered the total amount of wrack, in turn lowering the overall abundance of wrack dwellers. In the intertidal zone, invertebrate abundance is greatest in the top 12 in (30 cm) of sediment (Carley *et al.* 2010, p. 9). Intertidal fauna are burrowing organisms, typically 2 to 4 in (5 to 10 cm) deep; burrowing may ameliorate direct crushing. However, shear stress of ORVs can penetrate up to 12 in (30 cm) into the sand (Schlacher and Thompson 2007, p. 580).

Some early studies found minimal impacts to intertidal beach invertebrates from ORV use (Steinback and Ginsberg 2009, pp. 4–6; Van der Merwe and Van der Merwe 1991, p. 211; Wolcott and Wolcott 1984, p. 225). However, some attempts to determine whether ORVs had an impact on intertidal fauna have been unsuccessful because the naturally high variability of these invertebrate communities masked any effects of vehicle damage (Stephenson 1999, p. 16). Based on a review of the literature through 1999, Stephenson (1999, p. 33) concluded that vehicle impacts on the biota of the foreshore (intertidal zone) of sandy beaches have appeared to be minimal, at least when the vehicle use occurred during the day when studies typically take place, but very few elements of the foreshore biota had been examined.

Other studies have found higher impacts to benthic invertebrates from driving (Sheppard *et al.* 2009, p. 113; Schlacher *et al.* 2008b, pp. 345, 348; Schlacher *et al.* 2008c, pp. 878, 882;

Wheeler 1979, p. iii), although it can be difficult to discern results specific to the wet sand zone where red knots typically forage. Due to the compactness of sediments low on the beach profile, driving in this zone is thought to minimize impacts to the overall beach invertebrate community. However, the relative vulnerability of species in this zone is not well known, and driving low on the beach may expose a larger proportion of the total intertidal fauna to vehicles (Schlacher and Thompson 2007, p. 581). The severity of direct impacts (e.g., crushing) depends on the compactness of the sand, the sensitivity of individual species, and the depth at which they are buried in the sand (Schlacher *et al.* 2008b, p. 348; Schlacher *et al*. 2008c, p. 886). At least one study documented a positive response of shorebird populations following the exclusion of ORVs (Defeo *et al.* 2009, p. 3; Williams *et al.* 2004, p. 79), although the response could have been due to decreased disturbance (discussed below) as well as (or instead of) increased prey availability following the closure.

In summary, several studies have shown impacts from recreational activities on invertebrate species typical of those used by red knots, and in similar habitats. The extent to which mortality of beach invertebrates from recreational activities propagates through food webs is unresolved (Defeo *et al.* 2009, p. 3). However, we conclude that these activities likely cause at least localized reductions in red knot prey availability.

Food Availability—Horseshoe Crab Harvest

Reduced food availability at the Delaware Bay stopover site due to commercial harvest and subsequent population decline of the horseshoe crab is considered a primary causal factor in the decline of the *rufa* subspecies in the 2000s (Escudero *et al.* 2012, p. 362; McGowan *et al.* 2011a, pp. 12–14; CAFF 2010, p. 3; Niles *et al.* 2008, pp. 1–2; COSEWIC 2007, p. vi; González *et al*. 2006, p. 114; Baker *et al.* 2004, p. 875; Morrison *et al.* 2004, p. 67), although other possible causes or contributing factors have been postulated (DDNREC 2013; Fraser *et al.* 2013, p. 13; Schwarzer *et al.* 2012, pp. 725, 730–731; Escudero *et al.* 2012, p. 362; Espoz *et al.* 2008, p. 74; Niles *et al.* 2008, p. 101; also see Asynchronies, below). Due to harvest restrictions and other conservation actions, horseshoe crab populations showed some signs of recovery in the early 2000s, with apparent signs of red knot stabilization (survey counts, rates of weight gain) occurring a few years later (as might be expected due to biological lag times). Since about 2005, however, horseshoe crab population growth has stagnated for unknown reasons.

Under the current management framework (i.e., the ARM), the present horseshoe crab harvest is not considered a threat to the red knot because harvest levels are tied to red knot populations via scientific modeling. Most data suggest that the volume of horseshoe crab eggs is currently sufficient to support the Delaware Bay's stopover population of red knots at its present size. However, because of the uncertain trajectory of horseshoe crab population growth, it is not yet known if the egg resource will continue to adequately support red knot population growth over the next decade. In addition, implementation of the ARM could be impeded by insufficient funding for the shorebird and horseshoe crab monitoring programs that are necessary for the functioning of the ARM models.

Many studies have established that red knots stopping over in Delaware Bay during spring migration achieve remarkable and important weight gains to complete their migrations to the breeding grounds by feeding almost exclusively on a superabundance of horseshoe crab eggs (see Wintering and Migration Food). A temporal correlation occurred between increased horseshoe crab harvests in the 1990s and declining red knot counts in both Delaware Bay and Tierra del Fuego by the 2000s. Other shorebird species that rely on Delaware Bay also declined over this period (Mizrahi and Peters in Tanacredi *et al.* 2009, p. 78), although some shorebird declines began before the peak expansion of the horseshoe crab fishery (Botton *et al*. in Shuster *et al*. 2003, p. 24).

The causal chain from horseshoe crab harvest to red knot populations has several links, each with different lines of supporting evidence and various levels of uncertainty: (a) horseshoe crab harvest levels and Delaware Bay horseshoe crab populations (Link A); (b) horseshoe crab populations and red knot weight gain during the spring stopover (Link B); and (c) red knot weight gain and subsequent rates of survival, reproduction, or both (Link C). The evidence supporting each of these linkages is weighed below. Despite the various levels of uncertainty, the weight of evidence supports these linkages, points to past harvest as a key factor in the decline of the red knot, and underscores the importance of continued horseshoe crab management to meet the needs of the red knot.

Horseshoe Crab—Harvest and Population-Levels (Link A)

Historically, horseshoe crabs were harvested commercially for fertilizer and livestock feed. From the mid-1800s to the mid-1900s, harvest ranged from about 1 to 5 million crabs annually. Harvest numbers dropped to 250,000 to 500,000 crabs annually in the 1950s, which are considered the low point of horseshoe crab abundance. Only about 42,000 crabs were reported annually by the early 1960s. Early harvest records should be viewed with caution due to probable underreporting. The substantial commercial-scale harvesting of horseshoe crabs ceased in the 1960s (ASMFC 2013b, p. 7; ASMFC 2009a, p. 1). By 1977, the spawning population of horseshoe crabs in Delaware Bay was several times larger than during the 1960s, but was far from approaching the numbers and spawning intensity reported in the late 1800s (Shuster and Botton 1985, p. 363). No information is available on how these historical harvests of horseshoe crabs may have affected populations of red knots or other migratory shorebirds, but these historical harvests occurred at a time when shorebird numbers had also been markedly reduced by hunting (Botton *et al*. in Shuster *et al.* 2003, pp. 25–26; Dunne in NJAS 2007, p. 25); see Historical Distribution and Abundance and Factor B, above.

During the 1990s, reported commercial harvest of horseshoe crabs on the Atlantic coast of the United States increased dramatically (table 22). Modern harvests are for bait and the biomedical industry. Commercial fisheries for horseshoe crab consist primarily of directed trawls and hand harvest (e.g., collection from beaches during spawning) (ASMFC 2013b, p. 12; ASMFC 2009a, p. 14). Horseshoe crabs are used as bait in the American eel (*Anguilla rostrata*), conch (whelk) (*Busycon* spp.), and other fisheries. The American eel pot fishery prefers egg-laden female horseshoe crabs, while the conch pot fishery uses both male and female horseshoe crabs (ASMFC 2013b, p. 7; ASMFC 2009a, p. 1). The increase in harvest of horseshoe crabs during the 1990s was largely due to increased use as conch bait (ASMFC 2009a, p. 1).

Although also used in scientific research and for other medical purposes, the major biomedical use of horseshoe crabs is in the production of Limulus Amebocyte Lysate (LAL). The LAL is a clotting agent in horseshoe crab blood that makes it possible to detect human pathogens in patients, drugs, and intravenous devices (ASMFC 2013b, p. 8; ASMFC 2009a, p. 2). The "LAL test" is currently the worldwide standard for screening medical equipment and injectable drugs for bacterial contamination (ASMFC 2013b, p. 8; ASMFC 2009a, p. 2; ASMFC 1998, p. 12). Horseshoe crab blood is obtained from adult crabs that are released alive after extraction is complete or that are sold into the bait market (ASMFC 2013b, p. 8; ASMFC 2009a, pp. 2, 18). The ASMFC assumes a constant 15 percent mortality rate for bled crabs that are not turned over to the bait fishery (M. Hawk pers. comm. May 28, 2014; ASMFC 2013b, p. 9; ASMFC 2009a, p. 3) (e.g., when considering if the threshold has been exceeded, as discussed below). Starting in 2011, however, the ASMFC's Horseshoe Crab Technical Committee began recommending use of a range from 5 to 30 percent mortality to include known variances in conditions and situations that can occur over the geographical and temporal range of collecting and bleeding the horseshoe crabs (ASMFC 2013c, p. 8; ASMFC 2012a, p. 6). The estimated mortality rate includes all crabs rejected for biomedical use any time between capture and release.

Bait harvest and biomedical collection have been managed separately by the ASMFC since 1999 (ASMFC 1998, pp. iii–57). Biomedical collection is currently not capped, but ASMFC can consider implementing action to reduce mortality if estimated mortality exceeds a threshold of 57,500 crabs. The threshold has been exceeded every year since 2007 with biomedical mortality averaging 70,600 crabs, but thus far the ASMFC has opted only to issue voluntary best practices guidelines to the biomedical industry (ASMFC 2013b, p. 17; ASMFC 2009a, p. 18). The ASMFC implemented key reductions in the bait harvest in 2000, 2004, and 2006 (ASMFC 2013b, p. 9; ASMFC 2009a, p. 3), and harvest of female crabs has been prohibited in the Delaware Bay Region (New Jersey, Delaware, and parts of Maryland and Virginia) since 2006 (ASMFC 2013d). Several member States have voluntarily restricted harvests below their allotted quotas (ASMFC 2013c, p. 3; ASMFC 2012a, pp. 4; 13 N.J.S.A. 23:2B-21; N.J.R. 2139(a)). Along with the widespread use of bait-saving devices, these restrictions reduced reported landings (ASMFC 2009a, p. 1) from 1998 to 2012 by over 75 percent (table 23). In addition, the National Marine Fisheries Service (NMFS) established the Carl N. Shuster Jr. Horseshoe Crab Reserve in 2001, as recommended by the ASMFC. About 30 nautical miles (55.6 km) in radius and located in Federal waters off the mouth of the Delaware Bay, the reserve is closed to commercial horseshoe crab harvest except for limited biomedical collection authorized periodically by NMFS (NOAA 2001, pp. 8906–8911).

Evidence that commercial harvests caused horseshoe crab population declines in recent decades comes primarily from a strong temporal correlation between harvest levels (as measured by reported landings, tables 22 and 23) and population levels (as characterized by ASMFC during stock assessments).

Link A, Part 1—Horseshoe Crab Harvest Levels

The horseshoe crab landings given in pounds in tables 22 and 23 come from data reported to NMFS, but should be viewed with caution as these records are often incomplete and represent an underestimate of actual harvest (ASMFC 1998, p. 6). In addition, reporting has increased over the years, and the conversion factors used to convert crab numbers to pounds have varied widely. Despite these inaccuracies, the reported landings show that commercial harvest of horseshoe crabs increased substantially from 1990 to 1998 and has generally declined since then

(ASMFC 2013b, p. 8; ASMFC 2009a, p. 2). The ASMFC (1998, p. 6) also considered other data sources to corroborate a significant increase in harvest in the 1990s. These landings (pounds) may include biomedical collection, live trade, and bait fishery harvests (ASMFC 2013b, p. 16; ASMFC 2009a, p. 17).

Table 23 also shows the number of crabs harvested for bait, and the estimated number of crabs killed incidental to biomedical collection, as reported to ASMFC. Since 1998, States have been required to report annual bait landings to ASMFC, which considers these data reliable (ASMFC 2013b, p. 8; ASMFC 2009a, p. 2). A subtotal of the bait harvest is shown for the Delaware Bay Region (New Jersey, Delaware, and a part of the harvests in Maryland and Virginia), as managed by ASMFC. The numbers given in tables 22 and 23 do not reflect the changing sex ratio of crabs harvested in the Delaware Bay Region (S. Michels pers. comm. February 15, 2013); harvest of females has been prohibited since 2006 (ASMFC 2013d). In 2013 and 2014, the first years that the harvest level was determined using the ARM, the quota in the Delaware Bay Region was set at 500,000 males and 0 females (ASMFC 2013e, p. 2; ASMFC 2012b, p. 1); however, we do not yet have access to the actual number of crabs removed during these 2 years to compare against the quota. Since 2006, all four States in the Delaware Bay Region have frequently harvested fewer crabs than allowed by the ASMFC (ASMFC 2013c, p. 4; ASMFC 2012a, p. 13). Since 2006, New Jersey has opted not to use its quota by imposing a bait harvest moratorium, which the State has considered lifting amid considerable controversy between environmental and fishing groups (Augenstein 2013, entire; ASMFC 2013c, p. 17; ASMFC 2012a, p. 13; N.J.S.A. 23:2B-21; N.J.R. 2139(a)).

Estimates of biomedical collection increased from 130,000 crabs in 1989 to 260,000 in 1997 (ASMFC 2004, p. 12). Since mandatory reporting requirements took effect in 2004, biomedical-only crabs collected (i.e., crabs not counted against State bait harvest quotas) rose from 292,760 in 2004 (ASMFC 2009a, pp. 18, 41) to 545,164 in 2011 (ASMFC 2012a, p. 6), but fell slightly to 530,797 in 2012 (ASMFC 2013c, p. 8). Total estimated mortality of biomedical crabs for 2012 was 79,786 crabs (using a 15 percent post-release estimated mortality; see table 23), with a range of 31,189 to 152,681 crabs (using 5 to 30 percent estimated mortality) (ASMFC 2013c, p. 8). Using a constant 15 percent mortality of bled crabs, the estimated contribution of biomedical collection to total (biomedical plus bait) mortality rose from about 6 percent in 2004 to about 11 percent in 2011, but dropped to about 10 percent in 2012 (table 23) (ASMFC 2013b, p. 17). There has been uneven growth in the number of horseshoe crabs are being biomedically bled first before being used as bait; because such crabs count against harvest quotas (ASMFC 2013c, p. 8; ASMFC 2012a, p. 6), this practice helps reduce total mortality rates. However, the proportion of bled crabs counting toward the bait quotas decreased to 13 percent in 2013 from 22 percent in 2009, and 15 percent in 2004 (ASMFC 2013b, p. 17).

To put the reported harvest numbers in context, Smith *et al.* (2006, p. 461) used modeling of marked horseshoe crabs in 2003 to estimate the population in the Delaware Bay Region at about 20 million adults (90 percent confidence interval 13 to 28 million), of which 6.25 million were females. Subsequently, Smith (2013) used a different methodology, based on a trawl survey designed for this purpose, to estimate the size of the Delaware Bay horseshoe crab population. Smith (2013, p. 2) reported annual estimates of the baywide population size from 2002 to 2012, ranging from about 9.3 (2004) to about 32.2 million (2007) adult crabs, with an average of about 19 million and consistently more males than females (sex ratios of 1.65 to 3.12). The most recent (2012) estimate, 25.9 million adult crabs, was based on only a partial survey due to limited funding (Smith 2013, p. 2). Nonetheless, recent annual harvests of roughly 200,000 horseshoe crabs from the Delaware Bay Region represent about 1 percent of the long-term average adult population size of about 19 million.

Table 22. Reported Atlantic coast horseshoe crab landings (pounds), 1969 to 2012 (NMFS 2014)

Year	Total pounds reported to NMFS	Year	Total pounds reported to NMFS
1969	10,600	1991	385,487
1970	15,900	1992	321,995
1971	11,900	1993	821,205
1972	42,000	1994	1,171,571
1973	88,700	1995	2,416,168
1974	16,700	1996	5,159,326
1975	62,800	1997	5,983,033
1976	2,043,100	1998	6,835,305
1977	473,000	1999	5,542,506
1978	728,500	2000	3,756,475
1979	1,215,630	2001	2,336,645
1980	566,447	2002	2,772,010
1981	326,695	2003	2,624,248
1982	526,700	2004	974,425
1983	468,600	2005	1,421,957
1984	225,112	2006	1,350,094
1985	614,939	2007	1,804,968
1986	635,823	2008	1,315,135
1987	511,758	2009	1,819,646
1988	688,839	2010	1,197,883
1989	1,106,645	2011	1,508,615
1990	519,057	2012	1,703,062

Table 23. Reported Atlantic coast horseshoe crab landings (pounds and crabs), 1998 to 2012 (NMFS 2014; A. Nelson pers. comm. February 22, 2013 and November 27, 2012; ASMFC 2013c, pp. 4, 8; ASMFC 2009a, pp. 38–41); ND = no data available.

Year	Total pounds reported to NMFS (from Table 22)*	Numbers of crabs harvested for bait reported to ASMFC	Numbers of crabs harvested for bait reported to ASMFC, Delaware Bay Region subtotal	Estimated numbers of crabs killed by biomedical collection, based on 15 percent of the total biomedical collection reported to ASMFC
1998	6,835,305	2,748,585	862,462	ND
1999	5,542,506	2,600,914	1,013,996	ND
2000	3,756,475	1,903,415	767,988	ND
2001	2,336,645	1,013,697	607,602	ND
2002	2,772,010	1,265,925	728,266	ND
2003	2,624,248	1,052,493	584,394	ND
2004	974,425	681,323	278,280	45,670
2005	1,421,957	769,429	347,927	44,830
2006	1,350,094	840,944	270,241	49,182
2007	1,804,968	827,554	169,255	63,432
2008	1,315,135	660,983	190,828	63,285
2009	1,819,646	817,265	250,699	60,642
2010	1,197,883	605,511	165,852	75,428
2011	1,508,615	662,622	195,153	80,827
2012	1,703,062	729,100	201,840	79,786

*These data from NMFS (in pounds) are given only for context, excerpted from the longer time series presented in table 22. Because of inconsistent conversion factors, these pounds data cannot be converted to numbers of crabs, and thus cannot be directly compared to the data reported to ASMFC.

Link A, Part 2—Horseshoe Crab Population Levels

Through stock assessments, ASMFC weighs and analyzes horseshoe crab data from many different independent surveys and models (ASMFC 2004, pp. 14–24; ASMFC 2009a, pp. 14–23). In the 2004 assessment, ASMFC found a clear preponderance of evidence that horseshoe crab populations in the Delaware Bay Region declined from the late 1980s to 2003, and that declines early in this evaluation period were steeper than later declines (ASMFC 2004, p. 27). Genetic analysis also suggested that the Delaware Bay horseshoe crab population was exhibiting the effects of a recent population bottleneck in the mid-1990s (Pierce *et al.* 2000, pp. 690, 691, 697), and modeling confirmed that overharvest caused declines (Smith *et al.* in Tanacredi *et al.* 2009, p. 361). In the 2009 stock assessment, ASMFC concluded that there was no evidence of ongoing declines in the Delaware Bay Region, and that the demographic pattern of significant increases matched the expectations for a recovering population (ASMFC 2009a, p. 23). The 2009 stock assessment reflects substantial reductions in harvest levels, from their peak at 2 to 3 times maximum sustained yield in 1998 and 1999 to 23.2 percent of maximum sustained yield in 2008 (ASMFC 2009a, pp. 25, 57). These findings support the temporal correlation that rising harvest levels led to population declines through the 1990s, while management actions had started reversing the decline by the mid-2000s. The 2013 stock assessment update concluded that, in the Delaware Bay Region, there is evidence of increases in certain age or sex classes, but overall population trends have been largely stable (neither increasing nor decreasing) since the previous stock assessment in 2009 (ASMFC 2013b, p. 22).

Mortality from the biomedical harvest has not been factored into stock assessments to date (ASMFC 2013b, p. 18), as recommended by the Delaware Bay Ecosystem Technical Committee (ASMFC 2013e, p. 3). The ASMFC intends to include biomedical mortality in the next benchmark (peer reviewed) stock assessment (ASMFC 2013b, p. 18). However, confidentiality issues surrounding biomedical data must be resolved before they can be included in an assessment. The next stock assessment, scheduled for 2016, will be an update rather than a benchmark, and thus may not consider new data sources such as biomedical data (M. Hawk pers. comm. September 19, 2014).

In 2012, the ASMFC's Delaware Bay Ecosystem Technical Committee reviewed updated data from the same trawl and dredge surveys that were evaluated in the 2004 and 2009 assessments. From these data, the committee concluded that declines were observed during the 1990s, stabilization occurred in the early 2000s, various indicators have differed with no consistent trends since 2005, confidence intervals are large, there is no clear trend apparent in recent data, and the population has at least stabilized (ASMFC 2012c, pp. 10–12). These conclusions continue to hold with the 2013 stock assessment (ASMFC 2013b, p. 22), and generally support the link between harvest levels and available indicators of horseshoe crab abundance. The committee noted, however, that sustained horseshoe crab population increases have not been realized as expected. The reasons for this stagnation are unknown, and a recent change in sex ratios is also unexplained (i.e., several surveys found that the ratio of males to females increased sharply since 2010 despite a prohibition on harvesting females since 2006) (ASMFC 2013d; S. Michels pers. comm. February 15, 2013; ASMFC 2012d, pp. 17–18; ASMFC 2010, pp. 2–3). The committee speculated that some combination of the following factors may explain the lack of recent population growth, but committee members did not reach consensus regarding which factors are more likely (ASMFC 2012c, p. 12; ASMFC 2012d, p. 2).

- Insufficient time since management actions were taken. There would likely be at least a 10-year time lag between fishery restrictions and significant population changes, corresponding to the horseshoe crab's estimated age at sexual maturity (Sweka *et al.* 2007, p. 285; ASMFC 2004, p. 31). Based on modeling, Davis *et al.* (2006, p. 222) found that the horseshoe crab population in the Delaware Bay Region had been depleted and harvest levels at that time may have been too high to allow the population to rebuild within 15 years. The most recent harvest reductions were implemented in 2006 (ASMFC 2013b, pp. 9–10; ASMFC 2009a, p. 3; 38 N.J.R. 2139(a)).

- An early life-history (recruitment) bottleneck. Sweka *et al.* (2007, pp. 277, 282, 284) found that early-life-stage mortality, particularly mortality during the first year of life, was the most important parameter affecting modeled population growth, and that estimates of egg mortality have high uncertainty.

- Undocumented or underestimated mortality.

  - One possible source of error is the use of a constant 15 percent mortality for biomedically bled crabs. Some studies (Leschen and Correia 2010a, p. 135; Hurton *et al.* in Tanacredi *et al.* 2009, p. 331) have reported mortality rates of nearly 30 percent, although these results have been disputed (Dawson 2010, pp. 2–3; Leschen and Correia 2010b, pp. 8–10). Starting in 2011, the ASMFC's Horseshoe Crab Technical Committee began recommending use of a range from 5 to 30 percent mortality to reflect known levels of variability (ASMFC 2013c, p. 8; ASMFC 2012a, p. 6). However, ASMFC continues to assume a constant 15 percent mortality rate for bled crabs that are not turned over to the bait fishery (M. Hawk pers. comm. May 28, 2014; ASMFC 2013b, p. 9; ASMFC 2009a, p. 3). Further, none of these estimates account for sublethal effects. For example, in addition to 18 percent mortality of bled females, Anderson *et al.* (2013, pp. 137) found behavioral (e.g., slower movement, reduced activity levels, altered circatidal rhythms) and physiological (e.g., reduced blood proteins important to respiration and immune response) changes among the survivors. These findings suggest that biomedical bleeding may decrease female fitness and thereby impact populations (Anderson *et al.* (2013, pp. 137). As discussed above, biomedical mortality will be included in the next benchmark stock assessment (ASMFC 2013b, p. 18).

  - Poaching may be another factor, as documented by enforcement actions in New Jersey (Mucha 2011) and New York (Goodman 2013; Randazzo 2013; J. Gilmore pers. comm. October 24, 2012). However, New Jersey has had only minor cases of poaching, with violators possessing only a few crabs (ASMFC 2014a, p. 1; D. Fresco pers. comm. November 9, 2012). The high penalty in New Jersey—$10,000 for any quantity of illegal crabs—serves as a significant deterrent (ASMFC 2014a, p. 1).

    The ASMFC law enforcement committee had previously reported very few problems or issues (M. Hawk pers. comm. April 29, 2013). In early 2014, the ASMFC law enforcement committee provided an update of recent enforcement activities and issues relative to illegal harvest of horseshoe crabs. Those States where there is a significant horseshoe crab fishery reported moderate levels of illegal activity, and these are being countered or addressed to the extent that resources allow (ASMFC 2014a, p. 2). Only New York and Virginia reported what they considered notable illegal harvest (poaching) issues. New York is aware of significant harvest pressure in the spring and anticipates possible illegal activity by implementing spring enforcement details. Only a few enforcement cases have been made in New York, primarily on the new and full moons during April, May and June. There is a significant enforcement effort at peak times in support of the legal fishery in New York. Illegal activity in Virginia is mostly confined to secluded, shallow-water areas of the eastern shore. Because of this location, it is difficult to monitor activity closely and catch violators in Virginia. However, Virginia officers are well aware of the activity and make every effort to counteract it. A recent regulatory change to fishing hours in Virginia has cut down on the use of darkness as cover for illegal activity (ASMFC 2014a, p. 1).

As of early 2014, other States reported no or low levels of illegal activity, with few cases and less severe violations (ASMFC 2014a, p. 2). New Hampshire, Massachusetts, Rhode Island, New Jersey, Maryland, North Carolina, and South Carolina reported no or minimal poaching (ASMFC 2014a, pp. 1–2). States have in place or have recently adopted regulations to aid enforcement; for example, high monetary penalties, limited permits, harvest hours and seasons, harvest limits, and possession prohibitions (ASMFC 2014a, p. 2).

Although notable poaching has been reported in New York and in Virginia's non-ocean waters, these areas are outside the Delaware Bay Region and should not affect population trends in this Region, which is the only region where red knots are highly reliant on horseshoe crab eggs as a food resource. We have no data to indicate that other than minor poaching (well below the levels that would cause population effects) is occurring in the Delaware Bay Region.

- The harvest of horseshoe crabs from Federal waters that are not landed in any state, but exchanged directly to a dependent fishery (e.g., boat-to-boat transfers), is unregulated, and, therefore, the magnitude of any such harvest is unknown (ASMFC 1998, p. 27). There is no evidence that any such harvest is occurring from a targeted horseshoe crab fishery in Federal waters (M. Hawk pers. comm. April 29, 2013; G. Breese pers. comm. April 26, 2013). However, such offshore transfers may be occurring of bycatch crabs caught by offshore trawls primarily fishing for other species (ASMFC 2014a, p. 2, M. Hawk pers. comm. May 27, 2014); see the following bullet on bycatch. The level of any such unreported and unregulated harvest (i.e., that does not result in landings) is thought to be small and unlikely to have population-level effects (M. Hawk pers. comm. April 29, 2013; G. Breese pers. comm. April 26, 2013).

- Horseshoe crabs are taken as bycatch in a number of fisheries. The extent of horseshoe crab mortality due to bycatch is unknown (ASMFC 1998, pp. 22, 26); however, if landed in any state, these bycatch crabs must be included the landings data reported to ASMFC (table 23) (ASMFC 2013b, p. 15) and counted toward the yearly quota (M. Hawk pers. comm. May 27, 2014). No horseshoe crabs, even those caught as bycatch, may be kept unless the harvester holds a permit. Each State has different bycatch allowances, but the total reported landings must not exceed that State's total horseshoe crab quota (M. Hawk pers. comm. May 27, 2014). For example, Virginia has 86 permit holders authorized for horseshoe crab bycatch, of which about 15 actually landed crabs in 2014. These bycatch permits set a limit of 250 crabs per day, which is a small portion of Virginia's yearly quota (M. Hawk pers. comm. May 27, 2014).

  Horseshoe crabs caught as bycatch in the Carl N. Shuster Jr. Horseshoe Crab Reserve must be returned to the water (NOAA 2001, p. 8906). However, bycatch of horseshoe crabs in other Federal waters that are not landed in any state, but exchanged directly to a dependent fishery, is unreported and unregulated (see boat-to-boat transfers in the preceding bullet). In early 2014, several ASMFC law enforcement committee members reported hearing of potential horseshoe crab bycatch by offshore trawls, and that these crabs may be getting sold directly to fishermen for bait. While there is no indication of the extent or amount of this activity or whether it exceeds the legal bycatch allowances that are set by each State, there is no direct evidence of significant illegal activity and no enforcement cases (ASMFC 2014a, p. 2; M. Hawk pers. comm. May 27, 2014).

  - Commercial discard occurs if a waterman is not a licensed horseshoe crab permittee, if he or she has already caught more than the daily limit, or if the quota for a particular permit is closed (M. Hawk pers. comm. May 27, 2014). Commercial discard has not been quantified. Discard mortality is known to occur in various dredge fisheries. This mortality may vary seasonally with temperature and crab activity, and impacts both mature and immature horseshoe crabs (ASMFC 2013b, p. 15).

  - The ASMFC (2013a, p. 30) identified work to better estimate some of these mortality sources (e.g., characterizing landings and bycatch by life stage, estimating discard numbers and mortality rates) as moderate research priorities.

- Limitations in the ability of surveys to capture trends. Inherent variability in most of the data sets decreases the predictive power of the surveys, especially over short time periods. For the majority of horseshoe crab indices, detecting small changes in population size would require 10 to 15 years of data. Over the short term, these indices would be able to identify only a catastrophic decline in the horseshoe crab population (ASMFC 2004, p. 31).

- An ecological shift. Examples are available from other fisheries, such as weakfish (*Cynoscion regalis*). The weakfish quota was dramatically cut, but the population never rebounded. Despite some years of excellent recruitment, adult weakfish stocks have not recovered perhaps due to increased predation (S. Doctor pers. comm. November 8, 2012). Changes in predation, competition, or other ecological factors can cause a population to stabilize at a new, lower level. The ASMFC (2013a, p. 31) identified as a high research priority work to assess whether horseshoe crab population growth is or will be limited by prey availability.

In addition to the aforementioned potential causes for lack of recent growth in horseshoe crab populations, threats to horseshoe crab spawning habitat are discussed under Factor A above. Another potential threat to horseshoe crab populations emerged—the importation of nonnative horseshoe crab species for use as bait. A New York wholesaler imported about 2,000 Asian horseshoe crabs in 2011, and a dealer in Vietnam has reached out to U.S. watermen and wholesalers with the goal of developing an American market for Asian horseshoe crabs (ASMFC 2013f, p. 1). Nonnative species could carry diseases and parasites that could put the native species at risk, and exports to the U.S. bait market could hasten declines in the Asian species, which is discussed below. The Service currently lacks the regulatory authority to restrict the importation of these species at the Federal level (i.e., under the Lacey Act, see Factor D), although Congress is deliberating legislation to expand that authority to include nonnative horseshoe crabs (NECIS 2014; USFWS 2013, pp. 1–2). In the meantime, ASMFC has recommended that all member States ban the import and use of Asian horseshoe crabs as bait in State water fisheries along the Atlantic coast (ASMFC 2013g, entire), although only a few States have taken such action. Specifically, imports of Asian horseshoe crabs have been prohibited in Delaware and South Carolina, and restrictions are being formulated in Rhode Island and Maryland (M. Hawk pers. comm. June 2, 2014; DDNREC 2013). (See Factor D—United States—Invasive Species Control.)

Asian horseshoe crab species are themselves in decline (ASMFC 2013g, p. 2), and their status could indirectly affect the American species. Chinese scientists have reported rapid growth in biomedical collection and correspondingly rapid population declines in harvested populations. Anecdotal observations and predictions from scientists close to the industry suggest that such harvest is unsustainable. If the Asian biomedical industry were to collapse due to exhausted stocks of these species, then the worldwide demand for amebocyte lysate would be focused on the American horseshoe crab alone, potentially increasing biomedical collection pressure in the United States (Smith and Millard 2011, p. 1). However, research is being conducted on substitutes for LAL (PhysOrg 2011; Janke 2008, entire; Chen 2006, entire) and on artificial bait for the conch and eel fisheries (Wakefield 2013, entire; Bauers 2013; Ferrari and Targett 2003, entire). If successful, any such developments could reduce or eliminate the demand for harvesting horseshoe crabs.

Horseshoe Crab—Crab Population and Red Knot Weight Gain (Link B)

Attempts have generally not been made to tie weight gain in red knots during the spring stopover to the total horseshoe crab population size in the Delaware Bay Region. Instead, most studies have looked for correlations between red knot weight gain and either the abundance of spawning horseshoe crabs, or the density of horseshoe crab eggs in the top 2 in (5 cm) of sediment (within the reach of the birds). Other studies provide information regarding trends in egg sufficiency and red knot weight gain over time.

Link B, Part 1—Horseshoe Crab Spawning Abundance

A baywide horseshoe crab spawning survey has been conducted under consistent protocols since 1999. Baywide spawning activity shows no statistically significant trends from 1999 through 2012 (Zimmerman *et al.* 2013, p. 1). This is a change from Zimmerman *et al.* (2012, pp. 1–2), looking at the period 1999 to 2011, when, although there was no trend in females, numbers of spawning males showed a statistically significant increase from 1999 through 2011. This change can be explained by a decrease in males from 2011 to 2012 (Zimmerman *et al.* 2012, pp. 12–13). Although there was no statistically significant trend in females (baywide or in either State), the data plotted over time show a negative slope. Although there was no statistically significant trend in males (baywide or in either State), the data plotted over time show a positive slope (Zimmerman *et al.* 2013, p. 1). The ASMFC Delaware Bay Ecosystem Technical Committee recently questioned whether the spawning survey has reached "saturation" levels, at which appreciable increases in spawning crab numbers may not be detected under the current survey design. The committee is investigating this question (ASMFC 2012d, p. 7). The ASMFC (2013a, p. 38) has identified as a moderate priority research topic ground truthing the spawning survey method for calibration to the population scale.

Strong evidence for a link between numbers of spawning crabs and red knot weight gain comes from the modeling that underpins the ARM. The probability that a bird arriving at Delaware Bay weighing less than 6.3 oz (180 g) will attain a weight of greater than 6.3 oz (180 g) was positively related to the estimated female crab abundance on spawning beaches during the migration stopover (McGowan *et al.* 2011a, p. 12).

Link B, Part 2—Horseshoe Crab Egg Density

Due to the considerable vertical redistribution (digging up) of buried eggs (4 to 8 in (10 to 20 cm) deep) by waves and further spawning activity, surface egg densities (in the top 2 in (5 cm) of sediment) are not necessarily correlated with the density of spawning horseshoe crabs (Smith *et al.* 2002b, p. 733). Therefore, egg density surveys are not meant as an index of spawning horseshoe crab abundance. Neither are egg densities used to assess the overall horseshoe crab population, due to the long time to maturity, naturally high egg and larval mortality, and the fact that surface eggs typically do not survive (ASMFC 2013e, p. 3). Instead, attempts have been made to use the density of eggs in the top few inches of sediment as an index of food availability for shorebirds (Dey *et al.* 2013, p. 8), for example by correlating these egg densities with red knot weight gain.

Egg density surveys were conducted in New Jersey in 1985, 1986, 1990, and 1991, and annually since 1996. Surveys have been carried out in Delaware since 1997. Methodologies have evolved over time, but have been relatively consistent since 2005. Direct comparisons between New Jersey and Delaware egg density data are inappropriate due to differences in survey methodology between the two States, despite standardization efforts (ASMFC 2012d, pp. 11–12; Niles *et al.* 2008, pp. 33, 44, 46).

Niles *et al.* (2008, p. 45) reported New Jersey egg densities from 1985, 1986, 1990, and 1991 an order of magnitude higher than for the period starting in 1996. Conversion factors were developed to allow for comparison between the 1985 to 1986 and the 1990 to 1991 data points (Niles *et al.* 2008, p. 44), and statistical analysis found that data points from 2000 to 2004 can be directly compared to those from 2005 to 2012 without a conversion factor (i.e., a 2005 change in sampling method did not affect the egg density results) (Dey *et al.* 2011b, p. 12). However, comparisons between the earlier data points (1985 to 1999) and egg densities since 2000 are confounded by changes in methodology and investigators, and lack of conversion factors.

Higher confidence is attached to trends since 2005 because methodologies have been consistent over that period. The ASMFC's Delaware Bay Ecosystem Technical Committee reviewed egg density data from both States, and concluded there was no significant trend in baywide egg densities from 2005 to 2012. Looking at the two States separately, Delaware showed no significant trend in egg density, while the trends in New Jersey were positive.

Markedly higher egg densities on some beaches (e.g., Mispillion Harbor, Delaware and Moores Beach, New Jersey) strongly influence Statewide and baywide trends. These higher densities predictably occur in a few locations (ASMFC 2012d, p. 9). If one of these high-density beaches is excluded (Mispillion Harbor), Delaware shows no significant trend from 2005 to 2013, though there was a substantial increase from 2012 to 2013. There was a significant positive trend at Mispillion Harbor over this period, as well as in New Jersey (Dey *et al.* 2014, p. 5). Including Mispillion Harbor and focusing on the 3-week period when red knots are most abundant in Delaware Bay, Kalasz (2014, p. 5) found an increasing trend in the number of eggs available to shorebirds in Delaware from 2005 to 2013.

Using data from 2005 to 2012, Dey *et al.* (2013, pp. 8, 18) found a statistically strong relationship between the proportion of red knots reaching the estimated optimal departure weight (6.3 oz (180 g) or more) from May 26 to 28, and the baywide median density of horseshoe crab eggs, excluding Mispillion Harbor, during the third and fourth weeks of May. This statistical relationship continued to hold in 2013 (Dey *et al.* 2014, p. 4), and suggests that the egg survey data may provide a reasonable measure of egg availability and its link to red knot weight gain (ASMFC 2012d, p. 11). However, the exclusion of Mispillion Harbor is problematic because egg densities at this site are an order of magnitude higher than at other beaches (Dey *et al.* 2013, pp. 10, 14); Mispillion Harbor has supported large numbers of red knots even in years when the measure of baywide egg densities has been low, consistently containing upwards of 15 to 20 percent of all the knots recorded in Delaware Bay (Lathrop 2005, p. 4). A mathematical relationship between egg densities and red knot departure weights holds with the addition of Mispillion Harbor, but is statistically weaker (Dey *et al.* 2013, pp. 18–19; H. Sitters pers. comm. April 26, 2013). In addition, problems have been noted with both the egg density surveys and the characterization of red knot weights relative to particular dates; each are discussed below.

Regarding the egg surveys, samples are similarly collected across the bay, but egg separation and counting methodologies are substantially different between New Jersey and Delaware and have not been fully documented in either State. In addition, very high spatial and temporal variability in surface egg densities limits the statistical power of the surveys (ASMFC 2012d, p. 11). Based on the sampling methodology used in both States (Dey *et al.* 2011b, pp. 3–4), the surveys would be expected to have only about a 75 percent chance of detecting a major (50 percent) decline in egg density over 5 years (Pooler *et al.* 2003, p. 700). In addition, the sampled segments on a particular beach may not be representative of egg densities throughout that larger beach (Pooler *et al.* 2003, p. 700) and may not reflect the red knots' preferential feeding in microhabitats where eggs are concentrated, such as at horseshoe crab nests (Fraser *et al.* 2010, p. 99), the wrack line (Karpanty *et al.* 2011, p. 990; Nordstrom *et al.* 2006a, p. 438), and shoreline discontinuities (Botton *et al.* 1994, p. 614).

Data on the proportion of birds caught at 6.3 oz (180 g) or greater from May 26 to 28 should also be interpreted with caution (Dey *et al.* 2011a, p. 7). The proportion of the whole stopover population that is present in the bay and available to be caught and weighed from May 26 to 28 varies from year to year. In addition, the late May sampling event cannot take account of those birds that achieve adequate mass and either depart Delaware Bay early (Dey *et al.* 2011a, p. 7) or spend more time roosting away from the capture sites (which are located in foraging areas) (Robinson *et al.* 2003, p. 11). The fact that birds arrive and depart the stopover area at different times can also confound attempts to calculate weight gain over the course of the stopover season, underestimating the gains by as much as 30 to 70 percent (Gillings *et al.* 2009, pp. 55, 59; Zwarts *et al.* 1990, p. 352). Modeling for the ARM produced a strong finding that the probability of capturing light birds (less than 6.3 oz; 180 g) is considerably higher (0.071) than of capturing heavy birds (greater than 6.3 oz; 180 g) (0.019) (McGowan *et al.* 2011a, p. 8). In addition, a single target weight and date for departure is likely an oversimplification; while likely to hold true for the population average, individual birds likely employ diverse "strategies" for departure date and weight influenced by the bird's size, condition, arrival date, and other factors (Robinson *et al.* 2003, p. 13).

Despite the high uncertainty of the egg density data and a known bias in recorded red knot weights, these metrics do show a significant positive correlation to one another, and we have, therefore, considered this information. Although the birds captured and weighed at the end of May are very likely lighter than the population-wide average departure weight, these birds may represent a useful index of late-departing knots that may be particularly dependent on a superabundance of horseshoe crab eggs (see Asynchronies, below).

Link B, Part 3—Trends in Horseshoe Crab Egg Sufficiency

Looking at the duration that shorebirds spent in Delaware Bay early versus late in the stopover period, Wilson (1991, pp. 845–846) concluded there was no evidence of food depletion, but he did not account for time constraints that late-arriving birds may face. In 1990 and 1991, Botton *et al.* (1994, pp. 612–613) found that all but one of the seven beaches sampled were capable of supporting at least four birds per 3.3 ft (1 m) of shoreline, and the supply of eggs was sufficient to accommodate the number of birds using these beaches at that time.

By 2002 and 2003, Gillings *et al.* (2007, p. 513) found that few beaches provided high enough densities of buried eggs (2 to 8 in (5 to 20 cm) deep) for rapid egg consumption (i.e., through vertical redistribution, as discussed above), making birds dependent on a smaller number of sites where conditions were suitable for surface deposition (e.g., from the receding tide). Comparing survey data from 1992 and 2002, usage of Delaware Bay by foraging gulls declined despite growing regional gull populations, another indication that birds were responding to reduced availability of horseshoe crab eggs around 2002 (Sutton and Dowdell 2002, p. 6). Based on models of red knot foraging responses observed in 2003 and 2004, Hernandez (2005, p. 35) estimated egg densities needed to optimize foraging efficiency, and these estimates were generally consistent with requisite egg densities calculated by Haramis *et al.* (2007, p. 373) based on captive red knot feeding trials. These studies suggested that available egg densities in the early 2000s may have been insufficient for red knots to meet their energetic requirements (Niles *et al.* 2008, pp. 36–39). A geographic contraction of red knots into fewer areas of Delaware Bay may have also indicated egg insufficiency. From 1986 to 1990, red knots were relatively evenly distributed along the Delaware Bay shoreline in both New Jersey and Delaware. In comparison, there was a much greater concentration of red knots in the fewer areas of high horseshoe crab spawning activity from 2001 to 2005 (Lathrop 2005, p. 4), suggesting that, due to declining numbers, crabs (followed by the knots) had contracted to just the best spawning habitats. In 2004, Karpanty *et al.* (2006, p. 1706) found that only about 20 percent of the Delaware Bay shoreline contained enough eggs to have a greater than 50 percent chance of finding red knots, and that red knots attended most or all of the available egg concentrations.

Newer evidence suggests that the apparent downward trend in egg sufficiency may have stabilized by the mid-2000s. In 2004 and 2005, Karpanty *et al.* (2011, p. 992) found that eggs became depleted in the wrack line, but also found several other lines of evidence that egg numbers were sufficient for the red knot stopover populations present in those years. This evidence included egg counts over time, bird foraging rates and behaviors, egg exclosure experiments, and lack of competitive exclusion (Karpanty *et al.* 2011, p. 992). In addition Smith (2013, p. 1) found indications of a spatial redistribution of spawning horseshoe crabs, from more concentrated and patchy to more evenly distributed throughout the bay. This finding is a sign of reversal of the earlier geographic contraction documented by Lathrop (2005, p. 4).

Link B, Part 4—Trends in Red Knot Weight Gain

From 1997 to 2002, Baker *et al.* (2004, p. 878) found that an increasing proportion of red knots, particularly those birds that arrived late in Delaware Bay, failed to reach threshold departure masses of 6.3 to 7.1 oz (180 to 200 g). Despite using a slightly different target weight and departure date, Atkinson *et al.* (2003b, p. 3) had reached the same conclusion that, relative to 1997 and 1998, an increasing proportion of birds failed to reach target weights through 2002. Modeling conducted by Atkinson *et al.* (2007, p. 892) suggested that red knot fueling (temporal patterns and rates of weight gain) proceeded as normal from 1997 to 1999 and 2001 to 2002, but not in 2003 or 2005 due to poor foraging and weather conditions.

Dey *et al.* (2014, p. 4) found a significant quadratic (a mathematical relationship between one variable and the square of another variable) relationship between the percent of red knots weighing 6.3 oz (180 g) or more in late May (May 26 to 28) and time (1997 to 2013). The strength of the quadratic relationship owes much to the very low proportion (0 percent) of heavy birds in 2003, but it is still significant if the 2003 data are omitted. Charting this relationship shows a downward trend in the percent of heavy birds since 1997, which started to reverse by the late 2000s; however, the percent of heavy birds in late May has not yet fully returned to 1990s levels (Dey *et al.* 2014, p. 4). The percent of heavy knots was high in 2014, for the third year in a row (A. Dey pers. comm. July 23, 2014).

It is noteworthy that the downward trend in the percent of late-May heavy birds appears to have leveled off around 2005 (A. Dey pers. comm. October 12, 2012), around the same time that Karpanty *et al.* (2011, p. 992) found evidence of sufficient horseshoe crab eggs, and following the period of horseshoe crab population growth (ASMFC 2012c, pp. 10–12) that was discussed under Population Levels (Link A, Part 2), above. Peak counts of red knots in Delaware Bay have also been generally stable since approximately this same time (A. Dey pers. comm. October 12, 2012; Dey *et al.* 2011a, p. 3), although at a markedly reduced level compared to earlier decades. These lines of evidence suggest that the imminent threat of egg insufficiency was stabilized, though not fully abated, around 2005. Because of the uncertain trajectory of horseshoe crab population growth since 2005, it is not yet known if the egg resource will continue to adequately support red knot population growth in the future.

Horseshoe Crab—Red Knot Weight Gain and Survival/Reproduction (Link C)

In the causal chain from horseshoe crab harvest to red knot populations, the highest uncertainty is associated with the link between red knot weight gain at the Delaware Bay in May and the birds' survival, reproduction, or both, during the subsequent breeding season. Using data from 1997 to 2002 and slightly different target departure dates (May 31) and weights (6.9 oz (195 g)), early modeling by Atkinson *et al.* (2003b, pp. 15–16) found support for the hypothesis that birds with lower departure weights have lower survival rates and that survival rates apparently decreased over this time. Demonstrating the importance of stopover timing (see Asynchronies, below), survival rates of birds caught from May 10 to May 20 did not seem to change from 1997 to 2002, and was consistently high. However, for birds caught after May 20, the range of survival rates was much wider, and birds were predicted to have higher mortality rates (Atkinson *et al.* 2003b, p. 16).

More recently, two benchmark studies have attempted to measure the strength of the relationship between departure weight from Delaware Bay and subsequent survival using mathematical models. By necessity, this type of modeling relies on numerous assumptions, which increases uncertainty in the results. Both studies took advantage of the extensive body of red knot field data, which makes the models more robust than would be possible for less well-studied species. Nevertheless, the two modeling efforts produced somewhat inconsistent results, as discussed below.

Baker *et al*. (2004, pp. 878–897) found that annual survival declined significantly from an average of 85 percent from 1994 to 1998 to 56 percent from 1998 to 2001. Linking weight gain to survival, Baker *et al.* (2004, p. 878) found that red knots known to survive to a later year, through recaptures or resightings throughout the flyway, were heavier at initial capture than birds never seen again. According to Baker *et al.* (2004, entire), mean predicted body mass of known survivors was greater than 6.3 oz (180 g) in each year of the study (as cited in McGowan *et al.* 2011a, p. 14).

Using data from 1997 to 2008, McGowan *et al.* (2011a, p. 13) found considerably higher survival rates (around 92 percent) than Baker *et al.* (2004, entire) had reported. McGowan *et al.* (2011a, p. 9) did confirm that heavy birds had a higher average survival probability than light birds, but the difference was small (0.918 versus 0.915). Based on the work of Baker *et al.* (2004), McGowan *et al*. (2011a, p. 13) had expected a larger difference in survival rates between heavy and light birds.

However, the average survival rate (1997 to 2008) can mask differences among years. Looking at these temporal differences, the findings of McGowan *et al.* (2011a, entire) were more consistent with Baker *et al.* (2004, entire), and McGowan's year-specific survival rate estimates for 1997 to 2002 fell within the ranges presented by Baker *et al.* (2004). McGowan's lowest survival estimates occurred in 1998, just before the period of sharpest declines in red knot counts (McGowan *et al.* 2011a, p. 13) (see tables 3 and 12). Also, the survival of light birds was lower than heavy birds in 6 of the 11 years analyzed. For example, the 1998 to 1999 survival rate estimate was 0.851 for heavy birds and only 0.832 for light birds (McGowan *et al.* 2011a, p. 9). Finally, McGowan *et al.* (2011a, p. 14) noted that the data presented by Baker *et al*. (2004) show survival rates increased during 2001 and 2002. These points of comparison between the two studies suggest that the years of the Baker *et al.* (2004, entire) study may have corresponded to the period of sharpest red knot declines that have subsequently begun to stabilize. Stabilization around the mid-2000s is also supported by several other lines of evidence, as discussed under Trends in Red Knot Weight Gain (Link B, Part 4), above. However, McGowan *et al.* (2011a, p. 14) suggested several possible methodological reasons why their results differed from Baker *et al.* (2004, entire); primarily, that the newer study attempted to account for the known bias toward capturing lighter birds.

McGowan *et al.* (2011b, entire) simulated population changes of horseshoe crabs and red knots using reported horseshoe crab harvest from 1998 to 2008 and the red knot survival and mass relationships reported by McGowan *et al.* (2011a). These tests demonstrated that the survival estimates reported by McGowan *et al.* (2011a) are potentially consistent with a projected median red knot population decline of over 40 percent (McGowan *et al.* 2011a, p. 13), over the same period in which declining counts were recorded in both Delaware Bay and Tierra del Fuego.

A line of corroborating evidence comes from the demonstration of similar linkages in other *Calidris canutus* subspecies. For example, Morrison (2006, pp. 613–614) and Morrison *et al.* (2007, p. 479) linked survival rates to the departure condition of spring migrants in *C.c. islandica*.

In addition to survival, breeding success was suggested by Baker *et al.* (2004, pp. 875, 879) as being linked to food availability in Delaware Bay, based on a 47 percent decline in second-year birds observed in wintering flocks. However, there may be segregation of juvenile and adult red knots on the wintering grounds, and little information is available on where juveniles spent the winter months (USFWS and CWFNJ 2012, p. 1). Thus, shifting juvenile habitat use cannot be ruled out as a factor in the decline of young birds observed at known (adult) wintering areas.

Although Baker *et al.* (2004, p. 879) postulated that the observed decrease in second-year birds was linked to food availability in Delaware Bay, no direct links have been established between horseshoe crab egg availability and red knot reproductive success. Red knots typically do not rely on stored fat for egg production or the subsequent rearing of young, having used up most of those reserves for the final migration flight and initial survival on the breeding grounds (Morrison 2006, p. 612; Piersma *et al.* 2005, p. 270; Morrison and Hobson 2004, p. 341; Klaassen *et al.* 2001, p. 794). The fact that body stores are not directly used for egg or chick production suggests that horseshoe crab egg availability is unlikely to affect red knot reproductive rates, other than through an influence on the survival of prebreeding adults. However, studies of shorebirds as a group indicate that if birds arrive in a poor energetic state on the destination area, they would have a very small chance of reproducing successfully (Piersma and Baker 2000, p. 123). Further, from studies of the *Calidris canutus islandica*, Morrison (2006, pp. 610–612) and Morrison *et al.* (2005, p. 449) found that a major function of stored fat and protein may be to facilitate a transformation from a physiological state suitable for migration to one suitable, and possibly required, for successful breeding. These findings suggest that a more direct link between the condition of red knots leaving Delaware Bay and reproductive success could exist but has not yet been documented. Modeling for the ARM includes components to test for linkages between Delaware Bay departure weights and reproductive success and could provide future insights into this question (McGowan *et al.* 2011b, p. 118).

Horseshoe Crab—Adaptive Resource Management

In 2012, the ASMFC adopted the ARM for the management of the horseshoe crab population in the Delaware Bay Region (ASMFC 2012e, p. 1). The ARM was developed with input from shorebird and fisheries biologists from the Service, States, and other agencies and organizations. The ARM modeling links horseshoe crab and red knot populations, to meet the dual objectives of maximizing crab harvest and meeting red knot population targets (McGowan *et al.* 2011b, p. 121). The ARM uses competing models to test hypotheses and eventually reduce uncertainty about the influence that food conditions in Delaware Bay exert on red knot populations (McGowan *et al.* 2011b, pp. 130–131). The framework is designed as an iterative process that adapts to new information and the success of management actions (ASMFC 2012e, p. 3). Under the ARM, the horseshoe crab harvest caps authorized by ASMFC are explicitly linked to red knot population targets starting in 2013 (ASMFC 2012e, p. 4).

As long as the ARM is in place and functioning as intended, ongoing horseshoe crab bait harvests should not be a threat to the red knot. This conclusion about the ARM is based on: (1) technical soundness of the peer reviewed models; (2) the linking of horseshoe crab harvest quotas to red knot population targets; and (3) the adaptive nature of both the models and the framework that are intended to regularly adjust as new information becomes available. This conclusion is supported by the findings of Smith *et al.* (2013, p. 8) that red knot population trajectories under horseshoe crab harvest scenarios governed by the ARM almost matched simulated red knot population trajectories under a fixed horseshoe crab moratorium scenario. Thus, the bait harvest levels allowed by the ARM are expected to have a negligible effect on the red knot's Delaware Bay stopover population.

The harvest regulations recommended by the ARM require data from annual, baywide monitoring programs, including a horseshoe crab trawl survey (previously conducted by the Virginia Polytechnic Institute (Virginia Tech)) and the Delaware Bay Shorebird Monitoring Program. The Virginia Tech Trawl Survey was ended after 2012 due to lack of funding. We conclude that the Virginia Tech survey is the best benthic trawl survey to support the ARM. The ARM modelers have attempted to switch to another, newer survey, the North East Area Monitoring and Assessment Program (NEAMAP) coordinated by the Virginia Institute of Marine Science. In the absence of the Virginia Tech survey, we support the efforts of the ASMFC to adapt the NEAMAP data for use in the models, so that the ARM framework may continue to function. As of fall 2014, however, these efforts have not identified a method by which NEAMAP or other alternate data sets can be appropriately used for the full and proper functioning of the ARM models (ASMFC 2014b). Stable funding sources for the other baywide Shorebird Monitoring Program are also a concern. Insufficient monitoring has already affected the ability of the ASMFC to implement the ARM as intended (ASMFC 2014b; ASMFC 2012c, p. 13). If the ARM cannot be implemented in any given year, ASMFC would choose between two options based on which it determines to be more appropriate—either use the previous year's harvest levels (as previously set by the ARM), or revert to an earlier management regime (known as Addendum VI, which was in effect from August 2010 to February 2012) (ASMFC 2012e, p. 6; ASMFC 2010, entire). Although the horseshoe crab fishery would continue to be managed under either of these options, the explicit link to red knot populations would be lost. For the 2015 season, ASMFC (2014b) has opted to use the previous year's harvest levels.

In addition, some uncertainty exists regarding how to define the Delaware Bay horseshoe crab population. Currently all crabs harvested from New Jersey and Delaware, as well as part of the harvests from Maryland and Virginia, are believed to come from the Delaware Bay population. This conclusion was based on resightings in these four States of crabs that had been marked with tags in Delaware Bay from 1999 to 2003 (ASMFC 2006, p. 4). Further work (tagging and genetic analysis) suggests that little exchange occurs between the Delaware Bay and Chesapeake Bay horseshoe crab populations, but crabs do move between Delaware Bay and the Atlantic coastal embayments from New Jersey through Virginia (ASMFC 2012e, pp. 3–4; Swan 2005, p. 28; Pierce *et al.* 2000, p. 690). However, other information adds complexity to our understanding of the population structure. In a genetic analysis of horseshoe crabs from Maine to Florida's Gulf coast, King *et al.* (2005, p. 445) found four distinct regional groupings, including a mid-Atlantic group extending from Massachusetts to South Carolina. In addition, in a long-term tagging study, Swan (2005, p. 39) found evidence suggesting the existence of subpopulations of Delaware Bay horseshoe crabs. Finally, since most tagging efforts, and most resightings of tagged crabs, occur on spawning beaches, the distribution and movements of horseshoe crabs in offshore waters (where most of the harvest occurs via trawls) are poorly known (Swan 2005, pp. 30, 33, 37). In documenting the technical underpinnings of the ARM, the ASMFC (2009b, p. 7) acknowledged that the proportion of Maryland and Virginia landings that come from Delaware Bay is currently unresolved, but stated that their approach to estimating this proportion was conservative. We conclude that the ASMFC's current delineation of the Delaware Bay Region horseshoe crab population is based on best available information and is appropriate for use in the ARM modeling, but we acknowledge some uncertainty regarding the population structure and distribution of Delaware Bay horseshoe crabs. The ASMFC (2013a, p. 30) has identified as a moderate priority area of research characterizing the proportion of states' landings that comprise horseshoe crabs of Delaware Bay origin. We anticipate the ARM process will adapt to substantive new information that reduces uncertainty about the Delaware Bay horseshoe crab population structure and geographic distribution.

Likewise, we expect the ARM framework will continue to adapt as substantive new information becomes available about any important factors (other than the bait harvest) that may limit the continued growth of the of the Delaware Bay horseshoe crab population. Such factors are not currently well known, but could include demographic and ecological constraints on population growth, as well as sources of direct mortality that are not currently captured by the ARM models (e.g., biomedical, poaching, bycatch). In particular, accounting for biomedical mortality may become important if the contribution of the biomedical harvest to total mortality continues to increase. It should be noted, however, that much of the biomedical harvest occurs outside the Delaware Bay Region and would therefore fall outside of the ARM framework.

Food Availability—Summary

Reduced food availability at the Delaware Bay stopover site due to commercial harvest of the horseshoe crab is considered a primary causal factor in the decline of rufa red knot populations in the 2000s (Escudero *et al.* 2012, p. 362; McGowan *et al.* 2011a, pp. 12–14; Niles *et al.* 2008, pp. 1–2; Baker *et al.* 2004, p. 875). Due to harvest restrictions and other conservation actions, horseshoe crab populations showed some signs of recovery in the early 2000s, with apparent signs of red knot stabilization (survey counts, rates of weight gain) occurring a few years later (as might be expected due to biological lag times). Since about 2005, however, horseshoe crab population growth has stagnated for unknown reasons (ASMFC 2009a, p. 23; ASMFC 2013b, p. 22; ASMFC 2012c, pp. 10–12). Under the current management framework (the ARM), the present horseshoe crab harvest is not considered a threat to the red knot. However, continued implementation of the ARM is imperiled by lack of funding to support the requisite monitoring programs. With or without the ARM, it is not yet known if the horseshoe crab egg resource will continue to adequately support red knot population growth over the next decade.

The causal role of reduced Delaware Bay food supplies in driving red knot population declines shows the vulnerability of red knots to declines in the quality or quantity of their prey. This vulnerability has also been demonstrated in other *Calidris canutus* subspecies, although not to the severe extent experienced by the rufa red knot (B. Harrington pers. comm. November 14, 2013; van Gils *et al.* 2006, p. 2399; Atkinson *et al.* 2003a, p. 127; Piersma *et al.* 1993, pp. 343, 354). In addition to the fact that horseshoe crab population growth has stagnated, red knots now face several emerging threats to their food supplies throughout their nonbreeding range. These threats include small prey sizes (from unknown causes) at two key wintering sites on Tierra del Fuego (Escudero *et al.* 2012, pp. 359–362; Espoz *et al.* 2008, pp. 69, 74) warming water temperatures that may cause mollusk population declines and range contractions (including the likely loss of a key prey species from the Virginia spring stopover within the next decade (Jones *et al.* 2010, pp. 2255–2256), ocean acidification to which mollusks are particularly vulnerable (IGBP *et al.* 2013, pp. 9, 16; NRC 2010b, pp. 68–69), physical habitat changes from climate change affecting invertebrate communities (Chapter 5 in IPCC 2014, p. 21; Rehfisch and Crick 2003, p. 88; Najjar *et al.* 2000, p. 225), possibly increasing rates of mollusk diseases due to climate change (Ward and Lafferty 2004, p. 543), invasive marine species from ballast water and aquaculture (Seebens *et al.* 2013, p. 782; Ruesink *et al.* 2005, pp. 671–674; Grosholz 2002, p. 22–23), and the burial and crushing of invertebrate prey from sand placement and recreational activities (Sheppard *et al.* 2009, p. 113; Schlacher *et al.* 2008b, pp. 345, 348; Schlacher *et al.* 2008c, pp. 878, 882; Greene 2002, p. 24). Although threats to food quality and quantity are widespread, red knots in localized areas have shown some adaptive capacity to switch prey when the preferred prey species became reduced (Escudero *et al.* 2012, pp. 359, 362; Musmeci *et al.* 2011, entire), suggesting some adaptive capacity to cope with this threat. Nonetheless, based on the combination of documented past impacts and a spectrum of ongoing and emerging threats, we conclude that reduced quality and quantity of food supplies is a threat to the rufa red knot at the subspecies level, and the threat is likely to continue into the future.

Factor E—Asynchronies During the Annual Cycle

Asynchronies—Mechanisms and Vulnerabilities

Long-distance migrants breeding at high latitudes face severe time pressures and have evolved efficient behaviors and adaptations (both physiological and morphological) that allow the maximum possible migration speed, although the minimization of energy costs and predation risk may also be involved (Hedenström 2008, p. 287). For Arctic-nesting shorebirds, the timing of arrivals and departures from wintering, stopover, and breeding areas must be precise because prey abundance at staging areas is cyclical, and there is only a narrow window in the arctic summer for courtship and reproduction (Botton *et al.* in Shuster *et al*. 2003, p. 6). Because the arctic breeding season is short, northbound birds must reach the nesting grounds as soon as the snow has melted. Early arrival and rapid nesting increase reproductive success (Conklin *et al.* 2010, p. 2; Gill *et al.* 2013, p. 1; Henkel *et al.* 2012, p. 681; Myers *et al.* 1987, pp. 21–22), (e.g., through favorable territory or mate availability) (Conklin *et al.* 2010, p. 4; Grabowski *et al.* 2013, p. 1098). However, a countervailing time constraint is that the seasonal supply of food resources along the migration pathways prevents shorebirds from moving within flight distance of the breeding grounds until late spring (Myers *et al.* 1987, pp. 21–22). Harsh weather and risk of starvation (e.g., because inset prey are not yet available) on the breeding grounds are additional countervailing constraints on arriving in the Arctic too early (Grabowski *et al.* 2013, p. 1098; Gill *et al.* 2013, p. 1). The timing of southbound migration is also constrained, because the abundance of quality prey at stopover sites gradually decreases as the fall season progresses (van Gils *et al.* 2005b, pp. 126–127; Myers *et al.* 1987, pp. 21–22; Schneider and Harrington 1981, p. 801). Migration timing is also influenced by the enormous energy required for birds to complete the long-distance flights between wintering and breeding grounds. Northbound shorebirds migrate in a sequence of long-distance flights alternating with periods of intensive feeding to restore energy reserves. Most of the energy stores are depleted during the next flight; thus, a bird's ability to accumulate a small additional energetic reserve may be crucial if its migration gets delayed by poor weather or if feeding conditions are poor upon arrival at the next destination (Myers *et al.* 1987, pp. 21–22).

Migrating animals may incur fitness consequences if their migration timing and the availability of resources do not coincide (i.e., are asynchronous or "mismatched"), particularly for species like the red knot that show fidelity to sites with ephemeral food and habitat resources used to fuel long-distance migration. The joint dynamics of resource availability and migration timing may play a key role in influencing annual shorebird survival and reproduction. The mismatch hypothesis is of increasing relevance because of the potential asynchronies created by changes in phenology (the timing of periodic life-cycle events) related to global climate change (Galbraith *et al.* 2014, entire; Iwamura *et al.* 2013, p. 6; McGowan *et al*. 2011a, p. 2; Smith *et al.* 2011a, p. 575; Meltofte *et al.* 2007, p. 36). (In this document, we use "asynchronies" to mean mismatches between the timing of a species' annual breeding, migration, and wintering cycles and the windows of peak food availability or favorable weather conditions on which it depends.)

Globally, phenological changes are already underway and well-documented across many taxa, in both terrestrial and coastal ecosystems (Summary for Policymakers in IPCC 2014, p. 30). The likelihood of mismatches arises because species' phenological responses to a changing climate (i.e., the extent to which they change the timing of their annual cycles) vary widely (Hurlbert and Liang 2012, p. 3), throwing one or more species (or whole communities) out of sync with one another, with favorable environmental conditions, or both. The degree of mismatch may differ among species, and species with greater mismatches may be characterized by declining populations. For example, Møller *et al.* (2008, p. 16195) found that European bird species that declined from 1990 to 2000 did not advance (make earlier) their spring migrations, whereas those with stable or increasing populations advanced their migrations considerably; these results are consistent with other studies (Hurlbert and Liang 2012, p. 4).

Red knots have high vulnerability to asynchronies. First, late arrival of red knots in Delaware Bay from unknown causes was a key synergistic factor (acting in conjunction with reduced availability of horseshoe crab eggs) accounting for population declines in the 1990s and 2000s (Baker *et al*. 2004, p. 878), demonstrating that asynchronies can have population-level effects on this species. Second, several aspects of the red knot's life history, discussed below, indicate high vulnerability to asynchronies. Negative effects of trophic (food) mismatch are most likely for species with complex annual life cycles that migrate long distances to breed, and for species that depend on resources available on the breeding grounds for egg production and successful reproduction (Liebezeit *et al.* 2014, p. 2), all of which are characteristics of the red knot. (See the sections that follow regarding asynchronies between the timing of the red knot's seasonal windows and the periods of peak availability of its prey.) A recent assessment of the red knot's vulnerability to climate change indicated a large increase in extinction risk, in part due to its high degree of dependence on ecological synchronicities (Galbraith *et al.* 2014, p. 7 and Supplement 1).

The recent advances in timing of bird migration have been linked to changes in climatic conditions, but the mechanisms driving shifts in timing of migration are often unknown (Gill *et al.* 2013, p. 1). Shorebird migration depends primarily on celestial cues (e.g., day length) and is, therefore, less influenced by environmental variation (e.g., water or air temperatures) than are the life cycles of many of their invertebrate prey species (McGowan *et al.* 2011a, p. 16). Further, species that migrate long distances must typically rely on endogenous (caused by factors inside the animal) cues to initiate migration, whereas short-distance migrants may by more reliant on climatic cues to time migration (Liebezeit *et al.* 2014, p. 10; Gill *et al.* 2013, p. 1). To the extent that they rely on nonclimatic (e.g., celestial or endogenous) timing cues, long-distance migrant shorebirds like the red knot are vulnerable to increasing frequency or severity of asynchronies (both with weather and with their climatically-cued prey) due to climate change. Studying the bar-tailed godwit (*Limosa lapponica baueri*), an Arctic-nesting, long-distance migrant shorebird, Conklin *et al.* (2010, pp. 1–2) concluded that tightly scheduled movements on a global scale suggest endogenously controlled routines, with differences in breeding latitude as the primary driver of temporal variation throughout the annual cycle. Studying captive *Calidris canutus canutus* held under a constant temperature and light regime for 20 months, Cadée *et al.* (1996, p. 82) found evidence for endogenous circannual (approximately annual) rhythms of flight feather molt, body mass, and plumage molt. Studying *C.c. canutus* and *C.c. islandica,* Jenni-Eiermann *et al.* (2002, p. 331) and Landys *et al.* (2004, p. 665) found evidence that thyroid and corticosterone hormones play a role in regulating the annual cycles of physical changes.

We have no evidence concerning the exact nature of the external timers that synchronize such endogenous rhythms to the outside world (Cadée *et al.* 1996, p. 82). Photoperiod (i.e., day length) is known to be a powerful timer for many species' circannual rhythms, and a role for day length as a timer is consistent with observations that captive *Calidris canutus canutus* exposed to day length variation in outdoor aviaries retained pronounced annual cycles in molt and body mass; however, these experiments do not exclude a role for additional timers besides photoperiod. The complex nature of the annual changes in photoperiod experienced by trans-equatorial migrants is not fully understood; this is especially true for such birds like *C. canutus* where some populations winter in the southern hemisphere while other populations winter in the northern hemisphere (Cadée *et al.* 1996, p. 82). Uncertainty exists about the extent to which the timing of the red knot's annual cycle is controlled by endogenous and celestial factors (as opposed to climatic factors). However, based on the experiments with captive *C.c. canutus* and studies of other arctic-nesting, long-distance migrant shorebirds, we conclude that some degree of reliance on endogenous and celestial timing mechanisms will at least partly constrain the knot's ability to adapt to shifting temporal and geographic patterns of favorable food and weather conditions that are expected to occur with global climate change.

Changes in the timing of spring migration have been widely reported for many species in recent decades, and advances in migration are among the most commonly reported phenological responses to climatic change (Gill *et al.* 2013, p. 1). Bird species that migrate over shorter distances are frequently reported to have advanced more than longer distance migrants (Conklin *et al.* 2010, p. 4; Gill *et al.* 2013, p. 1; Hurlbert and Liang 2012, p. 1; Petersen *et al.* 2012, p. 65; Rubolini *et al.* 2007, p. 135). This pattern suggests long-distance migrants may be less capable of responding to changing conditions at their destinations, because these species have stronger endogenous control of migration timing, because the greater migration distance reduces their capacity to predict conditions on the breeding grounds, or both (Gill *et al.* 2013, p. 1). These findings suggest short-distance migrants may respond more strongly to climate change than long-distance migrants, such as the red knot, which might adapt more slowly, resulting in less time for breeding and potentially mis-timed breeding in this group. These results also suggest differential adaptation capacities between short- and long-distance migrants could alter the interspecific competition pressures faced by various species (Petersen *et al.* 2012, p. 70) (i.e., due to the formation of new and novel assemblages of bird species that did not previously occur together in space and time (see Root *et al.* 2013, p. 82)).

As discussed above, the red knot's long-distance migration strategy may slow or limit its ability to adapt its phenology to changing climatic conditions. Further data suggesting red knots may exhibit limited flexibility in the timing of its annual cycle come from the findings of Conklin *et al.* (2010), who studied an Alaska-breeding, long-distance migrant shorebird species (bar-tailed godwit) that shares several life history traits with the red knot. Conklin *et al.* (2010, pp. 2–4) found both northbound and southbound migration timing of bar-tailed godwits were significantly correlated with breeding latitude. The relationship between migration timing and latitude became stronger with each stage of the northbound migration, implying a tightening of these programmed schedules with proximity to the breeding grounds. The timing of southbound migration was unresponsive to the duration of the breeding season. The findings of this study suggest rigidity in the migration schedule, perhaps evolved to best exploit predictable peaks of food resources or favorable wind conditions. Like southern-wintering red knots, bar-tailed godwits undergo an extreme migration, which may limit flexibility in annual activities such as molt, fuelling, and migration route, and which may produce constraints operating on the entire annual cycle. The potentially conflicting pressures of optimal timing of migration and breeding may make such long-distance migrants particularly vulnerable to the effects of climate change, if rigid flight schedules contribute to a mismatch between breeding arrival and optimal nest initiation, or preclude adaptation to temporal shifts in resources or weather during migration (Conklin *et al.* 2010, p. 4).

The high latitude at which red knots breed may be another factor limiting their adaptive capacity regarding a phenological response to climate change. Hurlbert and Liang (2012, pp. 4–

5) found that, across North American passerines, a given change in spring temperature resulted in less of a phenological shift at higher compared to lower latitudes. The weaker phenological response to temperature change at high latitudes has been observed despite the fact that higher latitudes have experienced greater warming than lower latitudes over the past decade (Hurlbert and Liang (2012, p. 5). Likewise, Rubolini *et al.* (2007, p. 135) found the advance in first arrival date in European birds was weaker at extreme latitudes.

The successful annual migration and breeding of red knots is highly dependent on the timing of departures and arrivals to coincide with favorable food and weather conditions. The frequency and severity of asynchronies is likely to increase with climate change. In addition, stochastic encounters with unfavorable conditions are more likely to result in population-level effects for red knots now than when population sizes were larger, as reduced numbers may have reduced the resiliency of this subspecies to rebound from impacts.

Asynchronies—Delaware Bay

Because shorebird staging times are shortest and fueling rates are highest at the last stopover site before birds head to the arctic breeding grounds, there appears to be little "slack" time at late stages in the migration (González *et al.* 2006, p. 115; Piersma *et al.* 2005, p. 270) (i.e., birds need to arrive and depart within a narrow time window and need to attain rapid weight gain during that window). For a large majority of red knots, the final stopover before the Arctic is in Delaware Bay.

Delaware Bay—Late Arrivals

Baker *et al*. (2004, p. 878) found that the late arrival of red knots in Delaware Bay was a key synergistic factor (acting in conjunction with reduced availability of horseshoe crab eggs) accounting for declines in survival rates, comparing the period 1994 to 1996 with the period 1997 to 2000. These authors noted that red knots from southern wintering areas (Argentina and Chile) tended to arrive later than northern birds throughout the study period, but more so in 2000 and 2001. A large number of knots arrived late again in 2002 (Robinson *et al.* 2003, p. 11). In data from 1998 to 2002, Atkinson *et al.* (2003b, p. 16) found increasing evidence that numbers of light-weight birds were passing through the bay between May 20 and 30. Corroborating evidence comes from Argentina and suggests that, for unknown reasons, northward migration of Tierra del Fuego birds had become 1 to 2 weeks later since 2000 (Niles *et al*. 2008, p. 2), which probably led to more red knots arriving late in Delaware Bay.

Research has shown that late-arriving birds have the ability to make up lost time by gaining weight at a higher rate than usual, provided they have sufficient food resources (Niles *et al*. 2008, p. 2; Atkinson *et al.* 2007, pp. 885, 889; Robinson *et al.* 2003, pp. 12–13). However, late-arriving birds failed to do so in years (e.g., 2003, 2005) when horseshoe crab egg availability was low (Niles *et al*. 2008, p. 2; Atkinson *et al.* 2007, p. 885). Looking at data from 1998 to 2002, Atkinson *et al.* (2003b, p. 16) found that intra-season rates of weight gain had not changed significantly. Using an early model linking red knot weight gain and subsequent survival, these authors concluded that arriving late was actually a more significant factor than food availability in the declining percentage of red knots reaching target weights by the end of May (Atkinson *et al.* 2003b, p. 16). In a later modeling effort, Atkinson *et al.* (2007, p. 892) confirmed that fueling (temporal patterns and rates of weight gain) proceeded as normal from 1997 to 1999, from 2001 to 2002, and in 2004, but fueling was below normal in 2000, 2003, and 2005 due to poor foraging and weather conditions. The results of Atkinson *et al.* (2007, p. 892) suggest that the reduced survival rates calculated by Baker *et al.* (2004, entire) from 1998 to 2002 were more likely the result of late arrivals than food availability, since fueling was normal in all but one of those years.

The effects of weather on the red knot's migratory schedule were documented in 1999, when a La Niña event (an occasional abnormal cooling of tropical waters in the eastern Pacific from unknown causes) occurred and the red knots migrating to Delaware Bay were subject to extended, strong headwinds (Robinson *et al.* 2003, pp. 11–12). The first birds arrived almost a week later than normal. Although most red knots had left Delaware Bay by the end of May, an unusually large number (several thousand) of knots were recorded in central Canada in mid-June, suggesting that many birds did not reach the breeding grounds or quickly returned south without breeding in that year. It is possible that many birds did not put on adequate weight as a result of the weather-induced delay and were not in a good enough condition to breed (Robinson *et al.* 2003, pp. 11–12). In addition to the unknown causes that may have contributed to chronic late arrivals in Delaware Bay in the 2000s, stochastic weather events like the 1999 La Niña can affect the timing of the red knot's annual cycle and may become more erratic or severe due to climate change.

<u>Delaware Bay—Timing of Horseshoe Crab Spawning</u>

Even those red knots arriving early or on time in Delaware Bay are very likely to face poor feeding conditions if horseshoe crab spawning is delayed. Feeding conditions for red knots were poor in those years when the timing of the horseshoe crab spawn was out of sync with the birds' spring stopover period. In years that spawning was delayed due to known weather anomalies (e.g., cold weather, storms), the proportion of knots reaching weights of 6.3 oz (180 g) or greater at the end of May was very low (e.g., 0 percent in 2003) (Dey *et al.* 2011a, p. 7; Atkinson *et al*. 2007, p. 892). These observed correlations were confirmed by the ARM modeling. The models found strong evidence that the timing of horseshoe crab spawning, not simply crab abundance, is important to red knot refueling during stopover. If spawning is delayed, even with relatively high total crab abundance, the probability that a light bird will add enough mass to become a heavy bird before departure may be lower (McGowan *et al*. 2011a, p. 12). The timing of horseshoe crab spawning is closely tied to water temperatures, and can be delayed by storms. If water temperatures or storm patterns in the mid-Atlantic region were to change significantly, the timing of spawning could shift and become temporally mismatched with shorebird migration (McGowan *et al.* 2011a, p. 16).

Horseshoe Crab Spawn—Storms and Weather

Normal variation in weather is a natural occurrence and is not considered a population-level threat to the red knot. However, adverse weather events in Delaware Bay can throw off the timing of horseshoe crab spawning relative to the red knot's stopover period. Such events have the potential to impact a majority of the red knot population, as most birds pass through Delaware Bay in spring (Brown *et al.* 2001, p. 10). Synergistic effects have also been noted among such weather events, habitat conditions, and insufficient horseshoe crab eggs (Dey *et al.* 2011a, p. 7).

The Delaware Bay stopover period occurs between the typical nor'easter (October through April) and hurricane (June through November) storm seasons (National Hurricane Center 2012; Frumhoff *et al.* 2007, p. 30). However, late nor'easters do occur in May, such as occurred in 2008 when horseshoe crab spawning was delayed and red knot feeding conditions were poor. Unusual wind and rain conditions can also affect the red knots' distribution among Delaware Bay beaches and length of stay, causing variations in their activity and habitat selection. High wind and weather events are common in May and in some years limit horseshoe crab spawning to creek mouths that are protected from rough surf (Dey *et al.* 2011b, pp. 1–2; Clark *et al.* 1993, p. 702). High wave energies transport more eggs in the swash zone (the zone of wave action), but these eggs are dispersed or buried, and fewer eggs remain on the beach where they are available to shorebirds (Nordstrom *et al.* 2006a, p. 439).

High wave conditions curtail horseshoe crab spawning (Nordstrom *et al.* 2006a, p. 439). Smith *et al.* (2011a, pp. 575, 581) found that onshore winds that generate waves can delay spawning and create an asynchrony for migrating red knots. High levels of food abundance can offset some small mismatches in migration timing. Thus, increasing abundance of horseshoe crab eggs throughout the stopover period could act as a hedge against temporal mismatches between the horseshoe crab and shorebird migrations, at least in the near term. Also, select beaches with high spawning activity and capacity to retain eggs in surface sediments during episodes of high onshore winds could provide a reserve of horseshoe crab eggs during the shorebird stopover period, even in years when winds cause asynchrony between red knot migration and horseshoe crab spawning (Smith *et al.* 2011a, pp. 575, 581). Therefore, a superabundance of horseshoe crab eggs and sufficient high-quality foraging habitats can serve to partially offset asynchronies between the red knot stopover and the peak of horseshoe crab spawning.

Future frequency or intensity of storms in Delaware Bay during the stopover season may change due to climate change, but predictions about future tropical and extra-tropical storm patterns have only "low to medium confidence" (see Climate Change—Coastal Storms and Extreme Weather). (See Other Spring Stopover Areas, below, regarding apparent tendency toward more common early-season hurricanes.) Should storm patterns change, red knots in Delaware Bay would be more sensitive to the timing and location of coastal storms than to a change in overall frequency. Changes in the patterns of tropical or extra-tropical storms that increase the frequency or severity of these events in Delaware Bay during May would likely have dramatic effects on red knots and their habitats (Kalasz 2008, p. 41) (e.g., through direct mortality, delayed horseshoe crab spawning, delayed departure for the breeding grounds, and short-term habitat loss).

Horseshoe Crab Spawn—Water Temperatures

More certainty is associated with a correlation between the timing of horseshoe crab spawning and ocean water temperatures, based on a study by Smith and Michels (2006, pp. 487–488). Although horseshoe crabs spawn from late spring into early summer, migratory shorebirds use Delaware Bay for only a few key weeks in May and early June. In some years, horseshoe crab spawning has been early, with a high proportion of spawning activity occurring in May, and therefore better synchronized with the shorebird stopover period. In other years spawning has been late, with a low proportion of spawning in May, resulting in poor shorebird feeding conditions during the stopover period. Average daily water temperature has been statistically correlated with the percent of spawning that takes place in May, though the relationship is stronger in New Jersey than in Delaware. In the years with the lowest May spawning percentages, average water temperatures did not exceed 57.2 ºF (14 ºC) during May, and daily water temperatures were not consistently above 59 ºF (15 ºC) until late May. In the other years, daily water temperatures were consistently above 59 ºF (15 ºC) by mid-May (Smith and Michels 2006, pp. 487–488). After adjusting for the day of the first spring tide, the day of first spawning has been 4 days earlier for every 1.8ºF (1 ºC) rise in mean daily water temperature in May (Smith *et al.* 2010b, p. 563).

Climate change does not necessarily mean a linear increase in temperatures and an amelioration of winters in the mid-Atlantic region. As the climate changes, we could see both extremes of weather from year to year, with some years being warmer and others being colder. The colder years could cause horseshoe crab spawning to be delayed past the shorebird stopover period (Kalasz 2008, p. 41). In addition, impacts to red knots from increasingly extreme precipitation events (see Climate Change—Coastal Storms and Extreme Weather) are not known, but may include temporary water temperature changes that could affect the timing of horseshoe crab spawning activity.

Conversely, average air and water temperatures are expected to continue rising. In the Northeast, annual average air temperature has increased by 2 ºF (1.1 ºC) since 1970, with winter temperatures rising twice as much (USGCRP 2009, p. 107). Over the next several decades, temperatures in the Northeast are projected to rise an additional 2.5 to 4 ºF (1.4 to 2.2 ºC) in winter and 1.5 to 3.5 ºF (0.8 to 1.9 ºC) in summer (USGCRP 2009, p. 107). Coastal waters are “very likely” to continue to warm by as much 4 to 8 ºF (2.2 to 4.4 ºC) in this century, both in summer and winter (USGCRP 2009, p. 151). Spring migrating red knots could benefit if warming ocean temperatures result in fewer years of delayed horseshoe crab spawning. However, earlier spawning could exacerbate the problems faced by late-arriving knots that already struggle to gain sufficient weight. Under extreme warming, the timing of peak spawning could theoretically even shift earlier than the peak red knot stopover season. Using the findings of Smith *et al.* (2010b, entire), spawning could shift nearly 9 to 18 days earlier with water temperature increases of 4 to 8 ºF (2.2 to 4.4 ºC).

Asynchronies—Other Spring Stopover Areas

Outside of Delaware Bay, migrating red knots feed primarily on bivalves and other mollusks (see Wintering and Migration Food). Spring migrating knots seem to follow a northward “wave” in prey quality (i.e., flesh-to-shell ratios); research suggests that the birds locate and time their stopovers to coincide with local peaks in prey quality, which occur during the reproductive seasons of intertidal invertebrates (van Gils *et al.* 2005a, p. 2615) when normally hard-shelled bivalves (i.e., difficult to digest especially given the birds’ physiological digestive changes) are made available to knots through spat or juveniles with thinner shells. Thus, red knots are vulnerable to changes in the reproductive timing of their prey, such as could be precipitated by climate change. (See Reduced Food Availability, above, regarding threats from overall reduced prey quantity.)

Based on a long-term data set (1973 to 2001) from the western Wadden Sea, Philippart *et al.* (2003, p. 2171) found that population dynamics of common intertidal bivalves are strongly related to seawater temperatures, and rising seawater temperatures affect recruitment by decreasing reproductive output and advancing the timing of bivalve spawning in spring. For benthic mollusks, Poloczanska *et al.* (2013, p. 4) found advancing (earlier) phenology of about 10 days per decade, although sample size was only two.

Across regions and taxa, spring phenology has been reported advancing on average by 2.3 to 2.8 days per decade on land, and by 4.3 days per decade in the oceans. One reason for variability in estimates of responses could be that patterns of climate change are dynamic and highly heterogeneous across the Earth (Burrows *et al.* 2011, p. 652). Despite faster warming on land, change in seasonal timing is generally greater in the ocean because of smaller seasonal thermal variation. Shifts in the timing of spring temperatures (1960 to 2009) were 30 to 40 percent faster in the ocean than on land. Spring ocean temperatures arrived earlier by 2.08 days per decade in the Northern Hemisphere and 2.52 days per decade in the Southern Hemisphere (median values excluding equatorial regions), but by 1.46 and 2.15 days per decade, respectively, on land (Burrows *et al.* 2011, p. 654). Likewise, Poloczanska *et al.* (2013, pp. 3–4) found spring phenology in the ocean has advanced by 4.4 ± 0.7 days per decade, compared to 2.3 to 2.8 days per decade of spring phenological advancement on land. Thus, at the Arctic and middle latitudes, a greater advance in spring timing in the ocean than on land is evident (Burrows *et al.* 2011, p. 654; Poloczanska *et al.* 2013, p. 4), which suggests that the phenology of the red knot will advance more slowly than that of its marine prey (see Mechanisms and Vulnerabilities, above, regarding life history factors that suggest red knots will be slow or limited in their ability to advance their spring migrations).

Based on observations from 1998 to 2003, González *et al.* (2006, p. 109) found that an early March departure date of red knots from San Antonio Oeste, Argentina, generally corresponded to an early arrival date in Delaware Bay. The early migrating birds exhibited a higher return rate in later years, suggesting higher survival rates for red knots that arrive earlier in Delaware Bay. These findings are consistent with observation from Delaware Bay that an increasing number of late-arriving knots, along with reduced horseshoe crab egg availability, were both tied to lower survival rates observed in the early 2000s (Niles *et al*. 2008, p. 2; Baker *et al*. 2004, p. 878).

At Fracasso Beach on Península Valdés, Argentina, Hernández (2009, p. 208) found a significant correlation during March and April between the presence of shorebirds and the biomass of the clam *Darina solenoids*, suggesting that the occurrence of shorebirds at this site must depend largely on the available food supply. Analysis of weekly counts at Fracasso Beach during March and April from 1994 to 2005 showed some trends in the phenology of the migration of red knots. Generally, from 1994 to 1999, red knots occurred during both March and April, but in 2000 practically none arrived in March. Moreover, in 2004 and 2005, the first red knots were not recorded until May. Hernández (2009, p. 208) concluded that this delayed stopover at Península Valdés was reflected in similar changes at other sites along the West

Atlantic Flyway (e.g., San Antonio Oeste, Delaware Bay). Although the cause is unknown, these observations do suggest an advancement of spring phenology, but we do not know if this shift persisted, increased, or reversed in more recent years. Further, any advancement of spring timing during these years is apparently localized given the late arrivals in Delaware Bay over roughly the same period.

After 2000, increasing proportions of birds arrived late and with low weights at stopover sites in South and North America, suggesting that red knots face additional problems somewhere en route. Indeed, observations from a key Tierra del Fuego wintering area (Río Grande) in 1995, 2000, and 2008 indicated that wintering conditions at this site had deteriorated, as energy intake rates dropped sharply due to smaller prey sizes and human disturbance (Escudero *et al.* 2012, p. 362). Escudero *et al.* (2012, p. 362) suggested declining foraging conditions at Río Grande might offer at least a partial explanation for red knots after 2000 arriving late, and with low weights at stopover sites in South and North America.

We have no information to explain why the spring migration of some red knots wintering in Argentina and Chile apparently shifted later in the mid-2000s, exacerbating the population effects from reduced horseshoe crab egg supplies in Delaware Bay. Escudero *et al.* (2012, p. 362) suggested that problems in one wintering area may be a factor, but the full explanation is unknown. Regardless of the cause, if the trend of later spring migrations continues, it may exacerbate emerging asynchronies with mollusk prey at other stopover areas, since the reproductive window of bivalves and intertidal invertebrates is already shifting earlier in response to warming water temperatures, with further shifts likely in the future, as discussed above.

However, red knots may show at least some adaptive capacity in their migration strategies. For example, from 2000 to 2003, a study of a Tierra del Fuego wintering area (Río Grande) and the first major South American stopover site (San Antonio Oeste) found that red knots took a direct northward flight between the two areas in 2000 and 2001. However, in 2002, birds stopped to feed in intermediate wetlands, leaving Río Grande earlier but arriving later in San Antonio Oeste. In 2003, both early and late patterns were observed. Red knots arriving early at San Antonio Oeste also arrived significantly earlier in Delaware Bay (González *et al.* in IWSG 2003, p. 18). These findings, and those of González *et al.* (2006, p. 115), show some diversity and flexibility of the red knot migration strategies. These characteristics may be an advantage in helping red knots adapt to temporal changes in resource availability along the flyway.

In addition to the emergence of possible asynchronies with food resources, red knots on their northbound migration could face a new threat from changes in the timing of tropical storms. Kossin (2008, p. 1) found an apparent tendency toward more common early- and late-season hurricanes that correlated with warming sea surface temperatures, but the uncertainty in these relationships was high. Although the IPCC (2012, p. 159) concluded that uncertainty around the findings was too high for a meaningful assessment, Kossin (2008, p. 3) found warmer springtime sea surface temperature was associated with a lengthening of both the early and late hurricane season. An increase of 1.8ºF (1 ºC) corresponded to a marked shift of the earliest and latest storms by as much as 20 days. Although the confidence in some of these trends is low, the results are consistent between time periods and suggestive of changes in the distribution of formation dates associated with local climate variability (Kossin 2008, p. 3). Any advancement relative to the current hurricane season (June through November; National Hurricane Center 2012) could introduce a new exposure of red knots to tropical storms during their spring migration, including in spring stopover and staging areas (see Delaware Bay—Storms and Water, above).

Asynchronies—Fall Migration

Preliminary results of efforts to track red knot migration routes using geolocators found that three of four birds likely detoured from normal migration paths to avoid adverse weather during the fall migration. These birds travelled an extra 640 to 1,000 mi (1,030 to 1,600 km) to avoid storms (Niles 2104; Niles *et al*. 2010a, p. 129). The extra flying represents substantial additional energy expenditure, which on some occasions may lead to mortality (Niles *et al*. 2010a, p. 129). The timing of fall migration coincides with hurricane season. As discussed under Climate Change, increasing hurricane intensity is ongoing and expected to continue. Hurricane frequency is not expected to increase globally in the future, but may have increased in the North Atlantic over recent decades. However, predictions about changing storm patterns are associated with “low” to “medium” confidence levels (IPCC 2012, p. 13). Therefore, we are uncertain how or to what extent red knots will be affected by changing storm patterns during fall migration.

Red knots may also face asynchronies with the periods of peak prey abundance in fall, similar to those discussed above for the spring migration. Studying *Calidris canutus islandica* in the Dutch Wadden Sea, van Gils *et al.* (2005b, pp. 126–127) found that gizzards are smallest just following the breeding season because while in the Arctic the birds feed on soft-bodied arthropods. Upon arrival at the fall staging area, gizzards enlarge to their normal nonbreeding size. During their ‘small-gizzard’ phase the birds rely heavily on high-quality prey (e.g., high flesh-to-shell ratios), which are most abundant early in the stopover period when most birds arrive. Birds that arrive late at the staging area might struggle to keep their energy budgets balanced, let alone refuel to gain mass and continue on to the wintering grounds. This work by van Gils *et al.* (2005b, pp. 126–127) shows the importance of timing to food availability during fall migration in *C. canutus*.

The timing of fall migration in shorebirds including red knots is also important to avoid the peak migration of avian predators (see Factor C above) (L. Niles pers. comm. November 19, 2012; Meltofte *et al.* 2007, p. 27; Lank *et al.* 2003, p. 303). Studying Alaska-breeding western sandpipers (*Calidris mauri*), Niehaus and Ydenberg (2006, p. 16) found that peregrine falcons respond more strongly to annual variation in snowmelt than the sandpipers with regard to the timing of the southbound migration. Thus, early snowmelt years may correspond with relatively more dangerous southward migrations for the shorebirds, because in these years the sandpipers may be more likely to encounter falcons. This result suggests that advances in the date of snowmelt caused by climate change may produce species-specific effects on the migratory timing of some bird species, and highlights the need to consider climate change effects in an ecological framework including predator-prey interactions (Niehaus and Ydenberg 2006, p. 16).

Asynchronies—Breeding Grounds

For Arctic-nesting, long-distance migrant shorebirds, studies suggest that a bird's breeding site is endogenously controlled, and all other annual events are shifted temporally to optimize arrival on the breeding grounds (Conklin *et al.* 2010, p. 5). As explained previously, the northbound red knot migration is time-constricted. Birds must arrive on arctic breeding grounds at the right time and with sufficient remaining energy and nutrient stores. In fitness terms, everything else in the annual cycle may be subservient to arrival timing. Knots need to reach the Arctic just as snow is melting, lay their eggs, and hatch them in time for the insect emergence (Piersma *et al.* 2005, p. 270; Clark in Farrell and Martin 1997, p. 23). Like many Arctic-nesting birds, red knots require snow-free, exposed ground on which to nest, making snowmelt a necessary condition for nest initiation. Snowmelt timing is also correlated with the timing of the annual emergence of insect and other arthropod prey (Grabowski *et al.* 2013, p. 1097), which are the primary food source for red knot chicks, and for adults during the breeding season. Modeling results from the ARM suggest that indices of arctic conditions are predictors of the annual survival probability of adult red knots, and have stronger effects on survival than departure weights from Delaware Bay (McGowan *et al*. 2011a, p. 13).

Adverse weather in the Arctic can cause years with little to no productivity for shorebird species. Conditions for breeding are highly variable among sites and regions. The factors most affected by annual variation in weather include whether to breed upon arrival on the breeding grounds, the timing of egg-laying, and the chick growth period (Meltofte *et al.* 2007, p. 7). In much of the Arctic, initiation dates of clutches (the group of eggs laid by one female) are highly correlated with snowmelt dates. In regions and years where extensive snowmelt occurs before or soon after shorebird arrival, the decision to breed and clutch initiation dates both appear to be a function of food availability for females. Once incubation is initiated, adult shorebirds appear fairly resilient to variations in temperature, with nest abandonment generally limited to cases of severe weather when new snow covers the ground. Feeding conditions for chicks are highly influenced by weather, affecting juvenile production (Meltofte *et al.* 2007, p. 7). For a number of shorebird species, productivity has been correlated with climate variables known to affect nesting (in June) or brood-rearing (in July) success in a positive (temperature) or negative (snow depth, wind, precipitation) manner (Meltofte *et al.* 2007, p. 25).

Anticipated climate changes are expected to be particularly pronounced in the Arctic, and extensive and dramatic changes in snow and weather regimes are predicted for most tundra areas (Meltofte *et al.* 2007, p. 11) where red knots breed. (See Factor A—Breeding Habitat, above, for rates and predictions of arctic warming and the eco-regional classification of the red knot's current breeding range.) The rapid rate at which climate is changing in the polar regions will impact natural systems ("high confidence") and may exceed the rate at which some of their components can successfully adapt ("low" to "medium confidence") (Chapter 28 in IPCC 2014, p. 3). Shifts in the timing and magnitude of seasonal biomass production could disrupt matched phenologies in arctic food webs, leading to decreased survival of dependent species ("medium confidence"). If the timing of primary and secondary production is no longer matched to the timing of reproductive periods, survival could be impacted with cascading implications to higher trophic levels. This impact would be exacerbated if shifts in timing occur rapidly ("medium confidence") (Chapter 28 in IPCC 2014, p. 3). There is already evidence that trophic mismatch is affecting arctic and sub-arctic breeding species (Liebezeit *et al.* 2014, p. 2). Phenological responses attributable to warming are apparent in most arctic terrestrial ecosystems ("medium confidence") (Chapter 28 in IPCC 2014, p. 12), and the IPCC notes early warning signs that arctic ecosystems are already experiencing irreversible regime shifts (Summary for Policymakers in IPCC 2014, p. 12).

Despite the high likelihood of ecological disruptions from species' differentially changing phenologies, forecasting the effects of changing arctic weather patterns on shorebirds is associated with high uncertainty. Among arctic breeders, it is not clear that species will have the genetic or phenotypic flexibility to shift their reproductive behavior to track phenological changes in the environment, as the specific factors driving reproductive phenology for many species are poorly understood (Liebezeit *et al.* 2014, p. 2). Under late 20th century climate conditions, studies have found that shorebird reproductive success is closely tied to weather and temperature during the breeding season. However, these findings may tell us little about the effects of climate variables on reproductive rates in the future, over a longer time scale, and with a much larger amplitude of climate change. These effects are likely to involve more fundamental changes to Arctic ecosystems. Although arctic shorebirds are resilient to great interannual variability, we do not know to what extent the birds are able to adapt to the long-term and fast-changing climatic conditions that are predicted to occur in coming decades (Meltofte *et al.* 2007, p. 34), and that are already underway (Summary for Policymakers in IPCC 2014, p. 12).

Breeding Grounds—Insect Prey

Earlier spring thaws and ice melts will likely result in earlier invertebrate hatches because arctic invertebrate emergence is temperature dependent. Long–term field observations and recent experimental warming studies of arctic plots support this hypothesis. If birds are unable to alter migration timing, then arctic nesting shorebirds may have insufficient food resources for their young (Galbraith *et al.* 2014, p. 3; NRC 2011, p. 44).

Schekkerman *et al.* (2003, p. 340) found that growth rates of *Calidris canutus* chicks were strongly correlated with weather-induced and seasonal variation in the availability of invertebrate prey within arctic nesting habitats, underscoring the importance of timing of reproduction so that chicks can make full use of the summer peak in insect abundance. During studies of *C. canutus islandica* at a nesting area in eastern Canada, both adults and juveniles were found to put on large amounts of fat prior to migration, suggesting that they make a long-haul flight out of the Arctic to the first fall stopover site. The period of peak arthropod availability is not only during the peak chick rearing season, but also when many adult shorebirds (principally females that have abandoned broods to the care of the male) are actively accumulating fat and other body stores before departure from the Arctic (Meltofte *et al*. 2007, p. 24).

Tulp and Schekkerman (2008, p. 48) developed models of the relationship between weather and arthropod (i.e., insect) abundance based on 4 recent years, then used the models to project insect abundance backwards in time ("hindcast") based on weather records over a 30-year period. The hindcasted dates of peak arthropod abundance advanced during the study period, occurring 7 days earlier in 2003 than in 1973. The timing of the period during which shorebirds have a reasonable probability of finding enough food to grow has also changed, with the highest probabilities now occurring at earlier dates than in the past. At the same time, the overall length of the period with probabilities of finding enough food has remained unchanged (e.g., same number of days of availability, only sooner). The result is an advancement of the optimal breeding date for breeding birds. To take advantage of the new optimal breeding time, arctic shorebirds must advance the start of breeding, and this change could affect the entire migration schedule (Tulp and Schekkerman 2008, p. 48). If such a change is beyond the adaptive capacity of red knots, this species will likely face increasing asynchronies with its insect prey during the breeding season, thereby affecting reproductive output. Arthropod phenology is strongly influenced by spring and early-summer temperatures, so will likely continue to advance with earlier snowmelt (Grabowski *et al.* 2013). The potential uncoupling of phenology of food resources and breeding events is a major concern for the red knot (COSEWIC 2007, p. 40).

Even when insect abundance is high, energy budgets of breeding red knots may be tight due to high energy expenditure levels. During the incubation phase in the High Arctic, tundra-breeding shorebirds appear to incur among the highest daily energy expenditure levels of any time of the year (Piersma *et al.* 2003b, p. 356). The rates of energy expenditure measured in this region are among the highest reported in the literature, reaching inferred ceilings of sustainable energy turnover rates (Piersma *et al.* 2003b, p. 356). If asynchronous insect emergence requires birds to spend more time foraging, adverse effects to the energy budget would be further exacerbated, possibly impacting survival rates because red knots foraging away from the nest on open tundra expend almost twice as much energy as during nest incubation (Piersma *et al.* 2003b, p. 356).

Although not yet documented for red knots, the links between temperature, prey, and reproductive success have been established in other northern-nesting shorebirds. In one sub-Arctic-breeding shorebird species, Pearce-Higgens *et al*. (2010, p. 12) linked population changes to previous August temperatures through the effect of temperature on the abundance of the species' insect prey. Predictions of annual productivity, based on temperature-mediated reductions in prey abundance, closely match observed bird population trends, and forecasted warming indicates significant likelihood of northward range contraction (e.g., local extinction) (Pearce-Higgens *et al*. 2010, p. 12).

The overall abundance of insects on the breeding grounds may actually increase, though indirect effects of global change on insects (e.g., via their food plants) are difficult to predict (Lindström and Agrell 1999, p. 145). Even if overall insect abundance increases, however, it is likely that the timing of peak abundance has and will continue to shift as a result of climate change (Grabowski *et al.* 2013; Tulp and Schekkerman 2008, p. 48). Any changes in timing and availability of food are likely to have detrimental effects on the success of shorebird chicks (ACIA 2005, p. 418). The best available scientific data indicate that red knots will likely be negatively affected by increased asynchronies between their breeding season and the window of optimal insect abundance. However, we are uncertain how or to what extent red knots may be able to adapt their annual cycle, geographic range, or breeding strategy to cope with this aspect of ecosystem change in the Arctic.

Breeding Grounds—Snowmelt

Trends toward earlier snowmelt dates have been documented in North America in recent decades (IPCC 2007b, p. 264). Over the period 1872 to 2009, snow cover duration decreased in the northern hemisphere, especially at high latitudes, as a result of both later onset and, in particular, earlier snowmelt (Grabowski *et al.* 2013, p. 1097). Because of earlier spring snowmelt, the duration of the snow season in the northern hemisphere has declined by 5.3 days per decade since 1972 (IPCC 2013a, p. 42). The area of Arctic land covered by snow in early summer has shrunk by almost a fifth since 1966 (Astill 2012). Earlier snowmelts in the Arctic from 2020 to 2080 are "very likely" (ACIA 2005, p. 470).

The trends of arctic warming and earlier snowmelt may be influencing the behavior of Arctic-breeding animals, including timing of reproduction, with potential consequences for population viability (Grabowski *et al.* 2013, p. 1097). Many bird species are somewhat flexible in their breeding phenology, especially earlier or delayed nest initiation in response to timing of snowmelt (Grabowski *et al.* 2013, p. 1097). However, the studies discussed below show that, for shorebirds, there are limits to this flexibility in the timing of the breeding season.

Field studies from several breeding sites have shown the sensitivity of red knots to the date of snow melt. At 4 sites in the eastern Canadian Arctic, Smith *et al*. (2010a, p. 292) monitored the arrival of 12 species (including red knot) and found 821 nests over 11 years. Weather was highly variable over the course of the study, and the date of 50 percent snow cover varied by up to 3 weeks among years. In contrast, timing of bird arrival varied by 1 week or less at the sites and was not well predicted by local conditions such as temperature, wind, or snow melt. Timing of breeding was related to the date of 50 percent snow melt, with later snow melt resulting in delayed breeding (Smith *et al*. 2010a, p. 292). Studying another long distance shorebird species nesting in Alaska, Conklin *et al.* (2010, p. 4) found similar results—that the timing of migration seems quite consistent, both at the population and individual level, despite substantial annual variation in the date of snow melt. The findings of these studies suggest that the suite of cues that control the timing of shorebird arrival in the Arctic are not equipped to adjust for annual weather variations that take place on the breeding grounds. The existence of any environmental cues in nonbreeding areas indicating tundra conditions thousands of miles away is improbable (Conklin *et al.* 2010, p. 2). (See Mechanisms and Vulnerabilities, above, regarding aspects of the red knot's life history that are generally understood to limit a species' capacity for phenological adaptation.)

Comparing the periods 1984 to 1986 with 2007 to 2009, Grabowski *et al.* (2013, pp. 1097–1100) looked at the dates of egg laying relative to the dates at which 50 percent of the snow cover had melted for seven bird species, including two *Calidris* shorebirds, at a Low Arctic coastal island in western Canada (outside the red knot's breeding range). Across the sample of years studied, the shorebirds showed a statistically significant tendency to earlier lay date with earlier snowmelt dates. Although six of seven species showed a tendency for advancement of lay date with earlier melt, none was able to fully keep up with the earlier melt, probably because birds on migration cannot track spring conditions on nesting grounds, and because there is interannual variation in food availability when they reach the nesting grounds. Relative to other bird groups included in the study, shorebirds and passerines were most responsive, advancing their lay dates by about 4 to 8 days for 10 days advancement in melt. The species that showed the strongest advancements of lay date were those that rely on arthropods as food for their young. The findings of this study must be interpreted cautiously due to reliance on regional (rather than site-specific) snowmelt data. Nonetheless, these results are consistent with Conklin *et al.* (2010) and Smith *et al.* (2010a) in finding that Arctic-nesting shorebirds show limited ability to time their arrival to fit the timing of spring on the tundra. After arriving in the Arctic, however, these birds show considerable flexibility in subsequently matching laying dates to the timing of tundra spring season and the relative abundance of predators (Grabowski *et al.* 2013, pp. 1100–1103).

Liebezeit *et al.* (2014, pp. 1, 4, 7) found similar results studying one passerine and four shorebird species (including two *Calidris* species) at four Low Arctic tundra study sites on the north coast of Alaska (outside the red knot's breeding range) from 2002 to 2011. All species exhibited advanced dates of clutch initiation (i.e., the start of egg laying) ranging from 0.40 to 0.80 days per year, or 4 to 7 days over 9 years. This result corresponds with other arctic studies (including Grabowski *et al.* 2013) showing an advancement rate typically above 0.50 days per year for long-distance migrants. This is a more rapid breeding advancement than has been reported for birds that breed at lower latitudes, concomitant with the accelerated rate of climate changes at high latitudes. Timing of snowmelt was the most important variable in explaining clutch initiation advancement for four of the five species, confirming previous evidence that shorebird clutch initiation is closely linked to snowmelt. Timing of "green-up" was a much less important explanatory factor, and these authors found no evidence that high predator abundances led to earlier laying dates. These results support previous Arctic studies in finding that climate change in the cryosphere will have a strong impact on nesting phenology, although factors explaining changes in nest phenology are not necessarily uniform across the entire Arctic. These results show some arctic breeding shorebird species are altering their breeding phenology to initiate nesting earlier enabling them to, at least temporarily, avoid the negative consequences of a trophic mismatch. Some evidence indicates increasing temperatures could offer physiological relief from trophic constraints for arctic-breeding shorebirds. In the long-term, however, the potential for a decoupling of trophic phenomena at any number of stages during the breeding season is increased as arctic conditions are rapidly transforming in a changing climate (Liebezeit *et al.* 2014, pp. 1, 10).

In 1999, Morrison *et al.* (2005, p. 455) found that post-arrival body masses of *Calidris canutus islandica* at a breeding site on Ellesmere Island, Canada, were lower than the long-term mean. Many shorebirds were unable to breed, or bred late, due to extensive early-season (June) snow cover. The need to use stored energy reserves for survival or supplementing lower than usual local food resources in that year may have contributed to delayed or failed breeding (Morrison *et al.* 2005, p. 455). At a site on Southampton Island in Canada, late snowmelt and adverse weather conditions, combined with predation, contributed to poor productivity in 2004, and may have also significantly increased mortality of adult rufa red knots. Canadian researchers reported that most Arctic-breeding birds failed to breed successfully in 2004 (Niles *et al.* 2005, p. 4).

As years of late snowmelt have typically had an adverse effect on shorebird breeding, warming trends that result in a reduced frequency of late-melt years may have a short-term benefit to red knots by increasing both survival and productivity (Meltofte *et al.* 2007, p. 7). However, it is unknown how red knots will be affected if snowmelts become substantially earlier than the start of the traditional breeding season, particularly given the already tight time constraints of the northbound migration as discussed above. To better understand the consequences of breeding asynchronies in the Arctic, future studies will need to examine the regional phenological trajectories of both predators and prey, as well as the species-specific degree of phenotypic flexibility (Liebezeit *et al.* 2014, p. 10). It remains to be seen whether shorebirds can continue to respond if snowmelt continues its advancing trend (Grabowski *et al.* 2013, p. 1103) as projected over coming decades (ACIA 2005, p. 470). See Ims and Fuglei 2005 and Lindström and Agrell 1999 for consideration of the complex ways tundra ecosystems may respond to climate change, which further limit our ability to anticipate the red knot's likely response to changes in snowmelt timing.

Breeding Grounds—Snow Depth

Modeling for the ARM suggested that higher snow depth in the breeding grounds on June 10 (about 7 days after peak arrival of red knots) has a strong positive influence on red knot survival probability, regardless of the birds' weights upon departure from Delaware Bay (McGowan *et al.* 2011a, p. 13). In contrast, several studies to date have found a negative effect of snow cover on breeding success (McGowan *et al.* 2011a, p. 13; Meltofte *et al.* 2007, p. 25). These seemingly contradictory findings have many possible explanations: birds may skip breeding in years with heavy snow after arriving in the Arctic and survive at higher rates without the physiological stresses of breeding; snow may determine annual moisture and water in the environment and thereby drive the production of insect prey; red knot survival may be tied to lemming cycles, which are in turn closely linked to snow depth; or the selected weather stations may not be representative of mean snow depth throughout the red knot's breeding range (McGowan *et al.* 2011a, p. 13). Regardless of the explanation, if this strong linkage between snow depth and survival proves correct, arctic warming trends that reduce snow depths would adversely affect red knot survival rates. Such an impact could negate the potential benefits of increased productivity from earlier snowmelt.

Asynchronies—Summary

The red knot's life history strategy makes this species inherently vulnerable to mismatches in timing between its annual cycle and those periods of optimal food and weather conditions upon which it depends (Galbraith *et al.* 2014, p. 7 and Supplement 1; Liebezeit *et al.* 2014, p. 2; Conklin *et al.* 2010, p. 4; Gill *et al.* 2013, p. 1; Hurlbert and Liang 2012, pp. 4–5; McGowan *et al*. 2011a, pp. 2, 16; Smith *et al.* 2011a, p. 575; Meltofte *et al.* 2007, p. 36). For unknown reasons, more red knots arrived late in Delaware Bay in the early 2000s, which is generally accepted as a key causative factor (along with reduced supplies of horseshoe crab eggs) behind red knot population declines that were observed over this same timeframe (Baker *et al*. 2004, p. 878). Thus, the red knot's sensitivity to timing asynchronies has been demonstrated through a population-level response. Both adequate supplies of horseshoe crab eggs and high-quality foraging habitat in Delaware Bay can serve to partially mitigate minor asynchronies at this key stopover site (Smith *et al.* 2011a, pp. 575, 581). However, the factors that caused delays in the spring migrations of red knots from Argentina and Chile are still unknown (Niles *et al*. 2008, p. 2), and we have no information to indicate if this delay will reverse, persist, or intensify in the future.

Superimposed on this existing threat of late arrivals in Delaware Bay are new threats of asynchronies emerging due to climate change (Summary for Policymakers in IPCC 2014, p. 30;

Root *et al.* 2013, pp. 85–88; Hurlbert and Liang 2012, p. 4). Climate change is likely to affect the reproductive timing of horseshoe crabs in Delaware Bay, mollusk prey species at other stopover sites, or both, possibly pushing the peak seasonal availability of food outside of the windows when red knots rely on them (Burrows *et al.* 2011, p. 652; Poloczanska *et al.* 2013, pp. 3–4; Smith *et al.* 2010b, p. 563; van Gils *et al.* 2005a, p. 2615; van Gils *et al.* 2005b, pp. 126–127; Philippart *et al.* 2003, p. 2171). In addition, both field studies and modeling have shown strong links between the red knot's reproductive output and conditions in the Arctic including insect abundance and snow cover (Grabowski *et al.* 2013, p. 1097; McGowan *et al*. 2011a, p. 13; Smith *et al*. 2010a, p. 292; Meltofte *et al.* 2007, pp. 7, 25; Piersma *et al.* 2005, p. 270; Schekkerman *et al.* 2003, p. 340). Climate change may also cause shifts in the period of optimal arctic conditions relative to the time period when red knots currently breed (Grabowski *et al.* 2013, p. 1097; McGowan *et al*. 2011a, p. 13; Smith *et al*. 2010a, p. 292; Tulp and Schekkerman 2008, p. 48; Meltofte *et al.* 2007, pp. 7, 25; Piersma *et al.* 2005, p. 270; Schekkerman *et al.* 2003, p. 340).

The red knot's adaptive capacity to deal with numerous changes in the timing of resource availability across its geographic range is largely unknown (Liebezeit *et al.* 2014, pp. 1, 10; Grabowski *et al.* 2013, p. 1103; Meltofte *et al.* 2007, p. 34). A few examples suggest some flexibility in migration strategies (D. Newstead pers. comm. May 8, 2014; Grabowski *et al.* 2013, pp. 1097, 1100–1103; Smith *et al*. 2010a, p. 292; González *et al.* 2006, p. 115; González *et al.* in IWSG 2003, p. 18). However, available information suggests that the timing of the red knot's annual cycle is controlled at least partly by celestial and endogenous cues (Liebezeit *et al.* 2014, p. 10; Conklin *et al.* 2010, p. 5; Gill *et al.* 2013, p. 1; McGowan *et al.* 2011a, p. 16; Cadée *et al.* 1996, p. 82), while the reproductive seasons of prey species, including horseshoe crabs and mollusks, are largely driven by environmental cues such as water temperature (Smith *et al.* 2010b, p. 563; Philippart *et al.* 2003, p. 2171). These differences between the timing cues of red knots and their prey suggest limitations on the adaptive capacity of red knots to deal with numerous changes in the timing of resource availability across their geographic range.

Based on the combination of documented past impacts and a spectrum of ongoing and emerging threats, we conclude that asynchronies (mismatches between the timing of the red knot's annual cycles and the periods of favorable food and weather upon which it depends) are likely to cause deleterious subspecies-level effects.

Factor E—Human Disturbance

In some wintering and stopover areas, red knots and recreational users (e.g., pedestrians, ORVs, dog walkers, boaters) are concentrated on the same beaches (Niles *et al.* 2008, pp. 105–107; Tarr 2008, p. 134). Recreational activities affect red knots both directly and indirectly. These activities can cause habitat damage (Schlacher and Thompson 2008, p. 234; Anders and Leatherman 1987, p. 183), cause shorebirds to abandon otherwise preferred habitats, negatively affect the birds' energy balances, and reduce the amount of available prey (see Factor E—Reduced Food Availability). Effects to red knots from vehicle and pedestrian disturbance can also occur during construction of shoreline stabilization projects including beach nourishment (see Factor A—U.S. Shoreline Stabilization and Coastal Development). Red knots can also be disturbed by motorized and nonmotorized boats, fishing, kite surfing, aircraft, and research activities (K. Kalasz pers. comm. November 17, 2011; Niles *et al.* 2008, p. 106; Peters and Otis

2007, p. 196; Harrington 2005b, pp. 14–15; 19–21; Meyer *et al.* 1999, pp. 16–17; Burger 1986, p. 124) and by beach raking (also called grooming or cleaning, see Factor A—Beach Cleaning). In Delaware Bay, red knots could also potentially be disturbed by hand-harvest of horseshoe crabs (see Factor E—Reduced Food Availability) during the spring migration stopover period, but under the current management of this fishery State waters from New Jersey to coastal Virginia are closed to horseshoe crab harvest and landing from January 1 to June 7 each year (ASMFC 2012a, p. 4); thus, disturbance from horseshoe crab harvest is no longer occurring. Active management can be effective at reducing and minimizing the adverse effects of recreational disturbance (Burger and Niles 2013a, p. 20; Forys 2011, entire; Burger *et al.* 2004, entire), but such management is not occurring throughout the red knot's range. Because visitor density is strongly correlated with access points on recreational beaches (Tratalos *et al.* 2013, p. 447), management of access points is a key consideration for minimizing effects to shorebirds.

Disturbance—Timing and Extent

Although the timing, frequency, and duration of human and dog presence throughout the red knot's U.S. range are not fully known, periods of recreational use tend to coincide with the knot's spring and fall migration periods (WHSRN 2012; Maddock *et al*. 2009, entire; Mizrahi 2002, p. 2; Johnson and Baldassarre 1988, p. 220; Burger 1986, p. 124). Burger (1986, p. 128) found that red knots and other shorebirds at two sites in New Jersey reacted more strongly to disturbance (i.e., flew away from the beach where they were foraging or roosting) during peak migration periods (May and August) than in other months.

Human disturbance within otherwise suitable red knot migration and winter foraging or roosting areas was reported by biologists as negatively affecting red knots in Massachusetts, Virginia, North Carolina, South Carolina, Georgia, and Florida (USFWS 2011b, p. 29). Some disturbance issues also remain in New Jersey (both Delaware Bay and the Atlantic coast) despite ongoing, and largely successful, management efforts since 2003 (NJDEP 2013; USFWS 2011b, p. 29; Niles *et al.* 2008, pp. 105–106). Delaware also has a management program in place to limit disturbance (Kalasz 2008, pp. 36–38), and has implemented measures to reduce disturbance at Mispillion Harbor, a key Delaware Bay foraging site (DDNREC 2013). Both New Jersey and Delaware have established shorebird viewing areas on Delaware Bay, to minimize disturbance and provide public education (DDNREC 2013; NJDEP 2013). In Florida, the most immediate and tangible threat to migrating and wintering red knots is apparently chronic disturbance (Niles *et al.* 2008, p. 106; Niles *et al.* 2006, entire), which may be affecting the ability of birds to maintain adequate weights in some areas (Niles 2009, p. 8).

In many areas, migration and wintering habitat for the piping plover overlaps considerably with red knot habitats. Because the two species use similar habitats in the Southeast, and both are documented to be affected by disturbance, we can infer the extent of potential human disturbance to red knots from piping plover data in this region. Based on a preliminary review of disturbance in piping plover wintering habitats from North Carolina to Texas, pedestrians and dogs are widespread on beaches in this region (USFWS 2009, p. 46). LeDee *et al.* (2010, pp. 343–344) surveyed land managers of designated wintering piping plover critical habitat sites across seven southern States and documented the extent of beach access and recreation. All but 4 of the 43 reporting sites owned or managed by Federal, State, and local governmental agencies or by nongovernmental organizations allowed public beach access year-round (88 percent of the sites). At the sites allowing public access, 62 percent of site managers reported more than 10,000 visitors from September to March, and 31 percent reported more than 100,000 visitors in this period. However, more than 80 percent of the sites allowing public access did not allow vehicles on the beach, and half did not allow dogs during the winter season (as cited in USFWS 2012a, p. 35). Most North Carolina sites used by the highest numbers of red knots are protected through conservation management by Federal or State agencies, including signage and, in some areas, seasonal closures for other species such as piping plover, American oystercatcher (*Haematopus palliatus*), colonial waterbirds, and nesting sea turtles (NCWRC 2013). Much of Louisiana's red knot habitat is not readily accessed, minimizing many anthropogenic threats like disturbance (LDFW 2013).

Disturbance of red knots has also been reported from Canada. In the Province of Quebec, specifically on the Magdalen Islands, feeding and resting red knots are frequently disturbed by human activities such as clam harvesting and farming, kite surfing, and seal rookery observation (USFWS 2011b, p. 29). With the increasing popularity of ecotourism, more visitors from around the world come to the shores of the Bay of Fundy in Canada, but existing infrastructure is insufficient to minimize disturbance to roosting shorebirds during high-tide periods. In addition, access to the shoreline is increasing due to ORV use (WHSRN 2012).

Areas of South America also have documented red knot disturbance. Goldfeder and Blanco in Boere *et al.* (2006, p. 193) cited tourism as a threat to the red knot in Argentina. In Tierra del Fuego, wintering red knots are often disturbed around Río Grande City, Argentina, by ORVs, motorcycles, walkers, runners, fishermen, and dogs (Niles *et al.* 2008, p. 107; COSEWIC 2007, p. 36). The City of Río Grande has grown extensively towards the sea and river margins. Escudero *et al.* (2012, p. 358) reported that pedestrians, ORVs, and unleashed dogs on the gravel beach during high tide caused red knots to fly from one spot to another or to move farther away from feeding areas. During outgoing tides, as prime intertidal foraging habitats became exposed, red knots were disturbed and were flushed continuously by walkers, ORVs, and dogs (Escudero *et al.* 2012, p. 358).

In Patagonian Argentina, disturbance of migrating red knots has been reported from shorebird reserve areas at Río Gallegos, Península Valdés, Bahía San Antonio (San Antonio Oeste), and Bahía Samborombón (WHSRN 2012; Niles *et al.* 2008, p. 107). Coastal urban growth at Río Gallegos has increased disturbances to shorebirds, especially during high tide when they gather in a limited number of spots very close to shore. Dogs and people frequently interrupt the birds' resting and feeding activities. Various recreational activities, including boating, sport fishing, hiking, and dog walking, take place at urban sites near the coast and on the periphery of the city. These seasonal activities are concentrated in the austral spring and summer (WHSRN 2012), when red knots are present.

Both shorebirds and people are attracted to the pristine beaches in Bahía San Antonio, Argentina. For example, Las Grutas Beach draws 300,000 tourists every summer, a number that has increased 20 percent per year over the past decade, and the timing of which corresponds with the red knot's wintering use. New access points, buildings, and tourist amusement facilities are being constructed along the beach. Lack of planning for this rapid expansion has resulted in uncontrolled tourist disturbance of crucial roosting and feeding areas for migratory shorebirds, including red knots (WHSRN 2012).

Red knots on the coast of Rio Grande do Sul in southern Brazil make extensive use of ocean beaches for foraging and roosting (Harrington *et al.* 1986, p. 49). These Atlantic beaches are heavily used both for recreation and as travel corridors by commercial fishermen, resulting in frequent disturbance to red knots and other shorebirds at a key northward migration staging area (B. Harrington pers. comm. November 14, 2013). On the north coast of Brazil, in the Maranhão region, there are some disturbance issues, but no major impediments to foraging and red knots typically depart in good condition (L. Niles pers. comm. November 19, 2013). Along the coast of French Guiana, human impacts are very limited, occurring mainly in the cities and associated harbors of Cayenne, Rémire-Montjoly, and Kourou. Red knot habitats occur near Cayenne and Kourou. One site used by red knots for feeding and roosting, (Pointe des Roches on the left bank of the Kourou estuary), is subject to human disturbance as it is on the city beach front at Kourou (CSRPN 2013). Disturbance could be an issue at roost sites on Chiloè Island on the Pacific coast of Chile (B. Andres pers. comm. July 21, 2014).

Management efforts have begun to mitigate disturbance at some South American sites. Campaigns to build alternative ORV trails away from shorebird areas, and to raise public awareness, have helped reduce disturbance in Tierra del Fuego, Río Gallegos, and Bahía San Antonio (American Bird Conservancy 2012a, p. 5). The impact of human disturbance was successfully controlled at roosting and feeding sites at Los Alamos near Las Grutas (Bahía San Antonio) by "environmental rangers" charged with protecting shorebird roosting sites and providing environmental education (WHSRN 2012). However, other key shorebird sites do not yet have any protection.

Disturbance—Precluded Use of Preferred Habitats

Where shorebirds are habitually disturbed, they may be pushed out of otherwise preferred roosting and foraging habitats (Colwell *et al.* 2003, p. 492; Lafferty 2001a, p. 322; Luís *et al.* 2001, p. 72; Burton *et al.* 1996, pp. 193, 197–200; Burger *et al.* 1995, p. 62). Roosting knots are particularly vulnerable to disturbance because birds tend to concentrate in a few small areas during high tides, and availability of suitable roosting habitats is already constrained by predation pressures and energetic costs such as traveling between roosting and foraging areas (L. Niles pers. comm. November 19 and 20, 2012; Kalasz 2008, p. 9; Rogers *et al.* 2006a, p. 563; Colwell *et al.* 2003, p. 491; Rogers 2003, p. 74).

Exclusion of shorebirds from preferred habitats due to disturbance has been noted throughout the red knot's nonbreeding range. For example, Pfister *et al.* (1992, p. 115) found sharper declines in red knot abundance at a disturbed site in Massachusetts than at comparable but less disturbed areas. On the Atlantic coast of New Jersey, findings by Mizrahi (2002, p. 2) generally suggest a negative relationship between human and shorebird densities; specifically, sites that allowed swimming had the greatest densities of people and the fewest shorebirds.

At two sites on the Atlantic coast of New Jersey, Burger and Niles (2013a, pp. 22, 24) found that disturbed shorebird flocks often did not return to the same place or even general location along the beach once they were disturbed, with return rates at one site of only 8 percent for monospecific red knot flocks. Even when flocks returned, not all shorebirds did so, with half or less of the birds returning after a disturbance (Burger and Niles 2013a, p. 23). At one of these New Jersey study sites, Burger and Niles (2013b, p. 657) found that spatial use by shorebirds, especially red knots, depended upon whether the beach was open or closed. Of the species in this study, red knot behavior was most affected by beach closure. Knots spread out over the entire beach when it was closed, and concentrated behind the fencing when the beach was open (Burger and Niles 2013b, p. 657). When the beach was open, knots concentrated on the beach segments that were the greatest distance from where people could enter the beach. For all species, there was a significant difference in the mean flock size depending upon whether the beach was open or closed (Burger and Niles 2013b, p. 665).

In Delaware Bay, Karpanty *et al.* (2006, p. 1707) found that potential disturbance reduced the probability of finding red knots on a given beach, although the effect of disturbance was secondary to the influence of prey resources. Also in Delaware Bay, Harrington (2005b, p. 16) found that shorebird numbers were lower in areas of higher disturbance. In Florida, sanderlings seemed to concentrate where there were the fewest people (Burger and Gochfeld 1991, p. 263). From 1979 to 2007, the mean abundance of red knots on Mustang Island, Texas decreased 54 percent, while the mean number of people on the beach increased fivefold (Foster *et al.* 2009, p. 1079). In 2008, Escudero *et al.* (2012, p. 358) found that human disturbance pushed red knots off prime foraging areas near Río Grande in Argentinean Tierra del Fuego, and that disturbance was the main factor affecting roost site selection.

Although not specific to red knot, Forgues (2010, p. ii) found the abundance of shorebirds declined with increased ORV frequency, as did the number and size of roosts. Study sites with high ORV activity and relatively high invertebrate abundance suggest that shorebirds may be excluded from prime food sources due to disturbance from ORV activity itself (Forgues 2010, p. 7). Tarr (2008, p. 133) found that disturbance from ORVs decreased shorebird abundance and altered shorebird habitat use. In experimental plots, shorebirds decreased their use of the wet sand microhabitat and increased their use of the swash zone in response to vehicle disturbance (Tarr 2008, p. 144).

Disturbance—Effects to Energy Budgets

Disturbance of shorebirds can cause behavioral changes resulting in less time roosting or foraging, shifts in feeding times, decreased food intake, and more time and energy spent in alert postures or fleeing from disturbances (Defeo *et al.* 2009, p. 3; Tarr 2008, pp. 12, 134; Burger *et al*. 2007, p. 1164; Thomas *et al*. 2003, p. 67; Lafferty 2001a, p. 315; Lafferty 2001b, p. 1949; Elliott and Teas 1996, pp. 6–9; Burger 1994, p. 695; Burger 1991, p. 39; Johnson and Baldassarre 1988, p. 220). By reducing time spent foraging and increasing energy spent fleeing, disturbance may hinder red knots' ability to recuperate from migratory flights, maintain adequate weights, or build fat reserves for the next phase of the annual cycle (Harrington 2005b, pp. 1–2; Clark in Farrell and Martin 1997, p. 24; Burger *et al.* 1995, p. 62). In addition, stress such as frequent disturbance can cause red knots to stop molting before the process is complete (Niles 2010b), which could potentially interfere with the birds' completion of the next phase of their annual cycle.

Although population-level impacts cannot be concluded from species' differing behavioral responses to disturbance (Stillman *et al.* 2007, p. 73; Gill *et al.* 2001, p. 265), behavior-based models can be used to relate the number and magnitude of human disturbances to impacts on the fitness of individual birds (Goss-Custard *et al.* 2006, p. 88; West *et al.* 2002, p. 319). When the time and energy costs arising from disturbance were included, modeling by West *et al.* (2002, p. 319) showed that disturbance could be more damaging than permanent habitat loss. Modeling by Goss-Custard *et al.* (2006, p. 88) was used to establish critical thresholds for the frequency with which shorebirds can be disturbed before they die of starvation. There is evidence from modeling that, under some conditions, sanderlings could spend more energy responding to human disturbances than they were able to accrue in their daily foraging; disturbance can be energetically costly to shorebirds at a migration staging area (B. Harrington pers. comm. November 14, 2013). Birds can tolerate more disturbance before their fitness levels are reduced when feeding conditions are favorable (e.g., abundant prey, mild weather) (Niles *et al.* 2008, p. 105; Goss-Custard *et al.* 2006, p. 88).

At two Atlantic coast sites in New Jersey, Burger and Niles (2013a, p. 23) found that about 70 percent of shorebird flocks with red knots flew when disturbed by people, vehicles, or dogs, whether the flocks were monospecific or contained other species as well. In two New Jersey bays, Burger (1986, p. 125) found that 70 percent of shorebirds, including red knots, flew when disturbed, including 25 (Raritan Bay) to 48 (Delaware Bay) percent that flew away and did not return (see Precluded Use of Preferred Habitat, above). Birds in smaller flocks tended to be more easily disturbed than those in larger flocks. Explanatory variables for differences in response rate included date, duration of disturbance, distance between the disturbance and the birds, and the number of people involved in the disturbance (Burger 1986, pp. 126–127). On some Delaware Bay beaches, the percent of shorebirds (including red knots) that flew away and did not return in response to disturbance increased between 1982 and 2002 (Burger *et al.* 2004, p. 286). Also from Delaware Bay, data presented by Harrington (2005b, p. 19) suggest that shorebird foraging rates are affected by the presence of people. Along with reduced size of prey items, disturbance was a key factor explaining sharp declines in red knot food intake rates at Río Grande, Argentina, on Tierra del Fuego (Escudero *et al.* 2012, p. 362). Comparing conditions in 2008 with earlier studies, total red knot feeding time was 0.5 hour shorter due to continuous disturbance and flushing of the birds by people, dogs, and ORVs during prime feeding time just after high tide (Escudero *et al.* 2012, pp. 358, 362).

Although not specific to red knots, other shorebird studies support the conclusion that disturbance can impact energy budgets. In Florida, sanderlings ran or flew to new spots when people moved rapidly toward them, or when large groups moved along the beach no matter how slow the movement. The number of people on the beach contributed significantly to explaining variations in the amount of time sanderlings spent feeding, and active feeding time decreased from 1986 to 1990 (Burger and Gochfeld 1991, p. 263). At one California beach, Lafferty (2001b, p. 1949) found that more than 70 percent of birds flew when disturbed, and species that forage lower on the beach were disproportionally affected by disturbance because contact with people was more frequent. Although California is outside the range of the red knot, this finding would apply to species such as red knots that forage in the intertidal zone. Studying another *Calidris canutus* subspecies in Australia, Rogers *et al.* (2006b, p. 233) found that energy expenditure over a tidal cycle was sensitive to the amount of disturbance, and a relatively small increase in disturbance can result in a substantial increase in energy expenditure. Shorebirds may be able to compensate for these costs to some extent by extending their food intake, but only to a degree, and such compensation is dependent upon the availability of adequate food resources. The energetic costs of disturbance are greatest for heavy birds, such as just before departure on a migratory flight (Rogers *et al.* 2006b, p. 233).

Both modeling (West *et al.* 2002, p. 319) and empirical studies (Burger 1986, pp. 126–127) suggest that numerous small disturbances are generally more costly than fewer, larger disturbances. Burger *et al.* (2007, p. 1164) found that repeated disturbances to red knots and other shorebirds may have the effect of increasing interference competition for foraging space by giving a competitive advantage to gull species, which return to foraging more quickly than shorebirds following a response to vehicles, people, or dogs.

Tarr (2008, p. 133) found that vehicle disturbance decreased the amount of time that sanderlings spent roosting and resting. Forgues 2010 (pp. 39, 55) found that shorebirds spent significantly less time foraging and more time resting at sites with ORVs, and suggested that the increased amount of time spent resting may be a compensation method for energy lost from decreased foraging.

Shorebirds are more likely to be flushed by dogs than by people (Thomas *et al*. 2003, p. 67; Lafferty 2001a, p. 318; Lord *et al*. 2001, p. 233), and birds react to dogs from greater distances than to people (Lafferty 2001a, p. 319; Lafferty 2001b, pp. 1950, 1956). Pedestrians walking with dogs often go through flocks of foraging and roosting shorebirds, and unleashed dogs often chase the birds and can kill them (Lafferty 2001b, p. 1955; Burger 1986, p. 128). Burger *et al.* (2007, p. 1162) found that foraging shorebirds in migratory habitat do not return to the beach following a disturbance by a dog, and Burger *et al.* 2004 (pp. 286–287) found that disturbance by dogs was increasing in Delaware Bay even as management efforts have been successful at reducing other types of disturbances.

Disturbance—Summary

Red knots are exposed to disturbance from recreational and other human activities throughout their nonbreeding range (B. Andres pers. comm. July 21, 2014; B. Harrington pers. comm. November 14, 2013; CSRPN 2013; Escudero *et al.* 2012, p. 358; WHSRN 2012; USFWS 2011b, p. 29; Niles *et al.* 2008, p. 107). Excessive disturbance has been shown to preclude red knot use of otherwise preferred habitats (Burger and Niles 2013a, p. 23; Burger and Niles 2013b, p. 257; Escudero *et al.* 2012, p. 358; Foster *et al.* 2009, p. 1079; Karpanty *et al.* 2006, p. 1707; Harrington 2005b, p. 16) and can impact shorebird energy budgets (Burger and Niles 2013a, p. 23; Escudero *et al.* 2012, p. 362; Harrington 2005b, p. 19; Burger 1986, p. 125). Both of these effects are likely to exacerbate other threats to the red knot, such as habitat loss, reduced food availability, asynchronies in the annual cycle, and competition with gulls (see Cumulative Effects, below).

Factor E—Competition with Gulls

Gulls foraging on the beaches of Delaware Bay during the red knot's spring stopover period may directly or indirectly compete with shorebirds for horseshoe crab eggs. Botton

(1984, p. 209) noted that, in addition to shorebirds, large populations of laughing gulls (*Larus atricilla*) were predominant on New Jersey's horseshoe crab spawning beaches along Delaware Bay. Gull breeding colonies in Delaware are not located as close to the bayshore beaches as in New Jersey. However, immature, large-bodied gulls such as greater black-backed gull and herring gull, as well as some laughing gulls, most likely from New Jersey breeding colonies, do congregate on the Delaware shore during the spring, especially at Mispillion Harbor (Niles *et al.* 2008, p. 107).

Aerial surveys of breeding gull species on the Atlantic coast of New Jersey from 1976 to 2007 show that herring and greater black-backed gull populations were relatively stable. Greater black-backed gulls showed a slight increase in 2001 that had subsided by 2004. Laughing gull populations grew steadily from 1976 (fewer than 20,000 birds) to 1989 (nearly 60,000 birds). Following a dip in 1995, laughing gull numbers spiked in 2001 to nearly 80,000. From 2004 to 2007, laughing gull numbers returned to approximately the same levels that predominated in the 1980s (50,000 to 60,000 birds) (Dey *et al.* 2011b, p. 24).

From 1992 to 2002, the number of gulls recorded in single-day counts on Delaware Bay beaches in New Jersey ranged from 10,000 to 23,000 (Niles *et al.* 2008, p. 107). To allow for comparisons, gull counts on Delaware Bay were performed in spring 1990 to 1992 and again in 2002 using the same methodology (Sutton and Dowdell 2002, p. 3). Despite the increasing breeding populations documented by the aerial survey of New Jersey's nearby Atlantic coast, gull numbers on Delaware Bay beaches were significantly lower in 2002 than they were between 1990 and 1992. The highest laughing gull count in 2002 was only a third of the highest count of the 1990 to 1992 period. When comparing the average of the four 1990s counts to the average of the four 2002 counts, laughing gulls using Delaware Bay beaches declined by 61 percent (Sutton and Dowdell 2002, p. 5). Decreased gull usage of Delaware Bay, despite growing regional gull populations, may suggest that gulls were responding to reduced availability of horseshoe crab eggs by 2002 (Sutton and Dowdell 2002, p. 6).

Burger *et al.* (1979, p. 462) found that intraspecific (between members of the same species) aggressive interactions of shorebirds were more common than interspecific (between members of different species) interactions. Negative interactions between red knots and laughing gulls that resulted in disruption of knot behavior were no more prevalent than interactions with other shorebird species. However, larger-bodied species (like gulls) tended to successfully defend areas against smaller species. Total aggressive interactions increased as the density of birds increased in favored habitats, which indicated some competition for food resources (Burger *et al.* 1979, p. 462).

Sullivan (1986, pp. 376–377) found that aggression in ruddy turnstones increased as experimentally manipulated food resources (horseshoe crab eggs) changed from an even distribution to a more patchy distribution. Horseshoe crab eggs are typically patchy on Delaware Bay beaches, as evidenced by the very high variability of egg densities within and between sites (ASMFC 2012d, p. 11). The ruddy turnstones' decisions to defend food patches were likely driven by the energetic cost of locating new patches (Sullivan 1986, pp. 376–377), suggesting that aggression may increase as food availability decreases. Botton *et al*. (1994, p. 609) noted that flocks of shorebirds appeared to be deterred from landing on beaches when large flocks of gulls were present. When dense, mixed flocks of gulls and shorebirds were observed, gulls monopolized the waterline, limiting shorebirds to drier sand farther up the beach (Botton *et al.* 1994, p. 609).

Following up on earlier studies, Burger (undated, p. 9) studied foraging behavior in shorebirds and gulls on the New Jersey side of Delaware Bay in spring 2002 to determine if interference competition existed between shorebirds and gulls. For red knots, the time devoted to foraging when gulls were present was significantly less than when a nearest neighbor was any shorebird. Red knots spent more time being vigilant when their nearest neighbors were gulls rather than other shorebirds. Similarly, red knots engaged in more aggression when gulls were nearest neighbors, although they usually lost these encounters (Burger undated, p. 10; USFWS 2003, p. 42). The increased vigilance of red knots when feeding near gulls comes at the detriment of time spent feeding (Niles *et al.* 2008, p. 107), and red knot foraging efficiency is adversely affected by the mere presence of gulls. Hernandez (2005, p. 80) found that the foraging efficiency of knots feeding on horseshoe crab eggs decreased by as much as 40 percent when feeding close to a gull. As described under Background—Species Information—Migration and Wintering Food, above, red knots are present in Delaware Bay for a short time to replenish energy to complete migration to their arctic breeding grounds. Excessive competition from gulls that decreases energy intake rates would affect the ability of red knots to gain sufficient weight for the final leg of migration.

Despite the observed competitive behaviors between gulls and red knots, Karpanty *et al.* (2011, p. 992) did not observe red knots to be excluded from foraging by aggressive interactions with other red knots, other shorebirds, or gull species in experimental sections of beach in 2004 and 2005. These authors did observe knots foraging in plots with high egg densities and knots foraging throughout the tidal cycle in all microhabitats. Thus, red knots did not appear to be substantially affected by interspecific or intraspecific interference competition during this study.

Burger *et al.* (2007, p. 1162) found that gulls are more tolerant of human disturbance than are shorebirds. When disturbed by humans, gull numbers returned to pre-disturbance levels within 5 minutes. Even after 10 minutes, shorebird numbers failed to reach predisturbance levels. Repeated disturbances to red knots and other shorebirds may have the effect of increasing interference competition for foraging space by giving a competitive advantage to gull species, which return to foraging more quickly than shorebirds following a flight response to vehicles, people, or dogs (Burger *et al.* 2007, p. 1164). The size and aggression of gulls, coupled with their greater tolerance of human disturbance, give gulls a competitive advantage over shorebirds in prime feeding areas (Niles *et al.* 2008, p. 107).

Reduction of available horseshoe crab eggs or consolidation of spawning horseshoe crabs onto fewer beaches can increase interference competition among egg foragers. Karpanty *et al.* (2006, p. 1707) found a positive relationship between laughing gull numbers and red knot presence (i.e., more laughing gulls were present when red knots were also present), concluding that this correlation was likely due to the use by both bird species of the sandy beach areas with the highest densities of horseshoe crab eggs for foraging. Competition for horseshoe crab eggs increases with reduced egg availability, and the ability of shorebirds to compete with gulls for food decreases as shorebird flock size decreases (Breese 2010, p. 3; Niles *et al.* 2005, p. 4).

Competition between shorebirds and laughing gulls for horseshoe crab eggs increased in the 2000s as the decline in the horseshoe crab population concentrated spawning in a few favored areas (e.g., Mispillion Harbor, Delaware; Reeds Beach, New Jersey). These "hot spots" of horseshoe crab eggs concentrated foraging shorebirds and gulls, increasing competition for limited resources. Hot spots were known to shift in some years when severe wind and rough surf favored spawning in sheltered areas (e.g., creek mouths) (Kalasz *et al*. 2010, pp. 11–12). A reduced crab population, the contraction of spawning both spatially and temporally, and storm events that concentrated spawning into protected creek mouths exacerbated competition for available eggs in certain years (Dey *et al.* 2011b, p. 9). Delaware's shorebird conservation plan calls for control of gull populations if they exceed a natural size and negatively impact migrating birds (Kalasz 2008, p. 39).

In summary, competition with gulls can exacerbate food shortages in Delaware Bay (Dey *et al.* 2011b, p. 9; Kalasz *et al*. 2010, pp. 11–12; Niles *et al.* 2008, p. 107; Burger *et al.* 2007, p. 1164; Hernandez 2005, p. 80). Despite the growth of gull populations in southern New Jersey, numbers of gulls using Delaware Bay in spring decreased considerably from the early 1990s to the early 2000s (Dey *et al.* 2011b, p. 24; Sutton and Dowdell 2002, p. 5). Because more recent comparable survey data are not available, we cannot surmise if there are any recent trends in competition pressures, nor can we project a trend into the future. We conclude that gull competition was not a driving cause of red knot population declines in the 2000s, but was likely one of several factors (along with predation, storms, late arrivals of migrants, and human disturbance) that likely exacerbated the effects of reduced horseshoe crab egg availability.

Gull competition has not been reported as a threat to red knots outside of Delaware Bay (e.g., S. Koch pers. comm. March 5, 2013; K. Iaquinto pers. comm. February 22, 2013), but is likely to exacerbate other threats throughout the knot's range due to gulls' larger body sizes, high aggression (Burger undated, p. 10; Niles *et al.* 2008, p. 107; Burger *et al.* 1979, p. 462), tolerance of human disturbance (Burger *et al.* 2007, p. 1162), and generally stable or increasing populations. However, outside of Delaware Bay, there is typically less overlap between the diets of red knots (see Wintering and Migration Food) and most gulls species (generalist feeders). We expect the effects of gulls to be most pronounced where red knots become restricted to reduced areas of foraging habitat, which can occur as a result of reduced food resources, human disturbance or predation that excludes knots from quality habitats, or outright habitat loss (see Cumulative Effects below).

Factor E—Harmful Algal Blooms (HABs)

A harmful algal bloom (HAB) is the proliferation of a toxic or nuisance algal species (which can be microscopic or macroscopic, such as seaweed) that negatively affects natural resources or humans (Florida Fish and Wildlife Conservation Commission (FFWCC) 2011). While most species of microscopic marine life are harmless, there are a few dozen species that create toxins given the right conditions. During a "bloom" event, even nontoxic species can disrupt ecosystems through sheer overabundance (Woods Hole Oceanographic Institute (Woods Hole) 2012). The primary groups of microscopic species that form HABs are flagellates (including dinoflagellates), diatoms, and blue-green algae (which are actually cyanobacteria, a group of bacteria, rather than true algae). Of the approximately 85 HAB-forming species currently documented, almost all of them are plant-like microalgae that require light and carbon dioxide to produce their own food using chlorophyll (FFWCC 2011). Blooms can appear green, brown, or red-orange, or may be colorless, depending upon the species blooming and environmental conditions. Although HABs are popularly called “red tides,” this name can be misleading, as it includes many blooms that discolor the water but cause no harm, while also excluding blooms of highly toxic cells that cause problems at low (and essentially invisible) concentrations (Woods Hole 2012). Here, we use the term “red tide” to refer only to blooms of the dinoflagellate *Karenia brevis*.

HABs—Impacts to Shorebirds

Large die-offs of fish, mammals, and birds can be caused by HABs. Wildlife mortality associated with HABs can be caused by direct exposure to toxins, indirect exposure to toxins (i.e., as the toxins accumulate in the food web), or through ecosystem impacts (e.g., reductions in light penetration or oxygen levels in the water, alteration of food webs due to fish kills or other mass mortalities) (Woods Hole 2012; Anderson 2007, p. 5; FAO 2004, p. 1). Wildlife can be exposed to algal toxins through aerosol (airborne) transport or via consumption of toxic prey (FFWCC 2011; Steidinger *et al.* 1999, p. 6). Exposure of wildlife to algal toxins may continue for weeks after an HAB subsides, as toxins move through the food web (Abbott *et al*. 2009, p. 4).

Animals exposed to algal toxins through their diets may die or display impaired feeding and immune function, avoidance behavior, physiological dysfunction, reduced growth and reproduction, or pathological effects (Woods Hole 2012). A poorly defined but potentially significant concern relates to sublethal, chronic impacts from toxic HABs that can affect the structure and function of ecosystems (Anderson 2007, p. 4). Chronic toxin exposure may have long-term consequences affecting the sustainability or recovery of natural populations at higher trophic levels (e.g., species that feed higher in the food web). Ecosystem-level effects from toxic algae may be more pervasive than yet documented by science, affecting multiple trophic levels, depending on the ecosystem and the toxin involved (Anderson 2007, pp. 4–5).

When a toxic algal bloom occurs, benthic macrofaunal species can thus play a critical role in the transfer of toxins to higher trophic levels, and toxin uptake in suspension-feeding bivalves has been extensively studied (Bricelj *et al.* 2012, p. 30). For both humans and shorebirds, shellfish are a key route of exposure to algal toxins. When toxic algae are filtered from the water as food by shellfish, their toxins accumulate in those shellfish to levels that can be lethal to humans or other animals that eat the shellfish (Anderson 2007, p. 4). Several shellfish poisoning syndromes have been identified according to their symptoms. Those shellfish poisoning syndromes that occur prominently within the range of the red knot include Amnesic Shellfish Poisoning (ASP) (occurring in Atlantic Canada, caused by *Pseudo-nitzchia* spp.); Neurotoxic Shellfish Poisoning (NSP, also called “red tide”) (occurring on the U.S. coast from Texas to North Carolina, caused by *Karenia brevis* and other species); and Paralytic Shellfish Poisoning (PSP) (occurring in Atlantic Canada, the U.S. coast in New England, Argentina, and Tierra del Fuego, caused by *Alexandrium* spp. and others) (Woods Hole 2012; FAO 2004, p. 44). The highest levels of PSP toxins have been recorded in shellfish from Tierra del Fuego (International Atomic Energy Agency 2004), and high levels can persist in mollusks for months following a PSP bloom (FAO 2004, p. 44). In Florida, the St. Johns, St. Lucie, and

Caloosahatchee Rivers and estuaries have also been affected by persistent HABs of cyanobacteria (FFWCC 2011).

Algal toxins may be a direct cause of death in seabirds and shorebirds via an acute or lethal exposure, or birds can be exposed to chronic, sublethal levels of a toxin over the course of an extended bloom. Sub-acute doses may contribute to mortality due to an impaired ability to forage productively, disrupted migration behavior, reduced nesting success, or increased vulnerability to predation, dehydration, disease, or injury (VanDeventer 2007, p. 1). It is commonly believed that the primary risk to shorebirds during an HAB is via contamination of shellfish and other invertebrates that constitute their normal diet. Coquina clams (*Donax variabilis*) and other items that shorebirds feed upon can accumulate marine toxins during HABs and may pose a risk to foraging shorebirds. In addition to consuming toxins via their normal prey items, shorebirds have been observed consuming dead fish killed by HABs (VanDeventer 2007, p. 11). VanDeventer *et al.* (2011, p. 31) observed shorebirds, including sanderlings and ruddy turnstones, scavenging fish killed during a 2005 red tide along the central west coast of Florida. Brevetoxins (discussed below) were found both in the dead fish and in the livers of dead shorebirds that were collected from beaches and rehabilitation centers (VanDeventer *et al.* 2011, p. 31). Although scavenging has not been documented in red knots, clams and other red knot prey species are among the organisms that accumulate algal toxins.

Sick or dying birds often seek shelter in dense vegetation; thus, those that succumb to HAB exposure are not often observed or documented. Birds that are debilitated or die in exposed areas are subject to predation or may be swept away in tidal areas. When extensive fish kills occur from HABs, the carcasses of smaller birds such as shorebirds may go undetected. Some areas affected by HABs are remote and rarely visited. Thus, mortality of shorebirds associated with HABs is likely underreported.

Finally, HABs can affect shorebirds by affecting populations of their invertebrate prey species (see Factor E—Reduced Food Availability). In Australia’s Roebuck Bay, a bloom of toxic blue-green algae caused changes to the invertebrate community, with resistant species increasing and other species decreasing. These ecological changes affected the foraging behavior of shorebirds (Vivian 2013b).

HABs—Gulf of Mexico

Algal blooms causing massive fish kills in the Gulf of Mexico have been reported anecdotally since the 1500s, but written records exist only since 1844. The dinoflagellate *Karenia brevis* has been implicated in producing harmful red tides that occur annually in the Gulf of Mexico. Red tides cause extensive marine animal mortalities and human illness through the production of highly potent neurotoxins known as brevetoxins (FFWCC 2011). Brevetoxins are toxic to fish, marine mammals, birds, and humans, but not to shellfish (FAO 2004, p. 137). However, accumulation of brevetoxins by suspension-feeding bivalves has been well established in controlled laboratory studies (Bricelj *et al.* 2012, p. 30). Thus, brevetoxins can be transferred (e.g., via shellfish) through the food chain and are accumulated in or transferred by biota at many trophic levels (Landsberg *et al.* 2009, p. 598). Along Florida’s Gulf coast, high levels of brevetoxins have been documented in clams, including the red knot prey species *Donax variabilis*, sometimes persisting weeks after the bloom subsided (Bricelj *et al.* 2012, p. 30).

Karenia brevis has come to be known as the Florida red tide organism and has also been implicated in HABs in the Carolinas, Alabama, Mississippi, Louisiana, and Texas in the United States, as well as in Mexico (Marine Genomics Project 2010; Steidinger *et al.* 1999, pp. 3–4). Although red tides can occur throughout the year, most typically start from late August through November and last for 4 to 5 months. Red tides lasting as long as 21 months have occurred in Florida (FFWCC 2011).

There are anecdotal reports that red tide has caused mortality of red knots on Florida's west coast (B. Harrington pers. comm. November 14, 2013), but we have been unable to verify these. One wildlife rehabilitation facility has reported treating one red knot for presumed brevetoxicosis (red tide poisoning). Recovered off Fort Myers Beach, the bird was admitted on January 22, 2013, for being "weak, depressed, sitting on hocks, unwilling to stand" and had a brevetoxin level of 2.64 nanograms per milliliter. After treatment with fluids, anti-inflammatory drugs, and supportive care, the bird was released back to Fort Myers Beach on January 26 and flew away normally (H. Barron pers. comm. April 29, 2014). Though not documenting widespread effects or mortality, this report does confirm that red tide poisoning of red knots, unreported in the scientific literature, has occurred in Florida.

Although only anecdotal HAB-related red knot mortality has been reported from Florida, HABs have become a common feature of Florida's coastal environment and are associated with fish, invertebrate, bird, manatee, and other wildlife kills (Abbott *et al*. 2009, p. 3; Steidinger *et al.* 1999, pp. v, 3–4). Red tides occur nearly every year along Florida's Gulf coast, and may affect hundreds of square miles (FFWCC 2011). Red tides are most common off the central and southwestern coasts of Florida between Clearwater and Sanibel Island (FFWCC 2011), which constitute a key portion of the red knot's Southeast wintering area (Niles 2009, p. 4; Niles *et al.* 2008, p. 17). Brevetoxins from red tides accumulate in mollusks such as the small coquina clams that red knots are known to forage on in Florida. Reports of dead birds during red tide events are not unusual but are not well documented in the scientific literature. More often, red tides are documented by reports of fish kills, which can be extensive (FFWCC 2011).

A red tide event occurred in October 2009 along the Gulf coast of Texas during the period when red knots were using the area (Niles *et al.* 2009, Appendix 2). Aerosols produced by the red tide were present and affecting human breathing on Padre Island. Over a 2-week period, hundreds of thousands of dead fish littered beaches from Mustang Island, Texas, south into northern Tamaulipas, Mexico. Most shorebirds became conspicuously absent from Gulf coast beaches during that time (Niles *et al.* 2009, p. 5). A red knot captured and banded on October 6, 2009, was found 4 days later in poor condition on Mustang Island. The bird was captured by hand and taken to an animal rehabilitation facility. This bird had been resighted on October 7, the day after its original capture, walking normally and feeding. At the time of first capture the bird weighed 3.9 oz (113 g); its weight on arrival at the rehabilitation facility 4 days later was 2.7 oz (78 g) (Niles *et al.* 2009, p. 5). While there is no direct evidence, the red tide event is suspected as the reason for generally low weights and for a sharp decline in weights of red knots captured on Mustang Island during October 2009. Not only was the average mass of all the knots caught on Mustang Island low compared with other regions, but also average weights of individual catches declined significantly over the short period of field work (Niles *et al.* 2009, p. 4), coinciding with the red tide event.

Another Texas red tide event was documented by shorebird biologists in October 2011. Over a few days, the observed red knot population using Padre Island fell from 150 birds to only a few individuals. Captured birds were in extremely poor condition with weights as low as 2.9 oz (84 g) (Niles 2011c). Researchers picked up six red knots from the beach that were too weak to fly or stand and took them to a rehabilitator. Two knots that died before reaching the rehabilitation facility were tested for brevetoxin concentrations. Liver samples in both cases exceeded 2,400 nanograms of brevetoxin per gram of tissue (ng/g) (wet weight) (Newstead 2014a, p. 23). These levels are extremely high (Newstead 2014a, p. 23; Atwood 2008, p. 27). Samples from muscle and gastrointestinal tracts were also positive for brevetoxin, but at least an order of magnitude lower than in the livers. An HAB expert concluded that brevetoxins accounted for the mortality of these red knots (Newstead 2014a, p. 23). Whether the toxin was taken up by the birds through breathing or via consumption of contaminated food is unclear. However, other shorebird species that do not specialize on mollusks (especially sanderling and ruddy turnstone) were present during the red tide but did not appear to be affected by brevetoxins. This observation suggests uptake in the red knots may have been related to consumption of clams that had accumulated the toxin. In the case of this red tide event, the outbreak was confined to the Gulf beaches, but *Karenia brevis* is capable of spreading into bay habitats (e.g., Laguna Madre) as well. Red knots are apparently vulnerable to red tide toxins, so a widespread outbreak could significantly diminish the amount of available habitat (D. Newstead pers. comm. March 5, 2013). The total number of knots seen on the gulf beaches during fall 2011 was by far the lowest in recent years. Though it is unknown how many knots may have died as a result of this red tide, evidence from resighting of marked individuals later in the winter and following spring indicate some birds avoided the area during the red tide (Newstead 2014a, p. 24).

HABs—Uruguay

In April 2007, 312 red knots were found dead on the coast of southeastern Uruguay at Playa La Coronilla. Another 1,000 dead shorebirds were found nearby on the same day, also in southeastern Uruguay, but could not be confirmed to be red knots. Local bird experts suspected that the shorebird mortality event could be related to an HAB (BirdLife International 2007). However, the cause of death could not be determined, and no connection with an HAB could be established (J. Aldabe pers. comm. February 4, 2013). Red knots passing through Uruguay in April would be expected to be those that had wintered in Tierra del Fuego. A die-off of up to 1,300 red knots would account in large part for the 15 percent red knot decline observed in Tierra del Fuego in winter 2008.

HABs—Causes and Trends

During recent decades, the frequency, intensity, geographic distribution, and impacts of HABs have increased, along with the number of toxic compounds found in the marine food chain (Anderson 2007, p. 2; FAO 2004, p. 2). Coastal regions throughout the world are now subject to an unprecedented variety and frequency of HAB events. Many countries are faced with a large array of toxic or harmful species, as well as trends of increasing bloom incidence, larger areas affected, and more marine resources impacted. The causes behind this expansion are debated, with possible explanations ranging from natural mechanisms of species dispersal and enhancement to a host of human-related phenomena including climate change (Anderson 2007, pp. 3, 13; FAO 2004, p. 2). The influence of human activities in coastal waters may allow HABs to extend their ranges and times of residency (Steidinger *et al.* 1999, p. v).

Some new bloom events reflect indigenous algal populations discovered because of better detection methods and more observers. Several other "spreading events" are most easily attributed to natural dispersal via currents, rather than human activities (Anderson 2007, p. 11). However, human activities have contributed to the global HAB expansion by transporting toxic species in ship ballast water (Anderson 2007, p. 13). Another factor contributing to the global expansion in HABs is the substantial increase in aquaculture activities in many countries (Anderson 2007, p. 13), and the transfer of shellfish stocks from one area to another (FAO 2004, p. 2). Changed land use patterns, such as deforestation, can also cause shifts in phytoplankton species composition by increasing the concentrations of organic matter in land runoff. Acid precipitation can further increase the mobility of organic matter and trace metals in soils (FAO 2004, p. 1), which contribute to creating environmental conditions suitable for HABs.

Of the causal factors leading to HABs, excess nutrients often dominate the discussion (Steidinger *et al.* 1999, p. 2). Coastal waters are receiving large and increasing quantities of industrial, agricultural, and sewage effluents through a variety of pathways. In many urbanized coastal regions, these anthropogenic inputs have altered the size and composition of the nutrient pool which may, in turn, create a more favorable nutrient environment for certain HAB species (Anderson 2007, p. 13). Shallow and restricted coastal waters that are poorly flushed appear to be most susceptible to nutrient-related algal problems. Nutrient enrichment of such systems often leads to excessive production of organic matter (a process known as eutrophication) and increased frequencies and magnitudes of algal blooms (Anderson 2007, p. 14).

On a global basis, Anderson *et al.* (2002, p. 704) found strong correlations between total nitrogen input and phytoplankton production in estuarine and marine waters. There are also numerous examples of geographic regions (e.g., Chesapeake Bay, North Carolina's Albemarle-Pamlico Sound) where increases in nutrient loading have been linked with the development of large biomass blooms, leading to oxygen depletion and even toxic or harmful impacts on marine resources and ecosystems. Some regions have witnessed reductions in phytoplankton biomass or HAB incidence upon implementation of nutrient controls. Shifts in algal species composition have often been attributed to changes in the ratios of various nutrients (nitrogen, phosphorous, silicon) (Anderson *et al.* 2002, p. 704), and it is possible that algal species that are normally not toxic may be rendered toxic when exposed to atypical nutrient regimes resulting from human-caused eutrophication (FAO 2004, p. 1). The relationships between nutrient delivery and the development of blooms and their potential toxicity or harmfulness remain poorly understood. Due to the influence of several environmental and ecological factors, similar nutrient loads do not have the same impact in different environments, or in the same environment at different times. Eutrophication is one of several mechanisms by which harmful algae appear to be increasing in extent and duration in many locations (Anderson *et al.* 2002, p. 704).

Although important, eutrophication is not the only explanation for algal blooms or toxic outbreaks (Anderson *et al.* 2002, p. 704). The link is clear between nutrients and nontoxic algal blooms, which can cause oxygen depletion in the water, fish kills, and other ecosystem impacts

(Woods Hole 2012; Anderson 2007, p. 5; Anderson *et al.* 2002, p. 704; Steidinger *et al.* 1999, p. 2). However, the connection with excess nutrients is less clear for algal species that produce toxins, as toxic blooms can begin in open water miles away from shore or the immediate influence of human activities (Steidinger *et al.* 1999, p. 2). Many of the new or expanded HAB problems have occurred in waters with no influence from pollution or other anthropogenic effects (Anderson 2007, pp. 11, 13).

The overall effect of nutrient overenrichment on harmful algae is species specific. Nutrient enrichment has been strongly linked to stimulation of some harmful algal species, but for others it has apparently not been a contributing factor (Anderson *et al.* 2002, p. 704). There is no evidence of a direct link between Florida red tides and nutrient pollution (FFWCC 2011). Elevated nutrients in inshore areas do not start these blooms but, in some instances, can allow a bloom to persist in the nutrient-rich environment for a slightly longer period than normal (Steidinger *et al.* 1999, p. 2). For those regions and algal species where nutrient enrichment is a causative or contributing factor, increased coastal water temperatures and greater spring runoff associated with global warming may increase the frequency of HABs (USGCRP 2009, pp. 46, 150). The incidence of harmful algal blooms is expected to increase with climate change ("medium confidence") (Melillo *et al.* 2014, pp. 253–254).

Coastal managers are working toward mitigation, prevention, and control of HABs. Mitigation efforts are typically focused on protecting human health (Anderson 2007, p. 15), and are thus unlikely to prevent exposure of red knots. Several challenges hinder prevention efforts, including lack of information regarding the factors that cause blooms and limitations on the extent to which those factors can be modified or controlled (Anderson 2007, p. 16). Bloom control is the most challenging and controversial aspect of HAB management. Control refers to actions taken to suppress or destroy HABs, directly intervening in the bloom process. There are five categories or strategies that can be used to combat or suppress an invasive or harmful species, consisting of mechanical, biological, chemical, genetic, and environmental control. Several of these methods have been applied to HAB species (Anderson 2007, p. 18). However, the science behind HAB control is rudimentary and slow moving, and most control methods are currently infeasible, theoretical, or only possible on an experimental scale (Anderson 2007, pp. 18–20). It is likely that HABs will always be present in the coastal environment and, in the next few decades at least, are likely to continue to expand in geographic extent and frequency (Anderson 2007, p. 2).

HABs—Summary

To date, direct mortality of red knots from HABs have been documented only in Texas (Newstead 2014a, p. 23), although there are anecdotal reports that red tide has also caused red knot sickness and mortality on Florida's west coast (B. Harrington pers. comm. November 14, 2013). A large die-off in Uruguay may have also been linked to an HAB, but this link was not substantiated (J. Aldabe pers. comm. February 4, 2013). We conclude that some level of undocumented red knot mortality from HABs likely occurs most years, based on probable underreporting of shorebird mortalities from HABs and the direct exposure of red knots to algal toxins (particularly via contaminated prey) throughout the knot's nonbreeding range. We have no documented evidence that HABs were a driving factor in red knot population declines in the 2000s. However, HAB frequency and duration have increased and do not show signs of abating over the next few decades (Melillo *et al.* 2014, pp. 253–254; Anderson 2007, p. 2; FAO 2004, p. 2). Combined with other threats, ongoing and possibly increasing effects from HABs may be a regionally important contributor to red knot mortality.

Factor E—Oil Spills and Leaks

The red knot has the potential to be exposed to petroleum products throughout its migration and wintering range. Worldwide, more than 1.28 million tons (1.3 metric tons) of petroleum enters the sea annually (Blackburn *et al.* 2014, p. 1). Human-induced sources of petroleum in marine habitats include spills from shipping vessels; leaks or spills from offshore oil rigs or undersea pipelines; leaks, spills, or effluent from onshore facilities such as petroleum refineries and petrochemical plants; beach-stranded barrels and containers that fall from cargo ships or offshore rigs; discharges of ballast water from oil tankers; oil/water separators on production platforms; and terrestrial sources such as effluent from sewage treatment plants and runoff from roads and parking lots (Blackburn *et al.* 2014, p. 1). Several key red knot wintering or stopover areas contain large-scale operations for petroleum extraction, transportation, or both. Oil, as well as spill response activities, can directly and indirectly affect both the bird and its habitat through several pathways. With regard to potential effects on red knots and their habitats, the geographic location of a spill, weather conditions (e.g., prevailing winds), and type of oil spilled are as important, if not more so, than the volume of the discharge.

Oil Spills—Biological and Ecological Effects

Petroleum oils are complex and variable mixtures of many chemicals and include crude oils and their distilled products that are transported globally in large quantities. Overwhelming evidence exists that petroleum oils are toxic to birds (Leighton 1991, p. 43). Acute exposure to oil can result in death from hypothermia (i.e., from loss of the feathers' waterproofing and insulating capabilities), smothering, drowning, dehydration, starvation, or ingestion of toxins during preening (Henkel *et al.* 2012, p. 680; Peterson *et al.* 2003, p. 2085). In shorebirds, oil ingestion by foraging in contaminated intertidal habitats and consumption of contaminated prey may also be a major contamination pathway (Henkel *et al.* 2012, p. 680; Peterson *et al.* 2003, p. 2083). Mortality from ingested oil is primarily associated with acute toxicity involving the kidney, liver, or gastrointestinal tract (Henkel *et al.* 2012, p. 680; Leighton 1991, p. 46). In addition to causing acute toxicity, ingested oil can induce a variety of toxicologically significant systemic effects (Leighton 1991, p. 46). Since shorebird migration is energetically and physiologically demanding, the sublethal effects of oil may have severe consequences that lead to population-level effects (Henkel *et al.* 2012, p. 679). Oil can have long-term effects on populations through compromised health of exposed animals and chronic toxic exposures from foraging on persistently contaminated prey or habitats (Peterson *et al.* 2003, p. 2085).

Oiled birds may also experience decreased foraging success due to a decline in prey populations following a spill or due to increased time spent preening to remove oil from their feathers (Henkel *et al.* 2012, p. 681). Shorebirds oiled during the 1996 T/V *Anitra* spill in Delaware Bay showed significant negative correlations between the amount of oiling and foraging behaviors, and significant positive correlations between oiling and time spent standing and preening (Burger 1997a, p. 293). Moreover, oil can reduce invertebrate abundance or alter the intertidal invertebrate community that provides food for shorebirds (Henkel *et al.* 2012, p.

681; USFWS 2012a, p. 35). The resulting inadequate weight gain and diminished health may delay birds' departures, decrease their survival rates during migration, or reduce their reproductive fitness (Henkel *et al.* 2012, p. 681). In addition, reduced abundance of a preferred food may cause shorebirds to move and forage in other, potentially lower quality, habitats (Henkel *et al.* 2012, p. 681; USFWS 2012a, p. 35). Prey switching has not been documented in shorebirds following an oil spill (Henkel *et al.* 2012, p. 681). However shorebirds including red knots are known to switch habitats in response to disturbance (Burger *et al.* 1995, p. 62) and to switch prey types if supplies of the preferred prey are insufficient (Escudero *et al.* 2012, pp. 359, 362). A bird's inability to obtain adequate resources delays its premigratory fattening and can delay the departure to the breeding grounds; birds arriving on their breeding grounds later typically realize lower reproductive success (see Factor E—Asynchronies) (Henkel *et al.* 2012, p. 681).

In most nonbreeding areas, mollusks such as clams, mussels, and snails are the primary prey for red knots (see Migration and Wintering Food). Mollusks are highly sensitive to oil (Blackburn *et al.* 2014, p. 3). Large numbers of surf clams washed up on beaches after a 1996 spill off the coast of Rhode Island, and mass mortality of clams was caused by a 1978 spill off the coast of France (Blackburn *et al.* 2014, pp. 21–22). Oil ingested by mussels during filter feeding accumulates in their fatty tissues and may be retained on the gills. Mussels have limited capacity to metabolize oil, which prolongs their exposure and negatively impacts their feeding, growth, reproduction, embryo development, and immune response. Because many marine animals rely on mussels as a food source, the reduction in overall health of mussels chronically exposed to oil (which could affect their nutritive value) combined with increased mortality and population declines could have a large impact on the marine food web (Blackburn *et al.* 2014, pp. 3, 31). Snails in coastal environments have shown high levels of mortality after oil spills reduced recruitment of juveniles for years afterwards, and sublethal concentrations impair their mobility, foraging behavior, and reproduction (Blackburn *et al.* 2014, p. 3). See also Factor E—Reduced Food Availability.

Finally, efforts to prevent shoreline oiling and cleanup response activities can disturb shorebirds and their habitats (USFWS 2012a, p. 36; Burger 1997a, p. 293; Philadelphia Area Committee 1998, Annex E). Movement of response personnel on the beach and vessels in the water can flush both healthy and sick birds, causing disruptions in feeding and roosting behaviors (see Human Disturbance, above). In addition to causing disturbance, post-spill beach cleaning activities can impact habitat suitability and prey availability (see Factor A—Beach Cleaning, above). And lastly, dispersants used to break up oil can also have health effects on birds (NRC 2005, pp. 254–257).

Oil Spills—Canada

Oil spills are not a current threat to the red knot on its arctic breeding grounds. However, declining sea ice coverage is opening the possibility of shipping through previously unpassable parts of the Arctic. In 2007, the Northwest Passage, a sea lane running through much of the rufa red knot breeding range, was ice-free for the first time since satellite records began in 1978 (NRC 2013, p. 143; Astill 2012; Roach 2007). In 2013, for the first time, a large commercial freighter completed a voyage through the Northwest Passage, and smaller vessels have also begun to cross the region in the summer months (McGarrity and Gloystein 2013; Neuman 2013).

As of 2012, 185 voyages had been successfully completed through the Passage's 7 different routes. Plans have been announced for a large cruise ship, with capacity for over 1,000 passengers, to cross the Northwest Passage in 2016 (Andrews 2014). The freighter that crossed in 2013, delivering coal from Vancouver, Canada to Finland, had a strengthened hull to cope with floating ice (McGarrity and Gloystein 2013). However, Smith and Stephenson (2013, p. 1) found that, by mid-century, changing sea ice conditions would expand September navigability and open new routes through the Northwest Passage for moderately ice-strengthened ships, and even common open-water ships with no ice strengthening. These findings were the same under both climate scenarios analyzed by these authors (Smith and Stephenson 2013, p. 2).

Although sea ice currently represents the single greatest obstacle to trans-Arctic shipping, numerous additional factors, including dearth of services and infrastructure, high insurance and escort fees, unknown competitive response of the Suez and Panama Canals, poor charts, and other socioeconomic considerations, remain significant impediments to maritime activity in the region (Smith and Stephenson 2013, p. 2). Many maritime analysts believe that large volumes of commercial shipping via the Arctic are at least 10 years away (McGarrity and Gloystein 2013). However, we conclude that a substantial increase in commercial vessel traffic through the red knot's breeding grounds is likely over coming decades (NRC 2013, pp. 4, 16, 141–144). We have no data to indicate the rate at which vessel traffic will increase, where the predominant shipping lanes will be located relative to red knot breeding areas, or the proportion of vessels that would carry petroleum products. As this would be an entirely new shipping route, no data exist to suggest the risk of spills in this area relative to other shipping lanes (such as those discussed below). Thus, we are unable to assess the risk of oil spills in the breeding range. If a petroleum spill were to occur in this area, red knots would likely face considerably lower exposure than in their nonbreeding range, because the territorial-nesting birds are not concentrated in flocks, and because they rarely utilize salt-water habitats during the breeding season (Burger *et al.* 2012a, p. 26; Niles *et al.* 2008, p. 61; Harrington 2001, p. 16). Nonetheless, we conclude there would be some risk of exposure, as well as overall environmental damage that could indirectly impact breeding birds. (See Factor A—International Development—Canada regarding the potential for new development of the knot's breeding range as an indirect effect of warming that increases human access to the Arctic.)

The shorebird habitats of the Mingan Islands in the Gulf of St. Lawrence (Province of Quebec) are at risk from oil impacts because of their proximity to ships carrying oil through the archipelago to the Havre-Saint-Pierre harbor (Niles *et al.* 2008, p. 100). In March 1999, one ship spilled 40 tons (44 metric tons) of bunker fuel that washed ashore in the Mingan area. Oil from the 1999 spill did reach the islands used as a red knot foraging and staging area, but no information is available about the extent of impacts to prey species from the oil spill (USFWS 2011b, p. 23). If a similar accident were to occur during the July to October stopover period, it could have a serious impact on the red knots and their feeding areas (USFWS 2011b, p. 23; Niles *et al.* 2008, p. 100). In addition, some of the roughly 7,000 vessels per year that transit the St. Lawrence seaway illegally dump bilge waste water, which is another source of background-level oil and contaminant pollution affecting red knot foraging habitat and prey resources within the Mingan Island Archipelago (USFWS 2011b, p. 23). However, we have no specific information on the extent or severity of this contamination.

Oil Spills—Delaware Bay

The Delaware Bay and River are among the largest shipping ports in the world, especially for oil products (Clark in Farrell and Martin 1997, p. 24), and home to the fifth largest port complex in the United States in terms of total waterborne commerce (Philadelphia Area Committee 1998, Annex E). Every year, over 70 million tons of cargo move through the tri-state port complex, which consists of the ports of Philadelphia, Pennsylvania; Camden, Gloucester City, and Salem, New Jersey; and Wilmington, Delaware. This complex is the second largest U.S. oil port, handling about 85 percent of the east coast's oil imports (Philadelphia Area Committee 1998, Annex E).

The farthest upstream areas of Delaware Bay used by red knots (Niles *et al.* 2008, p. 43) are about 30 river miles (48 river km) downstream of the nearest port facilities, at Wilmington, Delaware. However, all vessel traffic must pass through the bay en route to and from the ports. In general, high-risk areas are where the greatest concentrations of chemical facilities are located, as major pollution incidents have typically occurred in locations where quantities of pollutant materials are stored, processed, or transported. Several areas considered high risk by the USCG are within the region used by red knots during spring migration, including Port Mahon and the Big Stone Beach Anchorage in Delaware, and the Delaware Bay and its approaches (Philadelphia Area Committee 1998, Annex E).

The narrow channel and frequent occurrence of strong wind and tide conditions increase the risk of oil spills in the Delaware River or Bay (Clark in Farrell and Martin 1997, p. 24); however, maritime accidents and groundings also frequently occur in fair weather and calm seas. Because the river is tidal, plumes of discharged material can spread upstream and downstream depending upon the tide. Generally, pollutants in the river travel proximally 4 mi (6.4 km) upstream during the flood cycle, and 5 mi (8 km) downstream during the ebb cycle. Wind direction and speed also play important roles in oil movement while free-floating oil remains on the water. As the Delaware River and upper bay are long and narrow, any medium or large spills are likely to affect both banks for several miles up and down the shorelines. In addition to direct spill effects, indirect impacts may occur during control of vessel traffic during a discharge, which can cause visual and noise disturbance to local wildlife, particularly shoreline-foraging species (Philadelphia Area Committee 1998, Annex E).

Table 24. Oil spills greater than 10,000 gallons (37,854 liters) in the Delaware River and Bay since 1985 (NOAA 2013d)

Vessel	Date	Volume (gallons)	Location	Approximate River Miles from Red Knot Habitat
M/V Athos 1	11/12/2004	265,000	Paulsboro, NJ	45
T/V Anitra	5/9/1996	42,000	Big Stone Anchorage, DE	0
T/V Presidente Rivera	6/24/1989	306,000	Marcus Hook, NJ	40
T/V Grand Eagle	9/28/1985	435,000	Marcus Hook, NJ	40
T/V Mystra	9/18/1985	10,000	Delaware Bay	0

Although there have been several thousand spills reported in the Delaware River since 1986, the average release was only about 150 gallons (gal) (568 liters (L)) per spill. Less than 1 percent of all spills in the port are greater than 10,000 gal (37,854 L). Table 24 shows the history of spills greater than 10,000 gal (37,854 L) in the port since 1985. Based on the history of spills in the Delaware River, a release of 200,000 to 500,000 gal (757,082 to 1.9 million L) of oil is the maximum that would be expected during a major incident. Major oil spills on the Delaware River to date have been less than the maximum. There is no known history of significant tank failures (discharges) in the port, although tank fires and explosions have occurred (Philadelphia Area Committee 1998, Annex E).

Although the *Anitra* spill occurred in May near red knot habitat, environmental conditions caused the oil to move around the Cape May Peninsula to the Atlantic coast of New Jersey by the second half of May. Thus, oil contamination of the bayshores was minimal during the period when the greatest concentrations of red knots were present in Delaware Bay (Burger 1997a, p. 291). However, unusually large numbers of shorebirds fed on the Atlantic coast in the spring of 1996 because cold waters delayed the horseshoe crab spawn in Delaware Bay (Burger 1997a, p. 292), thus increasing the number of birds exposed to the oil. These circumstances underscore the importance of spill location and environmental conditions, not just merely spill volume, in determining the impacts of a spill on red knots. Although red knots were present in at least one oiled location (Ocean City, New Jersey) (Burger 1997a, p. 292) and at least a few knots were oiled (J. Burger pers. comm. March 5, 2013), the vast majority of impacts were to sanderlings and other shorebird species (*Anitra* Natural Resource Trustees 2004, p. 5).

Large spills upriver, or moderate spills in the upper bay, have the potential to contact a significant portion of the shorebird concentration areas. Although the migration period when crabs and shorebirds are present is short, even a minor spill (i.e., less than 1,000 gal (3,785 L)) could, depending on the product spilled, affect beach quality for many years. Both New Jersey and Delaware officials work closely with Emergency Response managers and the USCG in planning for such an occurrence (Kalasz 2008, pp. 39–40; Clark in Farrell and Martin 1997, p. 24).

Oil Spills—Gulf of Mexico

As of 2010, there were 3,409 offshore petroleum production facilities in Federal waters within the Gulf of Mexico Outer Continental Shelf (OCS), down from 4,045 in 2001 (Bureau of Safety and Environmental Enforcement (BSEE) undated). Gulf of Mexico Federal offshore operations account for 23 percent of total U.S. crude oil production and 7 percent of total U.S. natural gas production. Over 40 percent of the total U.S. petroleum refining capacity, as well as 30 percent of the U.S. natural gas processing plant capacity, is located along the Gulf coast. Total liquid fuels production in 2011 was 10.3 million barrels per day (U.S. Energy Information Administration 2013). For the entire Gulf of Mexico region, total oil production in 2012 was 425 million barrels, down from 570 million barrels in 2009 (BSEE 2013).

The BSEE tracks spill incidents of one barrel or greater in size of petroleum and other toxic substances resulting from Federal OCS oil and gas activities (BSEE 2012). Table 25 shows the number of spills 50 barrels (2,100 gal (7,949 L)) or greater in the Gulf of Mexico since 1996.

These figures do not include incidents stemming from substantial extraction operations in State waters. Crude oil production in 2012 was an estimated 4.9 million barrels in Louisiana State waters (Louisiana Department of Natural Resources (LDNR) 2013), and over 272,000 barrels in Texas State waters (Railroad Commission of Texas 2013). In Louisiana, about 2,500 to 3,000 oil spills are reported in the Gulf region each year, ranging in size from very small to thousands of barrels (USFWS 2012a, p. 37).

Table 25. Federal Outer Continental Shelf spill incidents 50 barrels (2,100 gallons (7,949 liters)) or greater, resulting from oil and gas activities, 1996 to 2012 (BSEE 2012)

Year	Number of Incidents
2012	8
2011	3
2010	5
2009	11
2008	33
2007	4
2006	14
2005	49
2004	22
2003	12
2002	12
2001	9
2000	7
1999	5
1999	9
1997	3
1996	3

Nationwide, spill rates (the number of incidents per billion barrels of crude oil handled) in several sectors decreased or remained stable over recent decades. From 1964 to 2010, spill rates declined for OCS pipelines, and spill rates from tankers decreased substantially, probably because single-hulled tankers were largely phased out (see “International Laws and Regulations” under Factor D). Looking at the whole period from 1964 to 2010, nationwide spill rates for OCS platforms were unchanged for spills 1,000 barrels or greater, and decreased for spills 10,000 barrels or greater. However, spill rates at OCS platforms increased in the period 1996 to 2010 relative to the period 1985 to 1999, as the later period included several major hurricanes (e.g., Hurricane Katrina and Hurricane Rita) and the Deepwater Horizon spill (Anderson *et al.* 2012, pp. iii–iv). Generally decreasing spill rates were partially offset by increasing production, as shown in table 26.

In the Gulf of Mexico, threats from oil spills are primarily from the high volume of shipping vessels, from which most documented spills have originated, traveling offshore and within connected bays. In addition to the risk of leaks and spills from offshore oil rigs, pipelines, and petroleum refineries, there is a risk of leaks from oil-filled barrels and containers that routinely wash up on the Texas coast. Federal and State land managers have protective provisions in place to secure and remove the barrels, thus reducing the likelihood of contamination (M. Bimbi pers. comm. November 1, 2012).

Table 26. Nationwide Outer Continental Shelf petroleum production, and spills 1 barrel or greater, 1964 to 2009* (Anderson *et al.* 2012, p. 10)

			Barrels Spilled by Spill Size			Number of Spills by Spill Size		
Year	**Barrels Spilled per Billion Barrels Produced**	**Billions of Barrels Produced**	**Total**	**1 to 999 Barrels**	**1,000 Barrels or Greater**	**Total**	**1 to 999 Barrels**	**1,000 Barrels or Greater**
1964-1970	255,280	1.54	394,285	3,499	390,786	33	23	10
1971-1990	16,682	6.79	113,307	21,415	91,892	1,921	1,909	12
1991-2009	6,427	9.2	59,142	28,144	30,998	853	843	10
1964-2009	32,329	17.53	566,734	53,058	513,676	2,807	2,775	32

*Spill data for 1964 to 1970 are for spills of 50 barrels or greater. Barrels of production or spillage may not add due to rounding of decimals not shown. One barrel equals 42 gallons (159 liters).

Chronic spills of oil from rigs and pipelines and natural seeps in the Gulf of Mexico generally involve small quantities of oil. The oil from these smaller leaks and seeps, if they occur far enough from land, tend to wash ashore as tar balls. In cases such as this, the impact is limited to discrete areas of the beach, whereas oil slicks from larger spills coat longer stretches of the shoreline. In late July and early August 2009, for example, oil suspected to have originated from an offshore oil rig in Mexican waters was observed on 14 piping plovers in south Texas (USFWS 2012a, p. 37). Mexican waters were not included in the oil and gas production or spill statistics given above.

On April 20, 2010, an explosion and fire occurred on the mobile offshore drilling unit Deepwater Horizon, which was being used to drill a well in the Macondo prospect (Mississippi Canyon 252) (Natural Resource Trustees 2012, p. 7). The rig sank and left the well releasing tens of thousands of barrels of oil per day into the Gulf of Mexico. It is estimated that 5 million barrels (210 million gal (795 million L)) of oil were released from the Macondo wellhead. Of that, approximately 4.1 million barrels (172 million gal (651 million L)) of oil were released directly into the Gulf of Mexico over nearly 3 months. In what was the largest and most prolonged offshore oil spill in U.S. history, oil and dispersants impacted all aspects of the coastal and oceanic ecosystems (Natural Resource Trustees 2012, p. 7). Oiling from this event was documented on 1,102 mi (1,773 km) of shoreline, over 95 percent of which consisted of beaches and marshes. Two years later, oil remained on 427 mi (687 km), though at much lesser degrees of oiling (Michel *et al.* 2013, p. 1). Limited cleanup operations were still ongoing throughout the spill area in November 2012 (USFWS 2012a, p. 36). Red knots are not listed among the bird species rescued and collected during the response (USFWS 2011f). Of 135 red knot observations in the live bird oiling rate assessment data, none were reported as oiled (D. McClain pers. comm. June 15, 2014). However, a more comprehensive evaluation (e.g, oiling of red knot habitats, likelihood that oiled knots went undetected) may eventually become available through the Natural Resources Damage Assessment (NRDA) that is assessing injury to wildlife resources (Natural Resource Trustees 2012, pp. 8–9). D. Newstead (pers. comm. June 6, 2014) observed an oiled red knot on Padre Island, Texas around August 2010, independent of any surveys conducted as part of the spill response. We cannot confirm if this oil was from Deepwater Horizon (D. Newstead pers. comm. June 6, 2014). Geolocator data show that another red knot spent three days in southeast Louisiana from July 25 to 28, 2010 before returning to the south Texas coast (Niles 2012a). We have no information to indicate if this geolocator bird was exposed to oil from the Deepwater Horizon spill, but oiling of the shoreline was extensive during this timeframe.

On March 22, 2014, a bulk carrier moving through the Houston, Texas ship channel collided with a barge carrying oil, which resulted in approximately 168,000 gallons of marine fuel oil being released into Galveston Bay and dispersing into the Gulf of Mexico. A significant amount of oil from the spill drifted approximately 150 mi (241 km) south making landfall on Matagorda Island, a unit of the Aransas NWR (USFWS 2014c). This spill in Galveston Bay also resulted in moderate amounts of oil washing up on beaches on Mustang and North Padre Islands. Despite the moderate amounts of oil and quick cleanup, well over half of the shorebirds surveyed on Mustang and North Padre Islands in the days following the oil arriving on the beaches showed trace to heavy amounts of oiling. Red knots were not among the species documented in the oiled areas; however, their absence from the beach was a chance result of weather and tidal conditions that made preferable habitat available in the protected bay shorelines. Minor differences in tides and weather would likely have resulted in the exposure of a large percentage of the red knots along the south Texas coast to the oil (D. Newstead pers. comm. May 8, 2014).

Oil Spills—South America

South America—North Shore and Patagonia

Threats to red knot habitat in Maranhão, Brazil include oil pollution as well as habitat loss (see Factor A—International Coastal Development) from offshore petroleum exploration on the continental shelf (WHSRN 2012; Niles *et al.* 2008, p. 97; COSEWIC 2007, p. 37). Deep sea oil searches are underway in French Guiana, and oil spills could happen in the distant future if any extraction is pursued. Oil washing up on the shores is very rare in French Guiana (CSRPN 2013). The continental shelf of the Guianas holds important oil and gas resources. Oil drilling activities have started in Suriname and Guyana and present a potential threat for shorebirds through the risk of oil spills. Moreover, in Suriname, activities such as oil drilling are allowed in multiple use management areas and nature reserves providing that the flora and fauna are not harmed (Ottema and Spaans 2008, p. 345), but introducing the risk of impacts from a spill or leak.

Goldfeder and Blanco in Boere *et al.* (2006, p. 193) cited oil exploration as a potential threat to the red knot in Argentina. Oil pollution is a threat at several red knot wintering and stopover habitats along the Patagonian coast of Argentina including Península Valdés and Bahía Bustamante; at the latter site, 15 of 100 (15 percent) red knots were visibly oiled during a study in 1979 (Niles *et al.* 2008, p. 98; Harrington and Morrison 1980, p. 40). Further south in Argentina, at a shorebird reserve and red knot stopover area in Río Gallegos near Tierra del Fuego, the main threat comes from oil and coal transport activities. Crude oil and coal are loaded onto ships at a hydrocarbon port where the estuary empties into the sea adjacent to the salt marsh zone. This area has a history of oil tankers running aground because of extreme tides, strong winds, tidal currents, and piloting errors. A shipwreck at Río Gallegos could easily contaminate key areas used by shorebirds, including red knots (WHSRN 2012; Niles *et al.* 2008, p. 98; Ferrari *et al.* 2002, p. 39). However, oil pollution has decreased significantly along the Patagonian coast (Niles *et al.* 2008, p. 98).

South America—Tierra del Fuego

The risk of an oil spill is a primary threat to the largest red knot wintering areas in both the Chilean and Argentinean portions of Tierra del Fuego (WHSRN 2012; Niles *et al.* 2008, pp. 98–99; COSEWIC 2007, p. 36) due to the proximity of large-scale oil operations close to key red knot habitats. Oil operations have decreased in Chile around Bahía Lomas, but increased along the Argentinean coast of Tierra del Fuego (Niles *et al.* 2008, p. 98; COSEWIC 2007, pp. 36–37).

The region of Magellan, Chile, has traditionally been an important producer of oil and natural gas since the first oil discovery was made in 1945 within 6.2 mi (10 km) of the bayshore, in Manantiales. Production continues, although local oil activity has diminished over the last 20 years. Oil is extracted by drilling on land and offshore, the latter with no new drillings between 2000 and 2008. The largest single red knot wintering site, Bahía Lomas, has several oil platforms. Most are static, and several were closed around 2007 as the oil resource had been depleted (Niles *et al.* 2008, p. 98). However, the red knot area at Bahía Lomas remains at risk from a spill or leak from the remaining oil extraction facilities.

Exposure of red knots to hydrocarbon pollution at Bahía Lomas could also come from shipping accidents, as the site is located at the eastern end of the Strait of Magellan, an area historically characterized by high maritime shipping traffic (WHSRN 2012). Two oil spills from shipping have been recorded near the Strait of Magellan First Narrows (immediately west of Bahía Lomas), one involving 53,461 tons (48,500 metric tons) in 1974 and one involving 99 tons (90 metric tons) in 2004 (Niles *et al.* 2008, p. 98; COSEWIC 2007, p. 36). No incidents have been reported of red knots being affected by substantial oiling of the plumage or effects to the prey base. However, small amounts of oil have been noted on some red knots caught during banding operations (Niles *et al.* 2008, p. 98; COSEWIC 2007, p. 36).

In most years from 2000 to 2012 (10 of 12 years) for which survey data are available, Bahía Lomas supported over half of the total Argentina-Chile wintering population of red knots, rising to over 90 percent from 2010 through 2012 (G. Morrison pers. comm. August 31, 2012). Thus, a significant spill (or several small spills) has the potential to substantially impact red knot populations, depending on the timing and severity of oil contamination within red knot habitats. The National Oil Company extracts, transports, and stores oil in the area next to Bahía Lomas and has been an important and cooperative partner in conservation of the bay (WHSRN 2012), including recent efforts to develop a management plan for the area (Niles in Ydenberg and Lank 2011, p. 198).

On the nearby Atlantic Ocean coast of Argentinean Tierra del Fuego, oil drilling increased around 1998 (Niles *et al.* 2008, p. 98; COSEWIC 2007, pp. 36–37). In the Argentina portion of Tierra del Fuego, Bahía San Sebastián is the area most vulnerable from oil and gas operations that occur on lands near the coast and beach. During field work conducted in 1979, Harrington and Morrison (1980, p. 40) did not document any visible oiling among 150 red knots observed at Bahía San Sebastián. However, Bahía San Sebastián is surrounded by hundreds of oil wells (Gappa and Sueiro 2007, p. 680). An 18-in (46-cm) pipe submerged in the bay runs 2.9 mi (4.5 km) out to a buoy anchored to the seabed (WHSRN 2012). The pipe is used to load crude oil onto tankers bound for various distilleries in the country (WHSRN 2012; Gappa and Sueiro 2007, p. 680). Wind velocities over 37 mi per hour (60 km per hour) typically occur for 200 days of the year, and loading and transport of hydrocarbons often take place during rough seas. Thus, an oil spill is a persistent risk and could have long-term effects (Gappa and Sueiro 2007, p. 680). While companies have strict security controls, this activity remains a potential threat to shorebirds in the area (WHSRN 2012).

Farther south on Tierra del Fuego, the area near the shorebird reserves at Río Grande, Argentina, is important for onshore and offshore oil production, which could potentially contribute to oil pollution, especially from oil tankers loading around Río Grande City (Niles *et al.* 2008, pp. 98–99). During field work conducted in 1979, Harrington and Morrison (1980, p. 40) documented 1 visibly oiled bird among 150 red knots (1.3 percent) observed at Río Grande.

Oil Spills—Summary

Red knots are exposed to large-scale petroleum extraction and transportation operations in many key wintering and stopover habitats including Tierra del Fuego, Patagonia, the Gulf of Mexico, Delaware Bay, and the Gulf of St. Lawrence (NOAA 2013d; Anderson *et al.* 2012, p. 10; BSEE 2012; WHSRN 2012; USFWS 2011b, p. 23; Niles *et al.* 2008, p. 98, 100; Ottema and Spaans 2008, p. 345; COSEWIC 2007, pp. 36–37; Gappa and Sueiro 2007, p. 680; Ferrari *et al.* 2002, p. 39; Philadelphia Area Committee 1998, Annex E; Harrington and Morrison 1980, p. 40). To date, the documented effects to red knots from oil spills and leaks have been minimal; however, information regarding any oiling of red knots during the Deepwater Horizon spill has not yet been released (Natural Resource Trustees 2012, pp. 8–9). We conclude that high potential exists for small or medium spills to impact moderate numbers of red knots or their habitats, such that one or more such events is likely over the next few decades, based on the proximity of key red knot habitats to high-volume oil operations. Risk of a spill may decrease with improved spill contingency planning, infrastructure safety upgrades, and improved spill response and recovery methods. However, these decreases in risk (e.g., per barrel extracted or transported) could be offset if the total volume of petroleum extraction and transport continues to grow. A major spill affecting habitats in a key red knot concentration area (e.g., Tierra del Fuego, Gulf coasts of Florida or Texas, Delaware Bay, Mingan Archipelago) while knots are present is less likely but would be expected to cause population-level impacts. Oil spills are not a current threat to the red knot on its arctic breeding grounds. We conclude that a substantial increase in commercial vessel traffic through the red knot's breeding grounds is likely over coming decades (NRC 2013, pp. 4, 16, 141–144; Smith and Stephenson 2013, p. 2), but have no data to evaluate the risks of this potential future threat.

Factor E—Environmental Contaminants

Environmental contaminants can have profound effects on birds, acting from the molecular through population levels (Rattner and Ackerson 2008, p. 344). Little experimental work has been done on the toxic effects of organochlorines (e.g., polychlorinated biphenyls (PCBs); pesticides such as DDT (dichloro-diphenyl-trichloroethane), dieldrin, and chlordane) or trace elements (e.g., mercury, cadmium, arsenic, selenium) in shorebirds, but adult mortality due to organochlorine poisoning has been recorded (Braune and Noble 2009, pp. 200–201).

Contaminants—Canada

In 1991 and 1992, Braune and Noble (2009, p. 185) tested 12 shorebird species (not including *Calidris canutus*) from 4 sites across Canada (including 2 red knot stopover areas) for PCBs, organochlorine pesticides, mercury, selenium, cadmium, and arsenic. Contaminant exposure among species varied with diet, foraging behavior, and migration patterns. Diet composition seemed to provide a better explanation for contaminant exposure than bill length or probing behaviors. Based on the concentrations measured, researchers found no indication that contaminants were adversely affecting the shorebird species sampled in this study (Braune and Noble 2009, p. 201).

Heavy shipping traffic in the Gulf of St. Lawrence (Province of Quebec) presents a risk of environmental contamination, as well as possible oil spills (which were discussed above). Red knot habitats in the Mingan Islands are particularly at risk because large ships carrying titanium and iron navigate through the archipelago to the Havre-Saint-Pierre harbor throughout the year (COSEWIC 2007, p. 37).

At another red knot stopover area, the Bay of Fundy, chemicals such as herbicides and pesticides originate from farming activities along tidal rivers and accumulate in intertidal areas. These contaminants build up in the tissues of intertidal invertebrates (e.g., the burrowing amphipod *Corophium volutator* and the small clam *Macoma balthica*) that are, in turn, ingested by shorebirds, but with unknown consequences (WHSRN 2012).

Contaminants—Delaware Bay

The Delaware River and Bay biota are contaminated with PCBs and other pollutants (Suk and Fikslin 2006, p. 5). However, one preliminary study suggests that organic pollutants are not impacting shorebirds that eat horseshoe crab eggs. In 1992, USFWS (1996, p. i) tested horseshoe crab eggs, sand, and ruddy turnstones from two beaches on the Delaware side of Delaware Bay for organochlorines and trace metals. Sand, eggs, and bird tissues contained low to moderately elevated levels of contaminants. This limited study suggested that contamination of the shorebirds at Delaware Bay was probably not responsible for any decline in the population. However, at the time of this study, detection limits for organic contaminants were much higher than those that are now possible using current analytical capabilities. Thus, lower levels of contamination (which may impact wildlife) could not be detected by the testing that was performed (detection limits for horseshoe crab eggs were 0.07 to 0.20 parts per million (ppm), wet weight). Only one egg sample had a quantifiable level of PCBs, but this could have been due to the limitations of the tests to detect lower levels. A more extensive survey of horseshoe crab eggs throughout Delaware Bay would provide a more definitive assessment (USFWS 1996, p. i), especially if coupled with current analytical methods that can quantify residues at much lower concentrations. However, we are unaware of any plans to update this study.

Burger *et al.* (1993, p. 189) examined concentrations of lead, cadmium, mercury, selenium, chromium, and manganese in feathers of shorebirds, including red knots migrating north through Cape May, New Jersey, in 1991 and 1992. Although these authors predicted that metal levels would be positively correlated with weight, this was true only for mercury in red knots. Selenium was negatively correlated with weight in red knots. No other significant correlation of metal concentrations with weight was found. Selenium and manganese were highest in red knots, while lead, mercury, chromium, and cadmium were higher in other species (Burger *et al.* 1993, p. 189). Metal levels in the feathers partially reflect the extent of pollution at the location of the birds during feather formation, so these feather concentrations may not necessarily correspond to exposure during the Delaware Bay stopover (Burger *et al.* 1993, p. 193). The results of this study suggest that the levels of cadmium, lead, mercury, selenium, and manganese were similar to levels reported from other shorebird studies. However, the levels of chromium in this study were much higher than had been reported for other avian species (Burger *et al.* 1993, pp. 195–196).

Burger (1997b, p. 279) measured lead, mercury, cadmium, chromium, and manganese concentrations in the eggs of horseshoe crabs from 1993 to 1995, and from leg muscle tissues in 1995, in Delaware Bay. In eggs, mercury levels were below 100 parts per billion (ppb), or were nondetectable. Cadmium levels were generally low in 1993 and 1995 but were relatively higher in 1994. Lead levels in eggs decreased from 558 ppb in 1993 to 87 ppm in 1995. Selenium increased, chromium decreased, and manganese generally decreased. Leg muscles had significantly lower levels of all metals than eggs, except for mercury (Burger 1997b, p. 279). The high levels of some metals in eggs of horseshoe crabs may partially account for similar high levels in the feathers of shorebirds that feed on crab eggs while in Delaware Bay (Burger 1997b, p. 285).

Burger *et al.* (2002, p. 227) examined the levels of arsenic, cadmium, chromium, lead, manganese, mercury, and selenium in the eggs and tissues of 100 horseshoe crabs collected at 9 sites from Maine to Florida, including Delaware Bay. Arsenic levels were the highest, followed by manganese and selenium, while levels for the other metals averaged below 100 ppb for most tissues. The levels of contaminants found in horseshoe crabs, with the possible exceptions of arsenic in Florida and mercury in Barnegat Bay (New Jersey) and Prime Hook (Delaware), were below those known to cause adverse effects in the crabs themselves or in organisms that consume them or their eggs.

Revisiting the 1997 study specific to Delaware Bay, Burger *et al.* (2003, p. 36) examined the concentrations of arsenic, cadmium, chromium, lead, manganese, mercury, and selenium in the eggs and tissues of horseshoe crabs from eight locations on both sides of Delaware Bay.

Locational differences were detected but were small. Further, contaminant levels were generally low. The levels of contaminants found in horseshoe crabs were well below those known to cause adverse effects in the crabs themselves or in organisms that consume them or their eggs. Levels of these contaminants generally declined in the eggs of horseshoe crabs from 1993 to 2001, suggesting that these contaminants were not likely to be a problem for secondary consumers like red knot, or a cause of their decline.

Botton *et al.* (2006, p. 820) found no significant differences in the percentage of horseshoe crab eggs that completed development when cultured using water from Jamaica Bay (New York) or from lower Delaware Bay, a less polluted location. Only one percent of the embryos from Jamaica Bay exhibited developmental anomalies, a frequency comparable to a previously studied population from Delaware Bay. These authors suggested that the distribution and abundance of horseshoe crabs in Jamaica Bay were not limited by water quality (Botton *et al.* 2006, p. 820). This finding suggests that horseshoe crabs are not particularly sensitive to differences in water quality.

The USFWS (2007, p. ii) examined embryonic, larval, and juvenile horseshoe crab responses to a series of exposures (from 0 to 100 ppb) of methoprene, a mosquito larvicide (a pesticide that kills specific insect larvae). The results provided no evidence that a treatment effect occurred, with no obvious acute effects of environmentally relevant concentrations of methoprene on developing horseshoe crab embryos, larvae, or first molt juveniles. The study results suggested that exposure to methoprene may not be a limiting factor to horseshoe crab populations. However, horseshoe crab life stages after the first molt were not tested for methoprene effects, which have been found in other marine arthropod species. For example, Walker *et al*. (2005, pp. 118, 124) found that methoprene was toxic to lobster (*Homarus americanus*) stage II larvae at 1 ppb, and that stage IV larvae were more resistant but did exhibit significant increases in molt frequency beginning at exposures of 5 ppb. However, we do not have information on how or to what extent these levels of methoprene may affect horseshoe crab populations or red knots, through their consumption of exposed horseshoe crab eggs.

Contaminants—South America

Blanco *et al.* (2006, p. 59) documented the value of South American rice fields as an alternative feeding habitat for waterbirds. Agrochemicals are used in the management of rice fields. Although shorebirds are not considered harmful to the rice crop, they are exposed to lethal and sublethal doses of toxic products while foraging in these habitats. Rice fields act as important feeding areas for migratory shorebirds but can become toxic traps without adequate management (Blanco *et al.* 2006, p. 59). In rice field surveys from November 2004 to April 2005, red knots constituted only 0.7 percent of shorebirds observed, with three knots in Uruguay and none in Brazil or Argentina (Blanco *et al.* 2006, p. 59). Thus, exposure in these countries is low; however, much larger numbers of red knots (1,700) have been observed in rice fields in French Guiana (Niles 2012b), and 6 red knots have been reported from rice fields in Trinidad (eBird.org 2014).

Threats to red knot habitat in Maranhão, Brazil, include iron ore and gold mining, which can cause mercury contamination (WHSRN 2012; Niles *et al.* 2008, p. 97; COSEWIC 2007, p. 37). In the Guianas, drainage of pesticides from agricultural areas to the estuarine area may present a potential threat to shorebirds in the near future. The same could prove true for mercury, used in gold mining operations in the interior, and carried in sediment by the large rivers (Ottema and Spaans 2008, p. 345). Goldfeder and Blanco in Boere *et al.* (2006, p. 193) cited industrial pollution as a potential threat to the red knot in Argentina. The important migration stopover area at San Antonio Oeste, Argentina faces potential pollution from a soda ash factory built in 2005, which could release up to 250,000 tons of calcium chloride per year, affecting intertidal invertebrate food supplies. Garbage and port activities are additional sources of pollution in this region (WHSRN 2012; Niles *et al.* 2008, p. 98; COSEWIC 2007, p. 37).

At the southern Argentinean stopover of Río Gallegos, a trash dump adjoins the feeding and roosting areas used by shorebirds. Garbage is spread quickly by the strong winds characteristic of the region and is deposited over large parts of the estuary shore. This trash diminishes habitat quality, especially when plastics, such as polythene bags, cover foraging or roosting habitats (Niles *et al.* 2008, p. 98; Ferrari *et al.* 2002, p. 39). Pollution at Río Gallegos also stems from untreated sewage, but a project is under way to carry the waste offshore instead of discharging it into the shorebird habitats (WHSRN 2012) (see Factor A—Coastal Development—Other Countries).

In the past, organic waste from the City of Río Grande (in Argentinean Tierra del Fuego, population approximately 50,000), including that from a chicken farm, has been released at high tide over the flats where red knots feed (Atkinson *et al.* 2005, p. 745). We have no direct evidence of red knots having been affected by organic waste, but it remains a potential source of contamination risk (e.g., nutrients, trace metals, pesticides, pathogens, pharmaceuticals, endocrine disruptors) (Fisher *et al.* 2005, pp. iii, 4, 34) to the knots and their wintering habitat. As at Río Gallegos, wind-blown trash from a nearby landfill degrades shorebird habitats at one location in Río Grande, but the City is working to relocate the landfill. In addition, a methanol and urea plant and two seaports are in development (WHSRN 2012), which could also increase pollution.

Contaminants—Summary

Although red knots are exposed to a variety of contaminants across their nonbreeding range, we have no evidence that such exposure is impacting health, survival, or reproduction at the subspecies level. Exposure risks exist in localized red knot habitats in Canada, but best available data suggest shorebirds in Canada are not impacted by background levels of contamination (WHSRN 2012; Braune and Noble 2009, p. 201; COSEWIC 2007, p. 37). Levels of most metals in red knot feathers from the Delaware Bay have been somewhat high but generally similar to levels reported from other studies of shorebirds (Burger *et al.* 1993, pp. 195–196). One preliminary study suggests organochlorines and trace metals are not elevated in Delaware Bay shorebirds, although this finding cannot be confirmed without updated testing (USFWS 1996, p. i). Levels of metals in horseshoe crabs are generally low in the Delaware Bay region and not likely impacting red knots or recovery of the crab population (Burger *et al.* 2003, p. 36; Burger *et al.* 2002, p. 227; Burger 1997b, p. 285).

Horseshoe crab reproduction does not appear impacted by the mosquito control chemical methoprene (at least through the first juvenile molt) or by ambient water quality in mid-Atlantic estuaries (USFWS 2007, p. ii). Exposure of shorebirds to agricultural pollutants in rice fields may occur regionally in parts of South America, but red knot usage of rice field habitats was low in the several countries surveyed (Blanco *et al.* 2006, p. 59). Finally, localized urban pollution has been shown to impact South American red knot habitats (WHSRN 2012; Niles *et al.* 2008, p. 97–98; Ottema and Spaans 2008, p. 345; COSEWIC 2007, p. 37; Atkinson *et al.* 2005, p. 745; Ferrari *et al.* 2002, p. 39), but we are unaware of any documented health effects or population-level impacts. Thus, we conclude that environmental contaminants are not a threat to the red knot. However, see Cumulative Effects, below, regarding an unlikely but potentially high-impact synergistic effect among avian influenza, environmental contaminants, and climate change in Delaware Bay.

Factor E—Wind Energy Development

Wind Energy—Background

Within the red knot's U.S. wintering and migration range, substantial development of offshore wind facilities is planned, and the number of wind turbines installed on land has increased considerably over the past decade. The rate of wind energy development will likely continue to increase into the future as the United States looks to decrease reliance on the traditional sources of energy (e.g., fossil fuels) (Executive Office of the President 2013, entire). Analysis by the DOE shows the potential for wind energy to reduce greenhouse gas emissions in a rapid and cost-effective manner (DOE and BOEMRE 2011, p. 5; DOE 2008, p. 107), which may have potentially significant benefits to red knots by mitigating the effects of climate change. However, wind turbines can have direct (e.g., collision mortality) and indirect (e.g., migration disruption, displacement from habitat) impacts on birds.

We have minimal information on wind energy development trends in other countries within the range of the red knot. For example, wind energy development is widespread in southern Mexico and apparently continues to increase throughout the Pacific slope of the Isthmus of Tehuantepec, near a *Calidris canutus* (possibly *rufa*) wintering area (Newstead 2014b, p. 5). There are no wind turbines yet in French Guiana, but a project is planned to construct a few in the northern savannas about 0.5 mi (0.75 km) from the coast, where they would not likely present a threat to red knots (CSRPN 2013). In general, risks of red knot collisions and other effects of wind energy development are likely higher wherever large numbers of turbines are constructed along migratory pathways, either on land or offshore.

Large-scale installation of wind turbines represents a potential collision hazard for red knots during their migration (Burger *et al*. 2012c, p. 370; Burger *et al.* 2011, p. 348; Watts 2010, p. 1; Kuvlesky *et al.* 2007, p. 2487). Collision risks depend on the total number and height of turbines, turbine design and configuration, geography (e.g., migration "pinch points," corridors, peninsulas, islands), attractiveness of the habitat, behavior and ecology of the species, habitat and spatial use, and ability of the birds to perceive and avoid wind turbines at close range (Loss *et al.* 2013, pp. 206–207; Burger *et al.* 2011, p. 340; Kuvlesky *et al.* 2007, p. 2488; NRC 2007, p. 138; Chamberlain *et al*. 2006, p. 198; Drewitt and Langston 2006, p. 30).

Although little shorebird-specific information is available, the effect of weather on migrating bird flight altitudes both over land and offshore has been well documented. Numerous studies indicate that the risk of bird collisions with wind turbines increases as weather conditions worsen and visibility decreases (NRC 2007, p. 91; Drewitt and Langston 2006, p. 31; Hüppop *et al*. 2006, pp. 102, 105–107; Exo *et al*. 2003, p. 51). If birds are migrating at high altitudes and suddenly encounter fog, precipitation, or strong head winds, they may be forced to fly at lower altitudes, increasing their collision risks if they fly in the rotor (i.e., turbine blade) swept zone (Drewitt and Langston 2006, p. 31). The greatest collision risk occurs at night, particularly in unfavorable weather conditions. Behavioral observations have shown that most birds fly closer to the height of turbine rotor blades at night than during day, and that more birds collide with rotor blades at night than by day (Exo *et al*. 2003, p. 51).

Avoidance behavior (small-scale maneuvering to avoid collision) is also likely affected by weather, as well as by light conditions. It is reasonable to expect that avoidance rates would be much reduced at times of poor visibility, in poor weather, and at night (Chamberlain *et al.* 2006, p. 199). Red knots' visual acuity may be reduced under very low light conditions (e.g., new or quarter moon) based on their reduced ability to avoid mist nets under such conditions (Burger *et al.* 2011, p. 346).

Lighting on tall structures has been shown to be a significant risk factor in avian collisions (Kuvlesky *et al.* 2007, p. 2488; Manville 2009, entire). Particularly during inclement weather, birds can become disoriented and entrapped in areas of artificially lighted airspace. Although the response of red knots to lighting is not known, red knots are inferred to migrate during both night and day, based on flight durations and distances documented by geolocators (Normandeau Associates, Inc. 2011, p. 203), and lighting is generally required on wind turbines for aviation safety (Federal Aviation Administration 2007, pp. 33–34).

A number of studies from Europe suggest that wind facilities could displace migrating and breeding waterfowl and shorebirds (e.g., due to disturbance), disrupt daily movements (e.g., between foraging and roosting habitats), and disrupt migration activity (Kuvlesky *et al.* 2007, p. 2489). Studies of both onshore and offshore wind-energy facilities in Europe have reported disturbance effects ranging from 246 ft (75 m) to as far as 2,625 ft (800 m) from turbines for various bird groups, including shorebirds (NRC 2007; Drewitt and Langston 2006, p. 32). One study from Europe found that some waterbird species displayed increased avoidance of a wind farm 1.2 to 4.5 mi (2 to 4 km) from the turbines (Kuvlesky *et al.* 2007, p. 2490). Avoidance of wind energy facilities varies among species and depends on site, season, tide, and whether the facility is in operation. Disturbance tends to be greatest for migrating birds while feeding and resting (NRC 2007, p. 108). As with the potential for increasing hurricane frequency or severity (discussed under Asynchronies—Fall Migration, above), extra flying to avoid obstacles during migration represents additional energy expenditure (Niles *et al*. 2010a, p. 129), which could impact red knot survival as well as the timing of arrival at stopover areas (see Asynchronies, above). However, displacement of birds from habitats around wind facilities, as well as large- or small-scale avoidance behaviors, reduces the risks of turbine collisions.

In a literature review of inland, coastal, and offshore studies from Europe and the United States, Stewart *et al.* (2007, pp. 1, 6–9) found clear evidence that waterfowl and shorebirds decline in abundance around wind energy sites, although it is unclear from this study if the negative effect is a decline in population abundance or a decline due to avoidance. As might be expected, the bird taxa involved were correlated with wind farm location; shorebirds were often associated with coastal wind farms (Stewart *et al.* 2007, p. 5). The authors also report that longer operating times result in significantly greater declines in bird abundance than shorter operating times, which suggests that either birds do not become habituated to the presence of wind farms as previously thought, or that local population density declines in spite of habituation (Stewart *et al.* 2007, p. 6). These authors conclude that, while a precautionary approach should be adopted to wind farm development near shorebird aggregations in offshore and coastal locations, the effect of wind farm developments on bird populations must also be viewed in the context of the possible effect of climate change on those same bird populations in the absence of wind farms (Stewart *et al.* 2007, p. 1).

Some studies have found wind turbines cause only low levels of mortality. However, even low levels of mortality may still be significant to long-lived species with low productivity and slow maturation rates, especially for rarer species of conservation concern that already exhibit reduced populations and no longer may have the required redundancy, resiliency, and representation for long-term viability. In such cases, there could be significant effects at the population level (locally or regionally), particularly in situations where cumulative mortality takes place as a result of multiple wind energy installations (Loss *et al.* 2013, p. 208; Robinson-Willmott *et al.* 2013, entire; Watts 2010, entire; Drewitt and Langston 2006, p. 30).

Wind Energy—Offshore

In 2007, BOEM—formerly called the Minerals Management Service (MMS) and the Bureau of Ocean Energy Management, Regulation, and Enforcement (BOEMRE))—established an Alternative Energy and Alternate Use Program for the U.S. OCS, under which BOEM may issue leases, easements, and rights-of-way for the production and transmission of non-oil and -gas energy sources (MMS 2007, p. 2). Since 2009, DOI has developed a regulatory framework for offshore wind projects in Federal waters and launched an initiative to facilitate the siting, leasing, and construction of new projects (DOE and BOEMRE 2011, p. iii). In 2011, the DOE and BOEM released a National Offshore Wind Strategy (National Strategy) that articulates a national goal of 54 gigawatts (GW) of deployed offshore wind-generating capacity by 2030, with an interim target of 10 GW of capacity deployed by 2020. To achieve these targets, the United States would have to reduce the cost of offshore wind energy production and the construction timelines of offshore wind facilities. The National Strategy illustrates the commitment of DOE and DOI to spur the rapid and responsible development of offshore wind energy (DOE and BOEMRE 2011, p. iii).

In addition to these Federal efforts, several States are considering installation of offshore wind turbines in their jurisdictional ocean waters (i.e., up to 3 nautical miles (5.6 km) off the Atlantic coast; variable distances in the Gulf of Mexico) (DOE 2013; Rhode Island Coastal Resources Management Council 2012, p. i). Although New Jersey is pursuing wind projects in State waters (USFWS 2012e, entire), State officials concluded in 2009 that Delaware Bay is not an appropriate site for a large-scale wind turbine project because of potential impacts to shorebirds (NJDEP 2009a, p. 1; NJDEP 2009b, entire). Delaware has plans to document shorebird movement patterns to and from Delaware Bay during the stopover to identify siting locations that will minimize wind turbine impacts to these species (Kalasz 2008, p. 40).

To date, no offshore wind facilities have been installed in U.S. waters. However, BOEM has issued four limited leases under a 2007 Interim Policy: three offshore of New Jersey and one offshore of Delaware (BOEM undated, pp. 1–2). Two of these Interim Policy lessees have deployed meteorological buoys off the coast of New Jersey, while the other two relinquished their leases in 2012. In addition, BOEM anticipates issuing a limited lease during Fiscal Year 2014 for a project proposed offshore of Florida and is considering another Interim Policy lease offshore of Georgia. Leases issued under the Interim Policy have a 5-year term and provide no subsequent rights to commercial development. In contrast to an Interim Policy lease, a commercial lease grants an exclusive right to seek BOEM approval for the development of a leasehold (BOEM undated, pp. 1–2). A commercial lease does not grant the right to construct any facilities; actual construction of offshore renewable energy infrastructure is subject to subsequent phases of BOEM review and approval (BOEM 2012a, p. iii). Since 2009, BOEM has awarded five commercial wind energy leases off the Atlantic coast: two non-competitive leases (for the proposed Cape Wind project in Massachusetts and for an area off Delaware) and three competitive leases totaling more than 277,500 acres in Federal waters (two offshore of Massachusetts/Rhode Island and another offshore Virginia). As of 2014, BOEM had scheduled an another competitive auction (for Wind Energy Areas (WEAs) offshore of Maryland), expected to hold additional competitive auctions for WEAs offshore of Massachusetts and New Jersey in the coming year, announced new WEAs off the coast of North Carolina, and was in the early planning stages for areas offshore of New York (DOI 2014; BOEM undated, p. 1). Although there is considerable uncertainty about where offshore wind energy development is likely to occur on the Atlantic OCS, we conclude that the most likely areas are within existing Interim Policy and commercial leases, and within the WEAs, which are areas that BOEM has identified as areas that appear most suitable for commercial wind energy while presenting the fewest apparent environmental and user conflicts (BOEM 2012a, p. iv).

In addition to progress toward offshore wind energy development in the Atlantic OCS, offshore wind projects have been proposed off the coasts of Texas and Northern Mexico (D. Newstead pers. comm. March 5, 2013). Three turbines currently proposed for construction in the nearshore Gulf of Mexico are considered a pilot project for a future installation of a much larger project over a lease block that could accommodate up to 160 turbines. Because data has shown some red knots cross the Gulf of Mexico, Newstead (2014a, p. 23) concluded that an array of turbine spread over 20,000 acres in the nearshore Gulf of Mexico would present a significant set of obstacles.

In addition, five States have entered an agreement with the Federal government to facilitate wind energy development in the Great Lakes (Council on Environmental Quality 2012, p. 1). In some years, concentrations of red knots (400 to 1,500 birds, but more typically fewer than 100) have been observed at locations around the Great Lakes, and may represent weather-induced stops (eBird.org 2014; Morrison and Harrington 1992, p. 79).

Offshore wind resources within the U.S. range of the red knot show high potential for wind energy development (DOE and BOEMRE 2011, pp. 5–6). The BOEM has worked with State Task Forces to determine offshore wind development locations that are likely to minimize impacts to migratory birds. Areas being considered in the Atlantic OCS are greater than 9 mi (14.5 km) offshore (BOEM 2013a). Siting far offshore may succeed in reducing overall avian collision hazards, although species that rely on the offshore environment for breeding, feeding, or sheltering (e.g., certain seabirds and waterfowl) may have increased exposure risk to turbines farther offshore. Specific to red knots, geolocator results by Burger *et al.* (2012c, p. 373) suggest that collision risk may decrease far offshore, and Burger *et al.* (2011, p. 348) concluded that hazards to red knots from wind energy development likely increase for facilities situated closer to shore, particularly near bays and estuaries that serve as major stopover or wintering areas. Risks may increase near to shore because red knot flights in these areas are likely more frequent and at lower altitudes, and birds may be at higher risk during takeoff and landing (Burger *et al*. 2012c, pp. 374–376; Burger *et al*. 2011, p. 346).

Despite BOEM's efforts to site wind energy areas far offshore, however, red knot use areas delineated by Burger *et al.* (2012c, p. 373) based on geolocator results do appear to have some overlap with the both the offshore commercial wind leases executed to date and the WEAs where BOEM will focus future leases. These areas of overlap include an executed lease and a WEA roughly 14 and 27 miles, respectively, off the mouth of Delaware Bay (BOEM 2013b; BOEM undated, p. 1).

In addition to many site- and species-specific factors, avian collision risks are related to both the total number of turbines and the height of the turbines (Loss *et al.* 2013, pp. 206–207; Kuvlesky *et al.* 2007, p. 2488; Chamberlain *et al*. 2006, p. 198). Increasing power output per turbine is key to reducing the cost of offshore wind energy generation, necessitating the development of larger turbines (DOE and BOEMRE 2011, p. 15). The first BOEM-approved offshore wind facility, Cape Wind Energy, will include 130, 3.6-megawatt (MW) wind turbines, each with a maximum blade height of 440 ft (134 m) above sea level off the southeast coast of Massachusetts (BOEM 2012b, p. 1). The DOE and BOEM envision the height of offshore turbines increasing to 617 ft (188 m) above sea level for 8-MW turbines by 2020, and to 681 ft (207.5 m) above sea level for 10-MW turbines by 2030 (DOE and BOEMRE 2011, p. 15). Using a range of 3.6 to 10 MW of generating capacity per turbine, the national goal of 54 GW would require between 5,400 and 15,000 turbines to be installed in U.S. waters.

Buildout (when all available sites are either developed or restricted) of the wind industry offshore of the Atlantic coast will result in the largest network of overwater avian hazards ever constructed, adding a new source of mortality to many bird populations (Watts 2010, p. 1), some of which can little tolerate further reductions before realizing population-level effects. Watts (2010, p. 1) used a form of harvest theory called Potential Biological Removal to develop a population framework for estimating sustainable limits on human-induced bird mortality. Enough information was available from the literature for 46 nongame waterbird species to allow for estimates of sustainable mortality limits from all human-caused sources. Among these 46 populations, red knot stood out as having particularly low mortality limits (Watts 2010, p. 1).

Using an estimated rangewide population size of 20,000 red knots, Watts (2010, p. 39) estimated that human-induced direct mortality exceeding 451 birds per year would start to cause population declines. This estimate of 451 birds per year could change with the use of updated estimates of population size (see "Population Surveys and Estimates" under Species Information) and survival (e.g., Schwarzer *et al.* 2012, p. 729; McGowan *et al.* 2011a, p. 13). While the Watts (2010, p. 39) model underscores the vulnerability of red knot populations to direct human-caused mortality from any source (see also Oil Spills and Leaks, Harmful Algal Blooms, and Factor B, above), information regarding avian collision mortality at offshore wind farms is very limited, largely due to the difficulties of detecting collisions at sea (Drewitt and Langston 2006, p. 32). Therefore, we have only preliminary information on the actual red knot collision risk posed by offshore wind turbines (e.g., based on behavioral avoidance capacity, flight altitudes, migration routes). Best available data regarding these risk factors are presented below, but are currently insufficient to estimate the likely annual mortality of red knots per turbine, or total mortality upon buildout of offshore wind infrastructure.

Burger *et al.* (2011, pp. 341–342) used a weight-of-evidence approach to examine the risks and hazards from offshore wind development on the OCS for three species of coastal waterbirds, including red knot. Three levels of exposure were identified: micro-scale (whether the species is likely to fly within the rotor swept area, governed by behavioral avoidance abilities); meso-scale (occurrence within the rotor swept zone or hazard zone, governed by flight altitude); and macro-scale (occurrence of species within the geographical areas of interest). Regarding micro-scale exposure, little is known about the red knot's abilities to behaviorally avoid turbine collisions (Burger *et al.* 2011, p. 346), an important factor in determining collision risk (Chamberlain *et al.* 2006, p. 198). The red knot's visual acuity and maneuverability are known to be good, but no actual interactions with wind turbines have been observed. The red knot's ability to avoid turbines, even if normally good, could be reduced in poor visibility, high winds, or inclement weather. Avoidance may be more difficult upon descent after long migratory flights than on ascent (Burger *et al.* 2011, p. 346).

Regarding meso-scale exposure, the migratory flight altitude of red knots remains unknown (Normandeau Associates, Inc. 2011, p. 203). However, some experts estimate the normal cruising altitude of red knots during migration to be in the range of 3,281 to 9,843 ft (1,000 to 3,000 m), well above the estimated height of even a 10-MW turbine (681 ft; 207.5 m). However, much lower flight altitudes may be expected when red knots encounter bad weather or high winds, on ascent or descent from long-distance flights, during short-distance flights if they are blown off course, during short coastal migration flights, or during daily commuting flights (e.g., between foraging and roosting habitats) (Burger *et al.* 2012c, pp. 375–376; Burger *et al.* 2011, p. 346). As judged by tree heights, Burger *et al.* (2012c, p. 376) observed knots flying at heights of up to 400 ft (120 m) when flying away from disturbances and when moving between foraging and roosting areas. Based on observations of ruddy turnstones and other *Calidris canutus* subspecies departing from Iceland towards Nearctic breeding grounds in spring 1986 to 1988, Alerstam *et al.* (1990, p. 201) found that departing shorebirds climbed steeply, often by circling and soaring flight, with an average climbing rate of 3.3 ft per second (1.0 m per second) up to altitudes of 1,969 to 6,562 ft (600 to 2,000 m) above sea level. With unfavorable winds, the shorebirds descended to fly low over the sea surface (Alerstam *et al.* 1990, p. 201).

Based on studies of visible daytime migration in Germany, covering the lowest 656 to 984 ft (200 to 300 m) above the sea, 37 percent of shorebirds flew higher than 164 ft (50 m) (Exo *et al.* 2003, p. 51). However, as high-flying shorebirds are difficult to spot and low-flying shorebirds are mainly seen in headwinds, when many sea and coastal birds fly in lower altitudes than in tailwinds, the real percentages of high-flying shorebirds is probably much higher. In general, most migrating shorebirds tend to fly at greater heights. But, when moving between high-tide roosts and low-tide feeding areas, shorebirds generally fly at altitudes below 328 ft (100 m) (Exo *et al.* 2003, p. 51). These findings are not specific to red knots, but are consistent with Burger *et al.* (2012c) and reinforce the importance of weather conditions in affecting shorebird migration. Though not specific to shorebirds, Hüppop *et al*. (2006, p. 106) reported that many birds seem to migrate in much lower altitude strata at sea than they do over land.

Regarding macro-scale exposure, red knot migratory crossings of the Atlantic OCS are likely to occur broadly throughout this ocean region, with possible concentrations south of Cape Cod in fall and south of Delaware Bay in spring (Normandeau Associates, Inc. 2011, p. 201). Shorter-distance migrants (e.g., those wintering in the Southeast) were initially thought to be at lower risk of collision with offshore turbines, particularly turbines located far off the coast such as in the OCS (Burger *et al.* 2011, pp. 346, 348). However, information from nine geolocator tracks showed that both short-distance and long-distance (e.g., birds wintering in South America) migrants crossed the OCS at least twice per year, with some birds crossing as many as six times. These numbers reflect only long flights, and many more crossings of the OCS may occur as red knots make shorter flights between states (Burger *et al.* 2012c, p. 374). The reverse migration exhibited by some red knots within some stopover regions (i.e., southbound movements during spring on the order of 100 mi (160 km) (D'Amico *et al.* 2011, entire; Watts 2009a)) means that at least some birds may pass over a particular section of offshore waters multiple times per migration season, thereby increasing the collision risk for those individuals if turbines are present (Hüppop *et al*. 2006, p. 104). However, it is unknown how close to the coast or at what altitudes such flights tend to occur.

Geolocator results suggest that short-distance migrants may actually face greater collision hazards from wind development in the Atlantic OCS. Six birds that wintered in the Southeast spent an average of 218 days (60 percent of the year) migrating, stopping over, or wintering on the U.S. Atlantic coast, while 3 birds that wintered in South America spent only about 22 days (about 6 percent of the year) in this region (Burger *et al.* 2012c, p. 374). Thus, long-distance migrants may spend less time exposed to turbines built off the U.S. Atlantic coast.

South of the Atlantic coast stopovers, red knots' migratory pathways may be either coast-following, OCS-crossing, or a mixture of both (Normandeau Associates, Inc. 2011, p. 202). While some extent of coast-following is likely to occur, studies to date suggest that a large fraction of the population is likely to cross the OCS at significant distances offshore (e.g., to follow direct pathways between widely separated migration stopover points) (Burger *et al.* 2012c, p. 376; Normandeau Associates, Inc. 2011, p. 202). Based on the red knot's life history and geolocator results to date, macro-scale exposure of red knots to wind facilities is likely to be widely but thinly spread over the Atlantic OCS (Normandeau Associates, Inc. 2011, p. 202). Hazards to red knots from wind energy development likely increase for facilities situated closer to shore, particularly near bays and estuaries that serve as major stopover or wintering areas (Burger *et al.* 2011, p 348).

Although exposure of red knots to collisions with offshore wind turbines is broad geographically, exposure is much more restricted temporally, occurring mainly during brief portions of the spring and fall migration when long migratory flights occur over open water (Normandeau Associates, Inc. 2011, p. 202). The rest of the red knot's annual cycle is largely restricted to coastal and near-shore habitats (Normandeau Associates, Inc. 2011, p. 202), during which times collision hazards with land-based turbines (discussed below) would represent a greater hazard than for turbines in the offshore environment.

Taking advantage of the limited temporal exposure of migrating birds to offshore turbine collisions, the authorization for one offshore wind facility in New Jersey's State waters includes operational shutdowns during certain months when red knots and two federally listed bird species (piping plovers and roseate terns) may be present. The shutdowns would occur only during inclement weather conditions (USFWS 2012e, p. 3) that may prompt lower migration altitudes and hinder avoidance behaviors.

Robinson-Willmott *et al.* (2013, entire) ranked the relative vulnerability of 177 migratory bird species to collision and displacement from offshore wind turbines on the Atlantic OCS. The relative collision vulnerability of "red knot" was "medium" and the relative vulnerability to displacement was "lower" (Robinson-Willmott *et al.* 2013, pp. 52, 70). These authors used a mathematical formula to calculate population sensitivity scores, which were then used as inputs to both the collision and displacement vulnerability assessments (Robinson-Willmott *et al.* 2013, pp. 7, 20, 24). The population sensitivity formula includes four variables: global population size, proportion of the population in the OCS, threat ranking, and survival ranking (Robinson-Willmott *et al.* 2013, pp. 7). For three of these four variables (all but the global population size), the data utilized for "red knot" represent the global population across all subspecies of *Calidris canutus* and are not specific to *C.c. rufa* (J. Robinson-Willmott pers. comm. April 8, 2014). Recalculation specific to *C.c. rufa* would likely show a higher population sensitivity, which may in turn affect the collision and displacement vulnerability numerical scores. Nonetheless, as strictly qualitative descriptors, we do not disagree with "medium" (collision) and "lower" (displacement) vulnerability characterizations for rufa red knot on the Atlantic OCS. "Medium" may be a suitable characterization of collision vulnerability based on the preceding analysis, particularly considering the findings of Burger *et al.* 2012c, Burger *et al.* 2011, Normandeau Associates, Inc. 2011, Watts 2010, and Alerstam *et al.* 1990. "Lower" may be a suitable characterization of offshore displacement vulnerability because red knots do not rely on offshore environments to roost or feed; thus, although an occasional low-flying flock may expend additional energy to avoid turbines, we would not expect red knots to be displaced from important habitats due to the future development of offshore wind energy facilities (unless onshore transmission line connections or other supporting infrastructure are located near red knot habitats). We note that, while these "medium" (collision) and "lower" (displacement) descriptors may be appropriate far offshore, they may understate red knot vulnerability to offshore turbine development closer to the coasts (Burger *et al.* 2012c; Burger *et al.* 2011; Alerstam *et al.* 1990).

In addition to collision and displacement hazards, environmental effects of an offshore energy facility may also stem from associated infrastructure including the land based connections to the electrical grid. If located in important migration or wintering areas, this associated infrastructure could affect red knots through habitat modification or disturbance of birds on land.

Wind Energy—Terrestrial

The number of land-based wind turbines installed within the U.S. range of the red knot has increased substantially in the past decade (table 27). As of 2009, estimates of total avian mortality at U.S. turbines ranged from 58,000 to 440,000 birds per year, and were associated with high uncertainty due to inconsistencies in the duration and intensity of monitoring studies (Manville 2009, p. 268). A newer estimate considering only monopole-supported turbines puts this estimate of total U.S. avian mortality at 140,000 to 328,000 (Loss *et al*. 2013, p. 201). Although measures have been taken to increase the analytical rigor of collision studies, accessibility to relevant data remains a problem, particularly for the tallest (greater than 262 ft (80 m)) turbines, because most of the mortality data are in industry reports that are not subjected to scientific peer review or available to the public (Loss *et al*. 2013, pp. 201, 202, 207).

Table 27. Installed wind energy generation capacity by State within the U.S. range of the red knot (including interior migration pathways), 1999 and 2012 (DOE 2012). U.S. average turbine size was 1.97 MW in 2011, up from 0.89 MW in 2000 (University of Michigan 2012, p. 1). We divided the megawatts by these average turbine sizes to estimate the numbers of turbines.

	1999		2012	
State	**Megawatts**	**Estimated Number of Turbines**	**Megawatts**	**Estimated Number of Turbines**
Alabama	0.000	0	0	0
Arkansas	0.000	0	0	0
Colorado	21.600	24	2,301	1,168
Connecticut	0.000	0	0	0
Delaware	0.000	0	2	1
Florida	0.000	0	0	0
Georgia	0.000	0	0	0
Illinois	0.000	0	3,568	1,811
Indiana	0.000	0	1,543	783
Iowa	242.420	272	5,137	2,608
Kansas	1.500	2	2,712	1,377
Kentucky	0.000	0	0	0
Louisiana	0.000	0	0	0
Maine	0.100	0	431	219
Maryland	0.000	0	120	61
Massachusetts	0.300	0	100	51
Michigan	0.600	1	988	502
Minnesota	273.390	307	2,986	1,516
Mississippi	0.000	0	0	0
Missouri	0.000	0	459	233
Montana	0.100	1	645	327

Table 27. Installed wind energy generation capacity by State within the U.S. range of the red knot (including interior migration pathways), 1999 and 2012 (DOE 2012). U.S. average turbine size was 1.97 MW in 2011, up from 0.89 MW in 2000 (University of Michigan 2012, p. 1). We divided the megawatts by these average turbine sizes to estimate the numbers of turbines.

	1999		2012	
State	**Megawatts**	**Estimated Number of Turbines**	**Megawatts**	**Estimated Number of Turbines**
Nebraska	2.820	3	459	233
New Hampshire	0.050	0	171	87
New Jersey	0.000	0	9	5
New York	0.000	0	1,638	831
North Carolina	0.000	0	0	0
North Dakota	0.390	1	1,679	852
Ohio	0.000	0	426	216
Oklahoma	0.000	0	3,134	1,591
Pennsylvania	0.130	1	1,340	680
Rhode Island	0.000	0	9	5
South Carolina	0.000	0	0	0
South Dakota	0.000	0	784	398
Tennessee	0.000	0	29	15
Texas	183.520	206	12,212	6,199
Vermont	6.050	7	119	60
Virginia	0.000	0	0	0
West Virginia	0.000	0	583	296
Wisconsin	22.980	26	649	329
Wyoming	72.515	81	1,410	716
Total	828.465	931	45,643	23,169

In 2008, DOE released a report to investigate the feasibility of achieving 20 percent of U.S. electricity from wind by 2030 (DOE 2008, p. 1). The 20 percent wind scenario envisions 251 GW of land-based generation in addition to 54 GW of shallow-water offshore production (DOE 2008, p. 10). Using an average capacity of 2 MW per turbine (University of Michigan 2012, p. 1), a 251-GW target would require about 125,500 turbines. The DOI strongly supports renewable energy, including wind development, and the Service works to ensure that such development is bird- and habitat-friendly (Manville 2009, p. 268). In 2012, the Service updated earlier (2003) voluntary guidelines to provide a structured, scientific process for addressing wildlife conservation concerns at all stages of land-based wind energy development (USFWS 2012d, p. vi).

Although avian impacts from land-based wind turbines are generally better documented than in the offshore environment, relatively little shorebird-specific information is available. Compiling estimated mortality rates from nine U.S. wind facilities (including four in California), Erickson *et al.* (2001, pp. 2, 37, 39) calculated an average of 2.19 avian fatalities per turbine per year for all bird species combined (1.825 fatalities per turbine excluding California), and found that shorebirds constituted only 0.2 percent of the total (0.7 percent excluding California). Compiling 18 studies around the Great Lakes from 1999 to 2009, Akios (2011, pp. 9–10) found that mortality estimates for all species combined ranged from 0.4 to nearly 14 birds per turbine per year. Shorebirds accounted for 4.3 percent of the total at inland sites (nine studies at six sites), but accounted for only about 1.5 percent of the total at sites closer to the lakeshores (five studies at four sites) (Akios 2011, p. 14). Studies from Europe and New Jersey also suggest generally low collision susceptibility for shorebirds at coastal wind turbines (Normandeau Associates, Inc. 2011, p. 201).

Niemuth *et al.* (2013, p. 263) studied wetlands within 2,641 ft (805 m) of a wind turbine at two wind energy sites in the prairie pothole region in North and South Dakota. Compared to reference wetlands, occurrence of three shorebird species (not including red knot) was slightly but consistently lower at one of the two wind energy sites. These authors conclude that wetlands have conservation value for these species when wind turbines are present, but additional sampling is necessary to understand the effects of wind turbines on shorebirds (Niemuth *et al.* 2013, p. 263). Although it may be an overall adverse effect, displacement of birds from habitats around wind facilities somewhat reduces the risks of turbine collisions.

For all bird taxa combined, Loss *et al*. (2013, p. 206) found higher annual U.S. collision mortality rates than the 2.19 reported by Erickson *et al.* (2001, pp. 2, 37), based on a review of 68 studies. This review considered only monopole-supported turbines because lattice-supported turbines are largely being decommissioned due to higher avian collision hazards. These authors found mortality rate differences by region: 6.86 birds per turbine in the East, 2.92 birds per turbine in the Great Plains, and 4.72 birds per turbine in the West (excluding California) (Loss *et al.* 2013, p. 206).

Even in coastal states, most of the wind capacity installed to date is located along interior ridgelines or other areas away from the coast. With operations starting in 2005 (Atlantic County Utilities Authority 2012, p. 1), the 7.5-MW Jersey Atlantic Wind Farm was the first coastal wind farm in the United States (New Jersey Clean Energy Program undated). Located outside of Atlantic City, New Jersey (about 2 mi (3.2 km) inland from the nearest sandy beach, and surrounded by tidal marsh), the facility consists of five 380-ft (116-m) turbines (Atlantic County Utilities Authority 2012, p. 1). The NJAS (2009, entire; 2008a, entire; 2008b, entire) reported raw data from carcass searches conducted around the turbines. These figures have not yet been adjusted for observer efficiency, scavenger removal, or lack of searching in restricted-access areas, all of which would increase estimates of collision mortality (NJAS 2009, p. 2). In 3 years of searching, 38 carcasses from 25 species were attributed to turbine collision (NJAS 2009, pp. 2–3), or about 2.5 collisions per turbine per year. Of these, three carcasses (about eight percent) were shorebirds, and none were red knots (NJAS 2009, p. 3; NJAS 2008a, p. 5; NJAS 2008b, p. 9).

Considerable wind facility development has occurred in recent years near the Texas coast, south of Corpus Christi, and in the Mexican State of Tamaulipas; many additional wind energy projects are proposed in this region (D. Newstead pers. comm. March 5, 2103). As of 2011, coastal wind installations in Texas totaled more than 1,200 MW, or about 13 to 15 percent of the Statewide total (Reuters 2011). Kuvlesky *et al.* (2007, pp. 2487, 2492–2493) identified the lower Gulf coast of Texas as a region where wind energy development may have a potentially negative effect on migratory birds. Onshore wind energy development in the area of Laguna Madre may expose red knots to direct and indirect impacts during daily or seasonal movements (D. Newstead pers. comm. March 5, 2013). Geolocator data from Texas suggests that knots have a fairly specific flight vector upon departure toward their northbound stopover (Newstead 2014a, p. 22). Shorebirds departing the coast for destinations along the Central Flyway (see Migration—Midcontinent) may be at some risk from wind projects throughout the flyway, but especially those that are adjacent to the coast where birds on a northbound departure may not have reached sufficient altitude to clear turbine height before reaching migration altitude (D. Newstead pers. comm. March 5, 2013).

Based on the higher frequency and lower altitudes of red knot flights along the coasts, as well as the coastal location of most large, known U.S. nonbreeding red knot roosting and foraging areas (D. Newstead pers. comm. March 5, 2103; Burger *et al.* 2012c, pp. 375–376; Burger *et al.* 2011, p. 346; Stewart *et al.* 2007, p. 1; Alerstam *et al.* 1990, p. 201), we conclude that collision and displacement risks per turbine (notwithstanding differences in specific factors such as turbine size, design, operation, siting) are likely higher along the coasts than in either far offshore or far inland areas. Mapping for the United States shows coastal turbines concentrated in south Texas and the Northeast (USGS 2014).

Wind Energy—Summary

We analyzed shorebird mortality at land-based wind turbines in the United States (Akios 2011, p. 14; Erickson *et al.* 2001, pp. 2, 37, 39), and we considered the red knot's vulnerability factors for collisions with offshore wind turbines that we expect will be built in the next few decades (Burger *et al.* 2011, entire). We have minimal information regarding wind energy development in other countries. Based on our analysis of wind energy development in the United States, we expect ongoing improvements in turbine siting, design, and operation will help minimize bird collision hazards (USFWS 2012d, p. vi). However, we also expect cumulative avian collision mortality to increase through 2030 as the number of turbines continues to grow, and as wind energy development expands into coastal and offshore environments (DOE 2008, p. 1). Shorebirds as a group have constituted only a small percentage of collisions with U.S. turbines in studies conducted to date (Akios 2011, p. 14; NJAS 2009, p. 3; NJAS 2008a, p. 5; NJAS 2008b, p. 9; Erickson *et al.* 2001, pp. 2, 37, 39), but wind development along the coasts (where shorebirds might be at greater risk) did not begin until 2005 (New Jersey Clean Energy Program undated). Based on the higher frequency and lower altitudes of red knot flights along the coasts, as well as the coastal location of most well-known U.S. nonbreeding red knot roosting and foraging areas, we conclude that collision and displacement risks per turbine (notwithstanding differences in specific factors such as turbine size, design, operation, siting) are likely higher along the coasts (both on land and nearshore) than in areas either far offshore or far inland (D. Newstead pers. comm. March 5, 2103; Burger *et al.* 2012c, pp. 375–376; Burger *et al.*

2011, p. 346; Stewart *et al.* 2007, p. 1; Alerstam *et al.* 1990, p. 20). Likewise, hazards to red knots from offshore wind energy development likely increase for facilities situated closer to shore, particularly near bays and estuaries that serve as major stopover or wintering areas (Burger *et al.* 2012c, p. 373; Burger *et al.* 2011, p 348).

We are not aware of any documented red knot mortalities at any wind turbines to date, but low levels of red knot mortality from turbine collisions may be occurring now based on the number of turbines along the red knot's migratory routes (table 27) and the frequency with which red knots traverse these corridors. Based on the current number and geographic distribution of turbines, if any such mortality is occurring, it is likely not causing subspecies-level effects. However, our primary concern is that, as buildout of wind energy infrastructure progresses, especially near the coast, increasing mortality from turbine collisions may contribute to a subspecies-level effect due to the red knot's modeled vulnerability to direct human-caused mortality (Watts 2010, p. 1). We anticipate that the threat to red knots from wind turbines will be primarily related to collision or behavioral changes during migratory or daily flights. Unless facilities are constructed at key stopover or wintering habitats, we do not expect wind energy development to cause significant direct habitat loss or degradation or displacement of red knots from otherwise suitable habitats.

Factor E—Conservation Efforts

There are many components of Factor E, some of which are being partially managed through conservation efforts. For example, the reduced availability of horseshoe crab eggs from the past overharvest of crabs in Delaware Bay is currently being managed through the ASMFC's ARM framework (see Factor E—Reduced Food Availability and Factor D). This conservation effort more than others is likely having the greatest effect on the red knot subspecies as a whole because a large majority of the birds move through Delaware Bay during spring migration and depend on a superabundant supply of horseshoe crab eggs for refueling. Other factors potentially influencing horseshoe crab egg availability are outside the scope of the ARM, but some are being managed. For example, enforcement is ongoing to minimize poaching, and steps are being implemented to prevent the importation of nonnative horseshoe crab species that could impact native populations. Despite the ARM and other conservation efforts, horseshoe crab population growth has stagnated for unknown reasons, some of which (e.g., possible ecological shifts) may not be manageable. See Factor A regarding threats to, and conservation efforts to maintain, horseshoe crab spawning habitat.

Some threats to the red knot's other prey species (mainly mollusks) are being partially addressed. For example, the Service is working with partners to minimize the effects of shoreline stabilization projects on the invertebrate prey base for shorebirds (e.g., Rice 2009, entire), and management of ORVs is protecting the invertebrate prey resource in some areas. Other likely threats to the red knot's mollusk prey base (e.g., ocean acidification; warming coastal waters; marine diseases, parasites, and invasive species) cannot be managed at this time, although efforts to minimize ballast water discharges in coastal areas likely reduce the potential for introduction of new invasive species.

Other smaller-scale conservation efforts implemented to reduce Factor E threats include beach recreation management to reduce human disturbance, gull species population monitoring and management in Delaware Bay, research into HAB control, oil spill response plan development and implementation, sewage treatment in Río Gallegos (Argentina), and national and state wind turbine siting and operation guidelines. In contrast, no known conservation actions are available to address asynchronies during the annual cycle.

Factor E—Summary

Factor E includes a broad range of threats to the red knot. Reduced food availability at the Delaware Bay stopover site due to commercial harvest of the horseshoe crab is considered a primary causal factor in the decline of rufa red knot populations in the 2000s (Escudero *et al.* 2012, p. 362; McGowan *et al.* 2011a, pp. 12–14; Niles *et al.* 2008, pp. 1–2; Baker *et al.* 2004, p. 875). Under the current management framework (the ARM), the present horseshoe crab harvest is not considered a threat to the red knot. However, continued implementation of the ARM is imperiled by lack of funding to support the requisite monitoring programs. With or without the ARM, it is not yet known if the horseshoe crab egg resource will continue to adequately support red knot population growth over the next decade. Notwithstanding the importance of the horseshoe crab and Delaware Bay, the red knot faces a range of ongoing and emerging threats to its food resources throughout its range, including small prey sizes from unknown causes (Escudero *et al.* 2012, pp. 359–362; Espoz *et al.* 2008, pp. 69, 74), warming water and air temperatures (Jones *et al.* 2010, pp. 2255–2256), ocean acidification (IGBP *et al.* 2013, pp. 9, 16; NRC 2010b, pp. 68–69), physical habitat changes (Chapter 5 in IPCC 2014, p. 21; Rehfisch and Crick 2003, p. 88; Najjar *et al.* 2000, p. 225), possibly increased prevalence of disease and parasites (Ward and Lafferty 2004, p. 543), marine invasive species (Seebens *et al.* 2013, p. 782; Ruesink *et al.* 2005, pp. 671–674; Grosholz 2002, p. 22–23), and burial and crushing of invertebrate prey from sand placement and recreational activities (Sheppard *et al.* 2009, p. 113; Schlacher *et al.* 2008b, pp. 345, 348; Schlacher *et al.* 2008c, pp. 878, 882; Greene 2002, p. 24).

In addition, the red knot's life-history strategy makes this species inherently vulnerable to mismatches in timing between its annual cycle and those periods of optimal food and weather conditions upon which it depends (Galbraith *et al.* 2014, p. 7 and Supplement 1; Liebezeit *et al.* 2014, p. 2; Conklin *et al.* 2010, p. 4; Gill *et al.* 2013, p. 1; Hurlbert and Liang 2012, pp. 4–5; McGowan *et al*. 2011a, pp. 2, 16; Smith *et al.* 2011a, p. 575; Meltofte *et al.* 2007, p. 36). The red knot's sensitivity to timing asynchronies has been demonstrated through a population-level response, as the late arrivals of birds in Delaware Bay is generally accepted as a key causative factor (along with reduced supplies of horseshoe crab eggs) behind population declines in the 2000s (Baker *et al*. 2004, p. 878). The factors that caused delays in the spring migrations of red knots from Argentina and Chile are still unknown (Niles *et al*. 2008, p. 2), and we have no information to indicate if this delay will reverse, persist, or intensify in the future. Superimposed on the existing threat of late arrivals in Delaware Bay are new threats emerging due to climate change (Summary for Policymakers in IPCC 2014, p. 30; Root *et al.* 2013, pp. 85–88; Hurlbert and Liang 2012, p. 4), such as changes in the timing of reproduction for both horseshoe crabs and mollusks (Burrows *et al.* 2011, p. 652; Poloczanska *et al.* 2013, pp. 3–4; Smith *et al.* 2010b, p. 563; van Gils *et al.* 2005a, p. 2615; van Gils *et al.* 2005b, pp. 126–127; Philippart *et al.* 2003, p. 2171). Climate change may also cause shifts in the period of optimal arctic insect and snow conditions relative to the time period when red knots currently breed (Grabowski *et al.* 2013, p. 1097; McGowan *et al*. 2011a, p. 13; Smith *et al*. 2010a, p. 292; Tulp and Schekkerman 2008, p. 48; Meltofte *et al.* 2007, pp. 7, 25; Piersma *et al.* 2005, p. 270; Schekkerman *et al.* 2003, p. 340).

The red knot's adaptive capacity to deal with numerous changes in the timing of resource availability across its geographic range is largely unknown (Liebezeit *et al.* 2014, pp. 1, 10; Grabowski *et al.* 2013, p. 1103; Meltofte *et al.* 2007, p. 34). A few examples suggest some flexibility in red knot migration strategies (D. Newstead pers. comm. May 8, 2014; Grabowski *et al.* 2013, pp. 1097, 1100–1103; Smith *et al.* 2010a, p. 292; González *et al.* 2006, p. 115; González *et al.* in IWSG 2003, p. 18), but differences between the annual timing cues of red knots (at least partly celestial and endogenous) (Liebezeit *et al.* 2014, p. 10; Conklin *et al.* 2010, p. 5; Gill *et al.* 2013, p. 1; McGowan *et al.* 2011a, p. 16; Cadée *et al.* 1996, p. 82) and their prey (primarily environmental) (Smith *et al.* 2010b, p. 563; Philippart *et al.* 2003, p. 2171) suggest there are limitations on the adaptive capacity of red knots to cope with increasing frequency or severity of asynchronies.

Other factors are likely to exacerbate the effects of reduced prey availability and asynchronies, including human disturbance (Burger and Niles 2013a, p. 23; Burger and Niles 2013b, p. 657; Escudero *et al.* 2012, pp. 358, 362), competition with gulls (Niles *et al.* 2008, p. 107; Burger *et al.* 2007, p. 1162), and behavioral changes from wind energy development (Kuvlesky *et al.* 2007, p. 2489). Additional factors are likely to increase the levels of direct red knot mortality, such as HABs (Newstead 2014a, p. 23; Anderson 2007, p. 2), oil spills (Anderson *et al.* 2012, p. 10; WHSRN 2012; Kalasz 2008, pp. 39–40; Niles *et al.* 2008, p. 98, 100), and collisions with wind turbines (D. Newstead pers. comm. March 5, 2013; Burger *et al.* 2012c, p. 370; Burger *et al.* 2011, p. 348; Watts 2010, p. 1; Kuvlesky *et al.* 2007, p. 2487). In addition to elevating background mortality rates, these three factors pose the potential for a low-probability but high-impact event if a severe HAB or major oil or contaminant spill occurs when and where large numbers of red knots are present, or if a mass-collision event occurs at wind turbines during migration. Based on our review of the best available scientific and commercial data, the subspecies-level impacts from Factor E components are already occurring and are anticipated to continue and possibly increase into the future.

CUMULATIVE EFFECTS FROM FACTORS A THROUGH E

Cumulative means an increase in quantity, degree, or force by successive addition. Synergy means the interaction of elements that, when combined, produce a total effect that is greater than the sum of the individual elements. Red knots face a wide range of threats across their range on multiple geographic and temporal scales. The effects of some smaller threats may act in an additive fashion to ultimately impact populations or the subspecies as a whole (cumulative effects). Other threats may interact synergistically to increase or decrease the effects of each threat relative to the effects of each threat considered independently (synergistic effects).

An example of cumulative effects comes from local or regional sources of typically low-level but ongoing direct mortality, such as from hunting, normal levels of parasites and predation, stochastic weather events, toxic HAB events, oil pollution, and collisions with wind turbines. We have no evidence that any of these mortality sources individually are impacting red knot populations, but taken together, the cumulative effect of these threats may potentially aggravate population declines, or slow population recoveries, particularly since modeling has suggested that the red knot is inherently vulnerable to direct human-caused mortality (Watts 2010, p. 39). Red knots by nature flock together within wintering areas and at critical migration stopovers. (At study sites in New Jersey, mixed-species flocks that included red knots were larger than flocks without knots (Burger and Niles 2013a, p. 23.)) Surveys indicate that red knot populations using Tierra del Fuego and Delaware Bay have decreased by about 75 percent since the 1980s. As a result, flocks of hundreds or thousands of birds now represent a greater proportion of the total red knot population than in the past. Natural or anthropogenic stochastic events affecting flocks (e.g., storms, oil spills, red tides, hunting) can, therefore, be expected to have a greater impact on the red knot subspecies as a whole than in the past.

An example of a localized synergistic effect is increased beach cleaning following a storm, HAB event, or oil spill. Red knots and their habitats can be impacted by both the initial event, and then again by the cleanup activities. Sometimes such response efforts are necessary to minimize the birds' exposure to toxins, but nonetheless cause further disturbance and possibly alter habitats (e.g., N. Douglass pers. comm. December 4, 2006). Where storms occur in areas with hard stabilization structures, they are likely to cause net losses of habitat. In a synergistic effect, these same storms can also trigger or accelerate human efforts to stabilize the shoreline (e.g., NJDEP 2014; T. Pover pers. comm. March 14, 2014; Christie 2013; Regional Plan Association 2012, p. 1), further affecting shorebird habitats as discussed under Factor A. In addition to causing direct mortality and prompting human response actions, storm, oil spill, or HAB events can interact synergistically with several other threats, for example, exacerbating ongoing problems with habitat degradation or food availability through physical or toxic effects on habitat or prey species. Another possible synergistic effect involves the rapid and sustained decrease in arctic sea ice that may open up the red knot's breeding range to commercial shipping by mid-century (Smith and Stephenson 2013, p. 1), possibly including shipping of petroleum products and development of supporting infrastructure that would bring risks of oil spills or leaks. Reductions in Arctic sea ice will bring new challenges as navigation routes for commercial shipping open and marine access to the region for offshore oil and gas development, tourism, fishing and other activities (NRC 2013, p. 4).

Modeling the effect of winds on migration in *Calidris canutus canutus,* Shamoun-Baranes *et al.* (2010, p. 285) found that unpredictable winds affect flight times and that wind is a predominant driver of the use of an intermittently used emergency stopover site. This study points to the interactions between weather and habitat. The somewhat uncertain but nevertheless likely threat to red knots from changing frequency, intensity, geographic paths, or timing of coastal storms could have a synergistic effect with loss or degradation of stopover habitats (e.g., changing storm patterns could intensify the red knot's need for a robust network of stopover sites). Likewise, encounters with more frequent, severe, or aberrant storms during migration might not only exact some direct mortality and the energetic costs (to survivors) of extra flight miles, but also could induce red knots to increase their use of stopover habitats in areas where shorebird hunting is still practiced (Nebel 2011, p. 217).

Reduced food availability has also been shown to interact synergistically with asynchronies and several other threats. Escudero *et al.* (2012, p. 362) have suggested that declining prey quality in South American wintering areas may be a partial explanation for the increasing proportion of red knots arriving late in Delaware Bay in the 2000s. In turn, the best available data indicate that late arrivals in Delaware Bay were a key factor that acted synergistically with depressed horseshoe crab egg supplies, and together these two factors constitute the most well-supported explanation for red knot population declines in the 2000s (Niles *et al*. 2008, p. 2; Atkinson *et al.* 2007, p. 892; Baker *et al.* 2004, p. 878; Atkinson *et al.* 2003b, p. 16). Further synergistic effects in Delaware Bay affecting red knot weight gain have also been noted among food availability, ambient weather, storms, habitat conditions, and competition with gulls (Dey *et al.* 2011a, p. 7; Breese 2010, p. 3; Niles *et al.* 2005, p. 4). Philippart *et al.* (2003, p. 2171) concluded that prolonged periods of lowered bivalve recruitment and stocks due to rising water temperatures may lead to a reformulation of estuarine food webs and possibly a reduction of the resilience of the system to additional disturbances, such as shellfish harvest. Modeling by van Gils *et al.* (2005a, p. 2615) showed that, by selecting stopovers containing high-quality prey, *Calidris canutus* of various subspecies kept metabolic rates at a minimum, potentially reducing the spring migratory period by a full week; thus, not only can asynchronies cause red knots to arrive when food supplies are suboptimal, but so can suboptimal prey quality at a stopover cause an asynchrony for the next leg of the migratory journey (e.g., by delaying departure until adequate weight has been gained).

As discussed under Factor E—Reduced Food Availability, there are processes associated with ocean acidification and ocean warming that can combine to amplify the impact of each factor on ocean biology (Bijma *et al.* 2013, p. 495). The response to both changes together is often larger than the response to those changes taken separately (IGBP *et al.* 2013, p. 18; Hale *et al.* 2011, entire). These synergistic interactions are likely to impact bivalves and other mollusks, the red knot's primary prey in most nonbreeding areas.

While direct predation by peregrine falcons may account for only minor losses of individual birds, observations by shorebird biologists in Virginia, Delaware, and New Jersey have found that the presence of peregrine falcons significantly affects red knot foraging patterns, causing birds to abandon or avoid beaches that otherwise would be used for foraging. During times of limited food availability, this disturbance could reduce the proportion of red knots that can attain sufficient weight for successful migration and breeding in the Arctic. As with predation, human disturbance can also have a synergistic effect with reduced food availability. The combined effects of these two threats (food availability and disturbance) at one key wintering site (Río Grande, Argentina, in Tierra del Fuego) caused the red knot's energy intake rate to drop from the highest known for *C. canutus* anywhere in the world in 2000, to among the lowest in 2008 (Escudero *et al.* 2012, pp. 359–362). Especially when food resources are limited, human disturbance can also exacerbate competition in Delaware Bay by giving a competitive advantage to gull species, which return to foraging more quickly than shorebirds do, following a flight response to vehicles, people, or dogs (Burger *et al.* 2007, p. 1164). Shorebirds can tolerate more disturbance before their fitness levels are reduced when feeding conditions are favorable (e.g., abundant prey, mild weather) (Niles *et al.* 2008, p. 105; Goss-Custard *et al.* 2006, p. 88).

In Delaware Bay, the potential exists for an unlikely but, if it occurred, high-impact synergistic effect among disease, environmental contaminants, and climate change. Because Delaware Bay is a known hotspot for low pathogenicity avian influenza (LPAI) among shorebirds, this region may act as a place where novel avian viruses (potentially including high pathogenicity (HP) forms) can amplify and subsequently spread in North America (Brown *et al.* 2013, p. 2). The Delaware River and Bay are also contaminated with PCBs (Suk and Fikslin 2006, p. 5), which are known to suppress the immune systems in waterbirds, such as herring gulls and black-crowned night herons (*Nycticorax nycticorax*) (Grasman *et al*. 2013, pp. 548, 559). If resident Delaware Bay birds are immunosuppressed by PCB tissue concentrations (which is unknown but possible), the potential exists for resident bird species such as mallards (*Anas platyrhynchos*) (Fereidouni *et al.* 2009, pp. 1, 6) or herring gulls (Brown *et al.* 2008, p. 394) to more easily acquire a virulent HPAI, which could then be transmitted to red knots during the spring stopover. Health impacts and mortality from HPAI have been shown in *Calidris canutus islandica* (Reperant *et al.* 2011, entire) and can be presumed in the *rufa* subspecies. Such an occurrence would be likely to exact high mortality on red knots.

In mallards, Fereidouni *et al.* (2009, pp. 1, 6) found that prior exposure to LPAI conferred some immunity to HPAI and could, therefore, increase the risk of mallards transmitting virulent forms of the disease (i.e., they tend to survive the HPAI and, therefore, can spread it). Olsen *et al.* (2006, p. 388) suggested that many wild bird species may be partially immune to HPAI due to previous exposure to LPAI, enhancing their potential to carry HPAI to previously unaffected areas. The applicability of this finding to shorebirds is unknown, but these results suggest that species with high rates of LPAI (e.g., ruddy turnstone, mallards (Brown *et al.* 2013, p. 2)) could be at higher risk of transmitting HPAI, while red knots (with low rates of LPAI) could be more likely to die from HPAI, if exposed. Further, modeling has suggested that, if climate change leads to mismatches between the phenology of ruddy turnstones (the main LPAI carriers) and horseshoe crab spawning, the prevalence of LPAI in turnstones would be projected to increase even as their population size decreased (Brown and Rohani 2012, p. 1). Although the risk of a PCB-mediated HPAI outbreak in Delaware Bay is currently unquantifiable, the findings of Brown and Rohani (2012, p. 1) suggest that this risk could be increased by climate change (e.g., by further increasing LPAI infection rates among ruddy turnstones and thereby enhancing their potential to survive and subsequently spread HPAI, should it occur).

In the Arctic, synergistic interactions are expected to occur among shifting vegetation communities, loss of sea ice, changing relationships between red knots and their predators and competitors, and the timing of snow melt and insect emergence. Such changes are superimposed on the red knot’s breeding season that naturally has very tight tolerances in time and energy budgets due to the harsh tundra conditions and the knot’s exceptionally long migration. High uncertainty exists about when and how such synergistic effects may affect red knot survival or reproduction. However, certainty is high that ecosystem changes are already underway and will continue, and the extent of ecosystem change is profound (Fraser *et al.* 2013, entire; Gauthier *et al.* 2013, p. 10; Olofsson *et al.* 2013, entire; Schmidt *et al.* 2012, p. 4421; NRC 2011, pp. 1, 44; Meltofte *et al.* 2007, p. 35; Ims and Fuglei 2005, entire; Piersma and Lindström 2004, entire; Rehfisch and Crick 2003, entire; Piersma and Baker 2000, entire; Zöckler and Lysenko 2000, entire; Lindström and Agrell 1999, entire) (see Factors A and C). For example, as conditions warm, vegetative conditions in the current red knot breeding range are likely to become increasingly dominated by trees and shrubs over the next century. It is unknown if red knots will respond to vegetative and other ecosystem changes by shifting their breeding range north, where they could face greater energetic demands of a longer migration, competition with *Calidris canutus islandica*, and possibly no reduction in predation pressure if predator densities also shift north as temperatures warm. Alternatively, red knots may attempt to adapt to changing conditions within their current breeding range, where they could face unfavorable vegetative conditions and a new suite of predators and competitors expanding northward.

We conclude that a number of threats are likely contributing to habitat loss, anthropogenic mortality, or both, and thus contribute to the red knot's threatened status, particularly considering the cumulative and synergistic effects of these threats, and that several key populations of this species have already undergone considerable declines.

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APPENDIX I. ACRONYMS

ACIA = Arctic Climate Impact Assessment
AEE = Puerto Rico Electric Energy Authority
AI = avian influenza
AOU = American Ornithologists Union
ARM = Adaptive Resource Management
ASMFC = Atlantic States Marine Fisheries Service
ASP = Amnesic Shellfish Poisoning
BOEM = Bureau of Ocean Energy Management (formerly part of BOEMRE) (U.S.)
BOEMRE = Bureau of Ocean Energy Management, Regulation, and Enforcement (U.S.) (formerly MMS)
BTO = British Trust for Ornithology
BSEE = Bureau of Safety and Environmental Enforcement (formerly part of BOEMRE) (U.S.)
CAFF = Conservation of Arctic Flora and Fauna
CBBEP = Coastal Bend Bays and Estuaries Program
CCSP = U.S. Climate Change Science Program
CFR = Code of Federal Regulations (U.S.)
COSEWIC = Committee on the Status of Endangered Wildlife in Canada
CSRPN = French Guiana Regional Scientific Council for Natural Heritage
CVI = Coastal Vulnerability Index
CWFNJ = Conserve Wildlife Foundation of New Jersey
CWS = Canadian Wildlife Service
CZMA = Coastal Zone Management Act (U.S.)
DDNREC = Delaware Department of Natural Resources and Environmental Control
DDT = dichloro-diphenyl-trichloroethane
DOE = Department of Energy (U.S.)
DOI = Department of the Interior (U.S.)
FAO = United Nations Food and Agriculture Organization
FDA = Food and Drug Administration (U.S.)
FFWCC = Florida Fish and Wildlife Conservation Commission
FIFRA = Federal Insecticide, Fungicide, and Rodenticide Act (U.S.)
FR = Federal Register (U.S.)
GDNR = Georgia Department of Natural Resources
GHG = greenhouse gas
GIS = Geographic Information System
HAB = harmful algal bloom
HCP = Habitat Conservation Plan (under the Endangered Species Act)
HPAI = high pathogenicity avian influenza
IGBP = International Geosphere-Biosphere Programme
IMO = International Maritime Organization
ISS = International Shorebird Survey
INRMP = Integrated Natural Resource Management Plans (U.S.)
IPCC = International Panel on Climate Change
IWSG = International Wader Study Group
LAL = Limulus Amebocyte Lysate
LDFW = Louisiana Department of Wildlife and Fisheries
LDNR = Louisiana Department of Natural Resources
LPAI = low pathogenicity avian influenza
MARPOL = International Convention for the Prevention of Pollution from Ships
MBCA = Canadian Migratory Birds Convention Act
MBTA Migratory Bird Treaty Act (U.S.)
MDEP = Maine Department of Environmental Protection
MDIFW = Maine Department of Inland Fisheries and Wildlife
MFWP = Montana Fish, Wildlife, and Parks
MMS = Mineral Management Service (now BOEM and BSEE) (U.S.)

MOU = Memorandum of Understanding
NAS = National Audubon Society
NASA = National Aeronautics and Space Administration (U.S.)
NCWRC = North Carolina Wildlife Resources Commission
NDGFD = North Dakota North Dakota Game and Fish Department
NDVI = Normalized Difference Vegetation Index
NECIS = National Environmental Coalition on Invasive Species
NEPA = National Environmental Policy Act (U.S.)
NISA = National Invasive Species Act (U.S.)
NJAS = New Jersey Audubon Society
NJDEP = New Jersey Department of Environmental Protection
NMFS = National Marine Fisheries Service (U.S.)
NOAA = National Oceanic and Atmospheric Administration (U.S.)
NPS = National Park Service (U.S.)
NPSOA = National Park Service Organic Act (U.S.)
NRC = National Research Council (U.S.)
NSP = Neurotoxic Shellfish Poisoning
NWF = National Wildlife Federation
NWR = National Wildlife Refuge (U.S.)
NWRSIA = National Wildlife Refuge System Improvement Act (U.S.)
OCS = Outer Continental Shelf
ODWC = Oklahoma Department of Wildlife Conservation
OPA = Oil Pollution Act (U.S.)
ORV = off-road vehicle
PCB = polychlorinated biphenyl
P.L. = Public Law
ppt = parts per thousand
PSP = Paralytic Shellfish Poisoning
ROM = Royal Ontario Museum
SARA = Canadian Species at Risk Act
SCDNR = South Carolina Department of Natural Resources
SLAMM = Sea Level Affecting Marsh Modeling
SUNY = State University of New York
UNCLOS = United Nations Conference on the Law of the Sea
USACE = U.S. Army Corps of Engineers
USCG = U.S. Coast Guard
USDA = U.S. Department of Agriculture
USEPA = U.S. Environmental Protection Agency
USFWS = U.S. Fish and Wildlife Service
USGCRP = U.S. Global Change Research Program
USGS = U.S. Geologic Survey
VMRC = Virginia Marine Resources Commission
WDFW = Washington Department of Fish and Wildlife
WEA = Wind Energy Area (U.S.)
WGFD = Wyoming Game and Fish Department
WHO = World Health Organization
WHSRN = Western Hemisphere Shorebird Reserve Network
WMO = World Meteorological Organization
WRDA = Water Resources Development Acts (U.S.)
WWF = World Wildlife Fund

APPENDIX II. GLOSSARY

Adaptive Resource Management (ARM) = the management framework adopted by ASMFC in 2012 under which horseshoe crab bait harvest levels are tied to red knot populations via scientific modeling

adventitious molt = a bird's replacement of feathers outside of the normal molt season(s) due to accidental loss, e.g., from trauma, parasites

aeolian = relating to wind

anthropogenic = caused by humans

Arctic (or polar) amplification = the observation that areas near the poles are warming faster than the global average rate

asynchrony = a mismatch between the timing of a species' annual breeding, migration, or wintering cycles and the windows of food availability or favorable weather conditions on which it depends

athalassic = inland, in regard to saline lakes

backpassing = an engineering practice that reverses the natural migration of sediment by mechanically (via trucks) or hydraulically (via pipes) transporting sand from accreting, downdrift areas of the beach to eroding, updrift areas of the beach

beach nourishment (or beach fill) = an engineering practice of deliberately adding sand (or gravel or cobbles) to an eroding beach, or the construction of a beach where only a small beach, or no beach, previously existed

beach raking (or beach grooming or beach cleaning) = a common practice to remove wrack, trash, and other natural material or manmade debris by raking or sieving the sand, often with motorized equipment

benthic = bottom-dwelling, in regard to aquatic organisms

biomes = the world's major biological communities (e.g., aquatic, desert, grassland, forest, tundra), classified according to the predominant vegetation, characterized by adaptations of organisms to that particular environment, and maintained under the climatic conditions of the region

brackish = in a coastal water body, having 0.5 to 30 ppt of dissolved salts, typical of estuaries

buildout = the condition that exists when all available land is either developed or preserved and no further development is possible

bulkhead = a shore-parallel coastal stabilization structure, often a vertical or stepped metal or timber wall, intended to protect the shoreline from erosion by holding soil and preventing it from sliding seaward

casual (or irregular) = describes bird species recorded less than annually in a particular geographic area

climate = the mean and variability of different types of weather conditions over time, with 30 years being a typical period for such measurements, although shorter or longer periods also may be used

climate change = a change in the mean or variability of one or more measures of climate (e.g., temperature or precipitation) that persists for an extended period, typically decades or longer, whether the change is due to natural variability, human activity, or both

clutch = the group of eggs in a single nest

cumulative = an increase in quantity, degree, or force by successive addition

cyrosphere = those portions of Earth's surface where water is in solid form

desiccation = drying out

ectoparasite = a parasite that lives on the outer surface of the body

El Niño = an occasional abnormal warming of tropical waters in the eastern Pacific from unknown causes

endogenous = caused by factors originating inside the body energy budget = in animals, the balance between energy income (i.e., feeding) and energy usage (e.g., body heat, movement)

ephemeral = lasting a short time

epizootic = a disease epidemic simultaneously affecting many animals

estuary = a partially enclosed tidal area where fresh water and salt water mix

eutrophication = accumulation of nutrients in a water body, often due to human activities, that typically results in dense growth of algae and other organisms, the decay of which depletes the water of oxygen

fecundity = reproductive capacity

Federal consistency = a requirement under the CZMA that, in States with approved Coastal Zone Management Plans, Federal action agencies ensure the activities they fund or authorize are consistent, to the maximum extent practicable, with the enforceable policies of that State's federally approved coastal management program

fitness = the ability to both survive and reproduce

French West Indies (or French Antilles) = territories under French sovereignty in the Antilles islands of the Caribbean, specifically Guadeloupe and its dependencies, Martinique, Saint Martin, and Saint Barthélemy

geolocator = a small, light-sensitive device attached to a bird's leg that records data used to estimate the bird's movements

gizzard = a muscular organ in a bird used for grinding food

global sea level = the average height of all the Earth's oceans

greenhouse gas (GHG) = any gas that absorbs infrared radiation in the atmosphere; examples include carbon dioxide and methane

groin = a shore-perpendicular coastal stabilization structure built across a beach and out into the water to prevent erosion along a finite section of beach by reducing the alongshore transport of sediment; often in multiples comprising a "groin field"

the Guianas = the region of northern South America comprised of Suriname, Guyana, and French Guiana

harmful algal bloom (HAB) = the proliferation of a toxic or nuisance algal species (which can be microscopic or macroscopic, such as seaweed) that negatively affects natural resources or humans

hemiarctic = middle Arctic

High Arctic = the highest latitude portions of the Arctic, characterized by drier tundra on mineral soils

high pathogenicity avian influenza (HPAI) = specific ability of an influenza virus to cause disease in experimentally inoculated chickens (i.e., its virulence); does not reflect the capacity of the virus to produce disease in other species

hop migration = in birds, characterized by shorter flights and more frequent stopovers at sites a short distance apart

hummocky = characterized by knolls or mounds

hypersaline (or brine) = in an inland water body, having greater than 50 ppt dissolved salts

hyposaline = in an inland water body, having 3 to 20 ppt dissolved salts

immunocompetence = ability of the body to produce a normal immune response following exposure to disease

impoundment = a body of water confined within an enclosure (e.g., a reservoir)

jetty = a shore-perpendicular coastal stabilization structure built at a tidal inlet to stabilize the location of the inlet, usually for navigation. Jetties may be designed to keep sand out of a navigation channel or provide calm-water access to harbor facilities.

jump migration = in birds, characterized by longer flights (often thousands of miles) and less frequent stopovers at sites a long distance apart, requiring the accumulation of large stores of fuel (e.g., fat) at each stop

La Niña = an occasional abnormal cooling of tropical waters in the eastern Pacific from unknown causes

Lesser Antilles = a long arc of small islands forming the eastern boundary of the Caribbean Sea before curving west along the northern coast of South America

Limulus Amebocyte Lysate (LAL) = a clotting agent extracted from horseshoe crab blood that makes it possible to detect human pathogens in patients, drugs, and intravenous devices

littoral = pertaining to the coast

Low Arctic = lower latitude portions of the Arctic, characterized by moister tundra with moss carpets and peaty soils

macrofauna = animals large enough to be seen by the naked eye

macroinvertebrate = an invertebrate (animal without a backbone) large enough to be seen by the naked eye

macrophyte = a plant, especially a marine plant, large enough to be visible to the naked eye (e.g., seaweed)

macrotidal = having a tidal range greater than 13 ft (4 m)

mesosaline = in an inland water body, having 20 to 50 ppt dissolved salts

microtidal = having a tidal range less than 6.6 ft (2 m)

mollusk = an animal belonging to the invertebrate phylum Mollusca, characterized by a fleshy mantle and often a shell (e.g., clams, mussels, snails)

mudboil = bare patches that form on silt or clay soils

mussel spat = small, juvenile stages of mussel species

nor'easter = a common type of extra-tropical (i.e., non-tropical) storm, characterized by continuously strong northeasterly winds blowing from the ocean

Normalized Difference Vegetation Index (NDVI) = a measure of vegetation photosynthetic capacity

ocean acidification = a decrease in ocean pH over decades or longer that is caused primarily by uptake of carbon dioxide from the atmosphere

Outer Continental Shelf (OCS) = all submerged lands, subsoil, and seabed that belong to the United States and are lying seaward of and outside of the States' jurisdiction

overwash = a form of coastal flooding, typically on barrier islands, that can move sediment landward. Specifically, overwash is the flow of water and sediment over a beach crest that does not directly return to the ocean where it originated. Overwash begins when the run-up level of waves, usually coinciding with a storm surge, exceeds the local beach or dune crest height. Sediment transported by overwash can be deposited onto the upper beach or as far as the back bay, estuary, or lagoon. The process of overwash is an important mechanism by which some types of barriers migrate landward and upward over time, and overwash may be the means by which barrier coastlines are preserved through a natural process under the action of storms and relative sea level rise.

panne = a shallow, high salinity, mud-bottomed depression on a marsh surface

pathogen = an organism that causes disease

pectoral = relating to the chest

pH = percent hydrogen, a measure of acidity or alkalinity

phenology = the timing of periodic life-cycle events

physiognomic class = a level in vegetation classification hierarchies defined by the relative percent canopy cover of the tree, shrub, dwarf shrub, herb, and non-vascular life-forms in the uppermost strata during the peak of growing season

polar desert = an extremely cold, dry biome typically occurring at higher latitudes than tundra biomes within the High Arctic zone. Vegetation covers 5 percent or less of the ground surface, is less than 3.9 in (10 cm) high, and is dominated by herbs, lichens, and mosses.

precocial = referring to young animals, born or hatched relatively mature, mobile and able to feed themselves almost immediately prey quality = the flesh-to-shell ratio of hard shelled prey taken by red knots. Higher quality prey have more flesh, thinner shells, or both.

quadratic = a mathematical relationship between one variable and the square of another variable

radio telemetry (or radio tracking) = a wildlife research method that involves monitoring the radio signals sent from an animal-attached device to track the animal's movements. Telemetry refers to the process of transmitting the information through the atmosphere.

rare = describes bird species with near-annual occurrence in a particular geographic area, but usually single individuals to very small flocks of 2 to 5 birds

relative sea level = sea level related to the level of the continental crust. Changes in relative sea level reflect changes of the global sea level, movements of the continental crust (e.g., natural uplift of the land, or natural or human-caused subsidence (settling or "sinking") of the land), and regional differences in ocean circulation.

resighting = observation of a marked animal. By tracking where marked red knots are observed across seasons and years, researchers can draw inferences about wintering and migration areas, migration routes and timing, life history, regional population sizes, and survival rates.

restinga = an intertidal shelf of densely-packed dirt blown by strong, offshore winds

reverse migration = movement of birds in the opposite direction of the prevailing migratory movement for that season

revetment = a shore-parallel coastal stabilization structure, often a sloped rock or earthen wall, intended to protect the shoreline from erosion by providing a cover or facing of erosion-resistant material

sand scraping (or beach bulldozing) = the process of mechanically redistributing beach sand from the littoral zone (along the edge of the sea) to the upper beach to increase the size of the primary dune or to provide a source of sediment for beaches that have no existing dune; no new sediment is added to the system

seawall = a shore-parallel coastal stabilization structure, often a vertical rock or concrete wall, intended to prevent or alleviate flooding of developed areas by blocking storm surges and waves

sporozoan = an organism that reproduces by spores

stable isotope analysis = chemical analysis that produces a stable isotope "signature" (e.g., from feathers), which reflects the geographic region in which the birds were feeding when those feathers were grown. Isotopes are variants of a particular chemical element—while all isotopes of a given element have the same number of protons in each atom, they differ in neutron number.

staging area = those stopover sites with abundant, predictable food resources where birds prepare for an energetic challenge (usually a long flight over a barrier such as an ocean or a desert) requiring substantial fuel stores and physiological changes without which significant fitness costs are incurred

stopover area = areas where birds rest and feed during migration

sub-arctic = the latitudinal zone immediately south of the Arctic

subsaline = in an inland water body, having 1 to 3 ppt dissolved salts

subspecies composition = in regard to *Calidris canutus* present in a particular nonbreeding area, the proportion different subspecies (e.g., all *rufa*, all *roselaari*, or both subspecies occur in varying proportions at the same or different times of year)

substrate = the surface on (or medium in) which a plant or animal lives

superabundance = refers to the large volume of horseshoe crab eggs needed to support red knots and other shorebirds at the Delaware Bay spring staging area. Female crabs deposit their eggs in the sand at a depth that is beyond the reach of red knots and most other shorebirds. Wave action and burrowing by subsequent spawning horseshoe crabs move eggs toward the surface. Thus, a high density of spawning horseshoe crabs is needed for the eggs to become available to shorebirds. Further, a very large number of surface-available eggs is needed to support this staging phenomenon because of the small size of the eggs and the large number of shorebirds. Although a single horseshoe crab egg contains a very small amount of energy, eggs are present in such large numbers (superabundance) in Delaware Bay that birds can eat enough in 2 weeks to nearly double their weights.

synergy = the interaction of elements that, when combined, produce a total effect that is greater than the sum of the individual elements

terminal groin = a groin on the updrift side of an inlet usually extending far enough seaward to block all littoral transport of sediment, in order to prevent sediment from entering the inlet (e.g., to retain updrift beach width, prevent inlet sedimentation, or both); typically shorter than a jetty

threshold (or tipping point) = regarding a component or phenomenon within an Earth system, a point beyond which an abrupt or nonlinear transition to a different state occurs. Specific to ecological systems, an ecological threshold is the point at which there is an abrupt change in an ecosystem quality, property, or phenomenon, or where small changes in one or more external conditions produce large and persistent responses in an ecosystem.

trophic = relating to food or feeding

tropical storm = a rotating storm that forms over warm tropical oceans characterized by a low-pressure center, strong winds, and a spiral arrangement of thunderstorms that produce heavy rain; may be called hurricane, typhoon, cyclone, tropical storm, or tropical depression, depending on its location and strength.

tundra = a cold, dry, treeless biome occurring between the polar desert and ice cap to the north and the treeline to the south, having a permanently frozen subsoil and supporting low-growing vegetation such as lichens, mosses, and stunted shrubs

uncommon = describes bird species with annual occurrence in a particular geographic area, but not always to be encountered daily and from 1 to 10 birds, seldom more

vagrant (or accidental) = a bird that has strayed or been blown far from its usual range or migratory route; implies an extraordinary observational record, out of the normal pattern, and unlikely to occur again

virulent = regarding pathogens, highly harmful or infective

wrack = seaweed and other organic debris are deposited by the tides