DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

50 CFR Part 17

[Docket No. FWS–R5–ES–2013–0097; 4500030113]

RIN 1018–AY17

Endangered and Threatened Wildlife and Plants; Proposed Threatened Status for the Rufa Red Knot (*Calidris canutus rufa*)

AGENCY: Fish and Wildlife Service, Interior.

ACTION: Proposed rule.

SUMMARY: We, the U.S. Fish and Wildlife Service, propose to list the rufa red knot (*Calidris canutus rufa*) as a threatened species under the Endangered Species Act of 1973, as amended (Act). If we finalize this rule as proposed, it would extend the Act's protections to this species. The effect of this regulation will be to add this species to the List of Endangered and Threatened Wildlife.

DATES: We will accept all comments received or postmarked on or before November 29, 2013. Comments submitted electronically using the Federal eRulemaking Portal (see **ADDRESSES** section, below) must be received by 11:59 p.m. Eastern Time on the closing date. We must receive requests for public hearings, in writing, at the address shown in the **FOR FURTHER INFORMATION CONTACT** section by November 14, 2013.

ADDRESSES: *Document availability:* You may obtain copies of the proposed rule and its four supplemental documents on the Internet at *http://www.regulations.gov* at Docket Number FWS–R5–ES–2013–0097, or by mail from the New Jersey Field Office (see **FOR FURTHER INFORMATION CONTACT**).

Comment submission: You may submit written comments by one of the following methods:

(1) *Electronically:* Go to the Federal eRulemaking Portal: *http://www.regulations.gov*. In the Search box, enter FWS–R5–ES–2013–0097, which is the docket number for this rulemaking. You may submit a comment by clicking on ''Comment Now!''

(2) *By hard copy:* Submit by U.S. mail or hand-delivery to: Public Comments Processing, Attn: FWS–R5–ES–2013–0097; Division of Policy and Directives Management; U.S. Fish and Wildlife Service; 4401 N. Fairfax Drive, MS 2042–PDM; Arlington, Virginia 22203.

We request that you send comments only by the methods described above. We will post all information received on *http://www.regulations.gov*. This generally means that we will post any personal information you provide us (see the *Public Comments* section below for more details).

FOR FURTHER INFORMATION CONTACT: Eric Schrading, Acting Field Supervisor, U.S. Fish and Wildlife Service, New Jersey Field Office, 927 North Main Street, Building D, Pleasantville, New Jersey 08232, by telephone 609–383–3938 or by facsimile 609–646–0352. Persons who use a telecommunications device for the deaf (TDD) may call the Federal Information Relay Service (FIRS) at 800–877–8339.

SUPPLEMENTARY INFORMATION:

Executive Summary

Why we need to publish a rule. Under the Act, if a species is determined to be endangered or threatened throughout all or a significant portion of its range, we are required to promptly publish a proposal in the **Federal Register** and make a determination on our proposal within 1 year. Critical habitat shall be designated, to the maximum extent prudent and determinable, for any species determined to be an endangered or threatened species under the Act. Listing a species as an endangered or threatened species and designations and revisions of critical habitat can be completed only by issuing a rule.

This rule proposes listing the rufa red knot (*Calidris canutus rufa*) as a threatened species. The rufa red knot is a candidate species for which we have on file sufficient information on biological vulnerability and threats to support preparation of a listing proposal, but for which development of a listing regulation has been precluded by other higher priority listing activities. This rule reassesses all available information regarding status of and threats to the rufa red knot. We will also publish a proposal to designate critical habitat for the rufa red knot under the Act in the near future.

The basis for our action. Under the Act, we may determine that a species is an endangered or threatened species based on any of 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 determined that the rufa red knot is threatened due to loss of both breeding and nonbreeding habitat; potential for disruption of natural predator cycles on the breeding grounds; reduced prey availability throughout the nonbreeding range; and increasing frequency and severity of asynchronies (''mismatches'') in the timing of the birds' annual migratory cycle relative to favorable food and weather conditions.

We will seek peer review. We will seek comments from independent specialists to ensure that our designation is based on scientifically sound data, assumptions, and analyses. We will invite these peer reviewers to comment on our listing proposal. Because we will consider all comments and information received during the comment period, our final determinations may differ from this proposal.

Information Requested

Public Comments

We intend that any final action resulting from this proposed rule will be based on the best scientific and commercial data available and be as accurate and as effective as possible. Therefore, we request comments or information from the public, other concerned governmental agencies, Native American tribes, the scientific community, industry, or any other interested parties concerning this proposed rule. We particularly seek comments concerning:

(1) The rufa red knot's biology, range, and population trends, including:

(a) Biological or ecological requirements of the species, including habitat requirements for feeding, breeding, and sheltering;

(b) Genetics and taxonomy;

(c) Historical and current range including distribution patterns;

(d) Historical and current population levels and current and projected trends; and

(e) Past and ongoing conservation measures for the species, its habitat, or both.

(2) Factors that that may affect the continued existence of the species, which may include habitat modification or destruction, overutilization, disease, predation, the inadequacy of existing regulatory mechanisms, or other natural or manmade factors.

(3) Biological, commercial trade, or other relevant data concerning any threats (or lack thereof) to this species and regulations that may be addressing those threats.

(4) Additional information concerning the historical and current status, range, distribution, and population size of this species, including the locations of any additional populations of this species.

(5) Genetic, morphological, chemical, geolocator, telemetry, survey (e.g., resightings of marked birds), or other data that clarify the distribution of *Calidris canutus rufa* versus *C.c. roselaari* wintering and migration areas, including the subspecies compositions of those *C. canutus* that occur from southern Mexico to the Caribbean and Pacific coasts of South America.

(6) Information regarding intra- and inter-annual red knot movements within and between the Southeast United States-Caribbean and the Northwest Gulf of Mexico wintering regions, or other information that helps to clarify their geographic limits and degree of connectivity.

(7) Information that helps clarify the geographic extent of the rufa red knot's breeding range, and the extent to which rufa red knots from different wintering areas interbreed, as well as the geographic extent of the *Calidris canutus islandica* breeding range.

(8) Data regarding rates of rufa red knot reproductive success.

(9) Information regarding habitat loss or predation in rufa red knot breeding areas.

(10) Information regarding important rufa red knot stopover areas, including inland areas (such as the Mississippi Valley, Great Lakes, and Great Plains). We particularly seek information on the frequency, timing, and duration of use; numbers of birds; habitat and prey characteristics; foraging and roosting habits; and any threats associated with such areas.

(11) Data that support or refute the concept that juvenile rufa red knots at least partially segregate from adults during the nonbreeding seasons. We particularly seek information on juvenile wintering and migration locations; frequency, timing, and duration of juvenile use; numbers of juveniles and adults in these areas; juvenile habitat and prey characteristics; juvenile foraging and roosting habits; juvenile survival rates; and any threats associated with these areas.

(12) Data that clarify the degree of rufa red knot site fidelity to breeding locations, wintering regions, or migration stopover sites.

(13) Data regarding the percentage of rufa red knots that do not use Delaware Bay as a spring stopover site.

(14) Data regarding rufa red knot use of the Caribbean. We particularly seek information on the frequency, timing, and duration of use; numbers of birds; habitat and prey characteristics; foraging and roosting habits; and any threats associated with areas of red knot use in the Caribbean.

(15) Data regarding red knot use of wrack material as a microhabitat for foraging or roosting.

(16) Information regarding the frequency and severity of the threats to red knots (e.g., documented mortality levels from disease, harmful algal blooms, contaminants, oil spills, wind turbines), their habitats (e.g., effects of sea level rise, development, aquaculture), or their food resources (e.g., harvest of marine resources, climate change) outside the United States.

(17) Information regarding legal and illegal harvest (i.e., hunting or poaching) rates and trends in nonbreeding areas and the effects of harvest on the red knot.

(18) Information regarding non-U.S. laws, regulations, or policies relevant to the regulation of red knot hunting; classification of the red knot as a protected species; protection of red knot habitats; or threats to the red knot (e.g., to address the data gaps identified under Summary of Factors Affecting the Species).

Please include sufficient information with your submission (such as scientific journal articles or other publications) to allow us to verify any scientific or commercial information you include.

Please note that submissions merely stating support for or opposition to the action under consideration without providing supporting information, although noted, will not be considered in making a determination, as section 4(b)(1)(A) of the Act directs that determinations as to whether any species is an endangered or threatened species must be made ''solely on the basis of the best scientific and commercial data available.''

You may submit your comments and materials concerning this proposed rule by one of the methods listed in the **ADDRESSES** section. We request that you send comments only by the methods described in the **ADDRESSES** section.

If you submit information via *http://www.regulations.gov,* your entire submission—including any personal identifying information—will be posted on the Web site. If your submission is made via a hardcopy that includes personal identifying information, you may request at the top of your document that we withhold this information from public review. However, we cannot guarantee that we will be able to do so. We will post all hardcopy submissions on *http://www.regulations.gov*. Please include sufficient information with your comments to allow us to verify any scientific or commercial information you include.

Comments and materials we receive, as well as supporting documentation we used in preparing this proposed rule, will be available for public inspection on *http://www.regulations.gov,* or by appointment, during normal business hours, at the U.S. Fish and Wildlife Service, New Jersey Field Office (*http://www.fws.gov/northeast/njfieldoffice/*) (see **FOR FURTHER INFORMATION CONTACT**).

Public Hearings

Section 4(b)(5) of the Act provides for one or more public hearings on this proposal, if requested. Requests must be received within 45 days after the date of publication of this proposed rule in the **Federal Register**. Such requests must be sent to the address shown in the **FOR FURTHER INFORMATION CONTACT** section. We will schedule public hearings on this proposal, if any are requested, and announce the dates, times, and places of those hearings, as well as how to obtain reasonable accommodations, in the **Federal Register** and local newspapers at least 15 days before the hearing.

Persons needing reasonable accommodations to attend and participate in a public hearing should contact the New Jersey Field Office at 609–383–3938, as soon as possible. To allow sufficient time to process requests, please call no later than 1 week before any scheduled hearing date. Information regarding this proposed rule is available in alternative formats upon request.

Peer Review

In accordance with our joint policy on peer review published in the **Federal Register** on July 1, 1994 (59 FR 34270), we have sought the expert opinions of three appropriate and independent specialists regarding this proposed rule. The purpose of peer review is to ensure that our listing determination and critical habitat designation are based on scientifically sound data, assumptions, and analyses. The peer reviewers have expertise in the red knot's biology, habitat, or threats, which will inform our determination. We invite comment from the peer reviewers during this public comment period.

Previous Federal Action

Comprehensive information regarding previous federal actions relevant to the proposed listing of the rufa red knot is available as a supplemental document (''Previous Federal Actions'') on the Internet at *http://www.regulations.gov* (Docket No. FWS–R5–ES–2013–0097; see **ADDRESSES** section for further access instructions).

Background

Species Information

Comprehensive information regarding the rufa red knot's taxonomy, distribution, life history, habitat, and diet, as well as its historical and current abundance, is available as a supplemental document (''Rufa Red Knot Ecology and Abundance'') on the Internet at *http://www.regulations.gov* (Docket No. FWS–R5–ES–2013–0097; see **ADDRESSES** section for further access instructions). A brief summary is provided here.

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.) 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. During both the northbound (spring) and southbound (fall) migrations, red knots use key staging and stopover areas to rest and feed.

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).

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 (Committee on the Status of Endangered Wildlife in Canada (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 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 (see supplemental document—Rufa Red Knot Ecology and Abundance—figures 1 and 2).

Subspecies *Calidris canutus roselaari* breeds in western Alaska and on Wrangel Island, Russia (Carmona *et al.* in press; Buehler and Baker 2005, p. 498). Wintering areas for *C.c. roselaari* are poorly known (Harrington 2001, p. 5). In the past, *C. 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 new morphological evidence, resightings of marked birds, and results from geolocators (light-sensitive tracking devices), *C.c. roselaari* is now thought to be largely or wholly confined to the Pacific coast of the Americas during migration and in winter (Carmona *et al.* in press; Buchanan *et al.* 2011, p. 97; USFWS 2011a, pp. 305–306; Buchanan *et al.* 2010, p. 41; Soto-Montoya *et al.* 2009, p. 191; Niles *et al.* 2008, pp. 131–133; Tomkovich and Dondua 2008, p. 102). Although *C.c. roselaari* is generally considered to occur on the Pacific coast, a few *C. canutus* movements have recently been documented between Texas and the Pacific coast during spring migration (Carmona *et al.* in press). Despite a number of population-wide morphological differences (U.S. Fish and Wildlife Service (USFWS) 2011a, p. 305), the *rufa* and *roselaari* subspecies cannot be distinguished in the field (D. Newstead pers. comm. September 14, 2012). The subspecies composition of Pacific-wintering *C. canutus* from central Mexico to Chile is unknown.

Pursuant to the definitions in section 3 of the Act, ''the term species includes any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature.'' Based on the information in the supplemental document Rufa Red Knot Ecology and Abundance, the Service accepts the characterization of *Calidris canutus rufa* as a subspecies because each recognized subspecies is believed to occupy separate breeding areas, in addition to having morphological and behavioral character differences. Therefore, we find that *C.c. rufa* is a valid taxon that qualifies as a listable entity under the Act.

Breeding

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). Age of first breeding is uncertain but for most birds is probably at least 2 years (Harrington 2001, p. 21). Red knots generally nest in dry, slightly elevated tundra locations, often on windswept slopes with little vegetation. Breeding territories are located inland, but near arctic coasts, and foraging areas are located near nest sites in freshwater wetlands (Niles *et al.* 2008, p. 27; Harrington 2001, p. 8). On the breeding grounds, the red knot's diet consists mostly of terrestrial invertebrates such as insects (Harrington 2001, p. 11). Breeding occurs in June (Niles *et al.* 2008, pp. 25–26). 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: weather that affects nesting conditions and food availability (see Summary of Factors Affecting the Species—Factor E—Asynchronies) and the abundance of arctic lemmings (*Dicrostonyx torquatus* and *Lemmus sibericus*) that affects predation rates (see Summary of Factors Affecting the Species—Factor C—Predation—Breeding).

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 areas for the 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 *et al.* in press; L. Patrick pers. comm. August 31, 2012; Niles *et al.* 2008, p 17) (see supplemental

document—Rufa Red Knot Ecology and Abundance—figure 2). Smaller numbers of knots winter in the Caribbean, and along the central Gulf coast (Alabama, Mississippi), the mid-Atlantic, and the Northeast United States. *Calidris canutus* is also known to winter in Central America and northwest South America, but it is not yet clear if all these birds are the *rufa* subspecies. Little information exists on where juvenile red knots spend the winter months (USFWS and Conserve Wildlife Foundation 2012, p. 1), and there may be at least partial segregation of juvenile and adult red knots on the wintering grounds.

Migration

Each year red knots make one of the longest distance migrations known in the animal kingdom, traveling up to 19,000 miles (mi) (30,000 kilometers (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, leg muscles, gizzard (a muscular organ used for grinding food), stomach, intestines, and liver all decrease in size, while 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, fuels a body transformation to breeding condition (Morrison 2006, pp. 610–612). Red knots from different wintering areas appear to employ different migration strategies, including differences in timing, routes, and stopover areas. However, full segregation of migration strategies, routes, or stopover areas does not occur among red knots from different wintering areas.

Major 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 Virginia barrier islands (United States); and Delaware Bay (Delaware and New Jersey, United States) (Cohen *et al.* 2009, p. 939; Niles *et al.* 2008, p. 19; González 2005, p. 14). Important 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 (Newstead *et al.* in press; 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, pp. 66; Morrison and Harrington 1992, p. 74; Spaans 1978, p. 72). (See supplemental document—Rufa Red Knot Ecology and Abundance—figure 3.) 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 Canada during migration (Niles *et al.* 2008, p. 29).

Texas knots follow an inland flyway to and from the breeding grounds, using spring and fall stopovers along western Hudson Bay in Canada and in the northern Great Plains (Newstead *et al.* in press; Skagen *et al.* 1999). Stopover records from the Northern Plains are mainly in Canada, but small numbers of migrants have been sighted throughout the U.S. Great Plains States (eBird.org 2012). Some red knots wintering in the Southeastern United States and the Caribbean migrate north along the U.S. Atlantic coast before flying overland to central Canada from the mid-Atlantic, while others migrate overland directly to the Arctic from the Southeastern U.S. coast (Niles *et al.* in press). These eastern red knots typically make a short stop at James Bay in Canada, but may also stop briefly along the Great Lakes, perhaps in response to weather conditions (Niles et al. 2008, pp. 20, 24; Morrison and Harrington 1992, p. 79). Red knots are restricted to the ocean coasts during winter, and occur primarily along the coasts during migration. However, small 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 nearly every interior State (eBird.org 2012).

Migration and Wintering Habitat

Long-distance migrant shorebirds are highly dependent on the continued existence of quality habitat at a few key staging areas. These areas serve as stepping stones between wintering and breeding areas. Conditions or factors influencing shorebird populations on staging areas control much of the remainder of the annual cycle and survival of the birds (Skagen 2006, p. 316; International Wader Study Group 2003, p. 10). At some stages of migration, very high proportions of entire populations may use a single migration staging site to prepare for long flights. Red knots show some fidelity to particular migration staging areas between years (Duerr *et al.* 2011, p. 16; Harrington 2001, pp. 8–9, 21).

Habitats used by red knots in migration and wintering areas are similar in character, generally coastal marine and estuarine (partially enclosed tidal area where fresh and salt water mixes) habitats with large areas of exposed intertidal sediments. 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.* 2010a, 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 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 (K. Kalasz pers. comm. November 26, 2012; L. Niles pers. comm. November 19, 2012). 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. 2, 4–5).

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 (*Limulus polyphemus*) eggs (Piersma and van Gils 2011, p. 9; Harrington 2001, pp. 9–11). Mollusk prey are swallowed whole and crushed in the gizzard (Piersma and van Gils 2011, pp. 9–11). 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)). 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).

The primary prey of the rufa red knot in non-breeding habitats include blue mussel (*Mytilus edulis*) spat (juveniles); *Donax* and *Darina* clams; snails (*Littorina* spp.), and other mollusks, with polycheate worms, insect larvae, and crustaceans also eaten in some locations. 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 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), which provide a superabundant source of easily digestible food.

Red knots and other shorebirds that are long-distance migrants must take advantage of seasonally abundant food resources at intermediate stopovers to build up fat reserves for the next non-stop, long-distance flight (Clark *et al.* 1993, p. 694). Although foraging red knots can be found widely distributed in small numbers within suitable habitats during the migration period, birds tend to concentrate in those areas where abundant food resources are consistently available from year to year.

Abundance

In the United States, red knot populations declined sharply in the late 1800s and early 1900s due to excessive sport and market hunting, followed by hunting restrictions and signs of population recovery by the mid-1900s (Urner and Storer 1949, pp. 178–183; Stone 1937, p. 465; Bent 1927, p. 132). However, it is unclear whether the red knot population fully recovered its historical numbers (Harrington 2001, p. 22) following the period of unregulated hunting.

More recently, long-term survey data from two key areas (Tierra del Fuego wintering area and Delaware Bay spring stopover site) both show a roughly 75 percent decline in red knot numbers since the 1980s (A. Dey pers. comm. October 12, 2012; G. Morrison pers. comm. August 31, 2012; Dey *et al.* 2011a, pp. 2–3; 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). Survey data for the Virginia barrier islands spring stopover area show no trend since 1995 (B. Watts pers. comm. November 15, 2012). Survey data are also available for the Brazil, Northwest Gulf of Mexico, and Southeast-Caribbean wintering areas, but are insufficient to infer trends.

Climate Change

Comprehensive background information regarding climate change is available as a supplemental document (''Climate Change Background'') on the Internet at *http://www.regulations.gov* (Docket No. FWS–R5–ES–2013–0097; see **ADDRESSES** section for further access instructions). As explained in the supplemental document, the International Panel on Climate Change (IPCC) uses standardized terms to define levels of confidence (from ''very high'' to ''very low'') and likelihood (from ''virtually certain'' to ''exceptionally unlikely''). When used in this context, these terms are given in quotes in this document.

Summary of Factors Affecting the Species

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 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 Michaels 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 (National Research Council (NRC) 2010, 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 (NRC 2010, 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 (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).

Several threats are related to the possibility of changing storm 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 (e.g., key stopover areas) can pose a threat (see Factor E and the ''Coastal Storms and Extreme Weather''

section of the Climate Change Background supplemental document). 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.* 2012, p. 1; Dey *et al.* 2011b, pp. 1–2; Nebel 2011, p. 217).

With arctic warming, vegetation conditions in the red knot's breeding grounds are expected to change, causing the zone of nesting habitat to shift and perhaps contract, but this process may take decades to unfold (Feng *et al.* 2012, p. 1366; Meltofte *et al.* 2007, p. 36; Kaplan *et al.* 2003, p. 10). Ecological shifts in the Arctic may appear sooner. High uncertainty exists about when and how changing interactions among vegetation, predators, competitors, prey, parasites, and pathogens may affect the red knot, but the impacts are potentially profound (Fraser *et al.* 2013; entire; Schmidt *et al.* 2012, p. 4421; Meltofte *et al.* 2007, p. 35; Ims and Fuglei 2005, entire).

In summary, 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. The 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 expected to change, causing the zone of nesting habitat to shift and perhaps contract. Arctic freshwater systems—foraging areas for red knots during the nesting season—are particularly sensitive to climate change.

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. Outside of the breeding grounds, red knots rely 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).

According to the National Research Council (NRC) (2010, p. 43), the rate of global sea level rise has increased from about 0.02 in (0.6 mm) per year in the late 19th century to approximately 0.07 in (1.8 mm) per year in the last half of the 20th century. The rate of increase has accelerated, and over the past 15 years has been in excess of 0.12 in (3 mm) per year. In 2007, the IPCC estimated that sea level would ''likely'' rise by an additional 0.6 to 1.9 feet (ft) (0.18 to 0.59 meters (m)) by 2100 (NRC 2010, p. 44). This projection was based largely on the observed rates of change in ice sheets and projected future thermal expansion of the oceans but did not include the possibility of changes in ice sheet dynamics (e.g., rates and patterns of ice sheet growth versus loss). Scientists are working to improve how ice dynamics can be resolved in climate models. Recent research suggests that sea levels could potentially rise another 2.5 to 6.5 ft (0.8 to 2 m) by 2100, which is several times larger than the 2007 IPCC estimates (NRC 2010, p. 44; Pfeffer *et al.* 2008, p. 1340). However, projected rates of sea level rise estimates remain rather uncertain, due mainly to limits in scientific understanding of glacier and ice sheet dynamics (NRC 2010, p. 44; Pfeffer *et al.* 2008, p. 1342).

The amount of sea level change varies regionally because of different rates of settling (subsidence) or uplift of the land, and because of differences in ocean circulation (NRC 2010, p. 43). In the last century, for example, 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 (U.S. Environmental Protection Agency (USEPA) 2013). 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 (National Oceanic and Atmospheric Administration (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 1 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.

TABLE 1—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

TABLE 1—LOCAL SEA LEVEL TRENDS FROM WITHIN THE RANGE OF THE RED KNOT—Continued

[NOAA 2012a]

Station	Mean local sea level trend (mm per year)	Data period
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

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 may increase as sea level rises (Florida Oceans and Coastal Council 2010, p. 6). However, even if such a correlation is borne out, predicting the effect of sea level rise on beaches is more complex. Even if wetland or upland coastal lands are lost, sandy or muddy intertidal habitats can often migrate or reform. However, forecasting how such changes may unfold is complex and uncertain. 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 that is beyond the capability of current models (CCSP 2009b, pp. 27–28; Frumhoff *et al.* 2007, p. 29; 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). 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 2.6 to 6.6 ft (0.8 to 2 m) projected by 2100 (NRC 2010, p. 44). 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) (Western Hemisphere Shorebird Reserve Network (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 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).

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 (NRC 2010, 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.* in press; 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 provides an estimate of the land area that may be vulnerable to the effects of sea level rise, but does not incorporate regional variation in tidal regimes (Strauss *et al.* 2012, p. 2), coastal processes (e.g., barrier island migration), or environmental changes that may occur as sea level rises (e.g., salt marsh deterioration) (CCSP 2009b, p. 44). Because the majority of the 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. 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).

By altering coastal geomorphology, sea level rise will cause significant and often dramatic changes to coastal landforms including barrier islands,

beaches, and intertidal flats (CCSP 2009b, p. 13; Rehfisch and Crick 2003, p. 89), primary red knot habitats. 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 (Tebaldi *et al.* 2012, pp. 7–8). Rising sea level not only increases the likelihood of coastal flooding, but also changes the template for waves and tides to sculpt the coast, which can lead to loss of land orders of magnitude greater than that from direct inundation alone (Ashton *et al.* 2007, p. 1). 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 uncertain, hampered by limited understanding of coastal responses and the innate complexity of the coastal zone (Ashton *et al.* 2007, p. 9). Coastal responses to climate change will not likely be homogeneous along the coast, due to local differences in geology and other factors (Ashton *et al.* 2007, p. 9).

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 future storm events (Ashton *et al.* 2009, p. 9). Most erosion events on the Atlantic and Gulf coasts are the result of storms. 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). The Atlantic and Gulf coast 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). As discussed in the supplemental document Climate Change Background, 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).

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 supplemental document—Climate Change Background). 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 that are dynamic and subject to seasonal erosion and accretion (the accumulation of sediment). Sea level rise and shoreline erosion have reduced 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, red knot 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.

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).

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 breaking up and drowning in place, rather than migrating (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 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 (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).

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 International Shorebird Survey (eBird.org 2012) 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 2). 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).

TABLE 2—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.

United States—Northeast and Mid-Atlantic

In the Northeast (Maine to New Jersey), the 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 the Northeast 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). Along the ocean shores of the mid-Atlantic (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

The rate of sea level rise in the Delaware Bay over the past century was about 0.12 in (3 mm) per year (table 1; 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).

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). 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) National Wildlife Refuges, 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).

At present, 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, 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 (Niles *et al.* 2012, p. 1). Several areas were eroded to exposed sod bank or rubble (used in shoreline stabilization), which do not provide suitable spawning habitat. Creek mouths may now constitute the bulk of the remaining intact spawning areas (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.

United States—Southeast and the Gulf Coast

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. The highest rates of erosion in Florida are generally localized around tidal inlets (Morton and Miller 2005, p. 1). 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 2). 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 National Wildlife Refuge (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, 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 2013; 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 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).

Compared to shoreline erosion in some other Gulf coast states, 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, a preferred red knot habitat (see the ''Migration and Wintering Habitat'' section of the Rufa Red Knot Ecology and Abundance supplemental document). 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).

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.

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 a mid-range estimate (3.3-ft; 1-m) 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 are accelerating beyond those that have occurred over recent millennia. 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. 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. The primary red knot foraging habitats, intertidal flats and sandy beaches, will likely be locally or regionally inundated, but replacement habitats are likely to reform along the shoreline in its new position. However, if shorelines experience a decades-long period of high instability and landward migration, the formation rate of new beach habitats may be slower than the inundation rate of existing habitats. In addition, low-lying and narrow islands (e.g., in the Caribbean and along the Gulf and Atlantic coasts) may disintegrate rather than migrate, 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.

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.

Sea level rise and 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 (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. 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).

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 reduce 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.)

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). Since 2006 a new terminal groin has been constructed at one South Carolina site, three 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, but 2011 legislation authorized an exception for construction of up to four new terminal groins (Rice 2012a, p. 7). 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).

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 Delaware Bay, hard structures also cause or accelerate loss of horseshoe crab spawning habitat (CCSP 2009b, p. 82; Botton *et al. in* Shuster *et al.* 2003, p. 16; Botton *et al.* 1988, entire), and shorebird habitat has been, and may continue to be, lost where bulkheads have been built (Clark *in* Farrell and Martin 1997, p. 24). 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.

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.

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). 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; Atlantic States Marine Fisheries Commission (ASMFC) 1998, p. 28), and is being pursued as a means of restoring shorebird habitat in Delaware Bay following Hurricane Sandy (Niles *et al.* 2013, entire; USACE 2012, entire). Beach nourishment was part of a 2009 project to maintain important shorebird foraging habitat at Mispillion Harbor, Delaware (Kalasz pers. comm. March 29, 2013; Siok and Wilson 2011, entire). 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.

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 bayside 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). Preclusion of overwash also impedes the formation of new red knot habitats. 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 (*Charadrius melodus*) (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, below, 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 3).

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 2012). Instead, red knots tend to concentrate along the ocean coasts (eBird.org 2012), which are more heavily developed (Titus *et al.* 2009, p. 4) even in the Southeast. 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).

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).

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. While some beaches in New Jersey and Delaware have had development removed, new development and redevelopment continues on the Delaware side of the bay (Niles *et al.* 2008, p. 40). New Jersey has not conducted beach nourishment in the Delaware Bay, but 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 (Delaware Department of Natural Resources and Environmental Control 2010, p. 4).

TABLE 3—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.

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 4 gives the miles of developed and undeveloped beach from North Carolina to Texas. (Note the difference between tables 3 and 4; table 3 gives all dry land within 3.3 ft (1 m) of high water, while table 4 is limited to sandy, oceanfront beaches.) Regionwide, about 40 percent of the southeast and Gulf coast is already developed, as shown in table 4. Not all of the remaining 60 percent in the ''undeveloped'' category, however, 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 4), 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 5 shows the extent of southeast and Gulf coast shoreline with shore-parallel structures, beach nourishment, or both.

TABLE 4—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 5—APPROXIMATE SHORELINE MILES OF SANDY, OCEANFRONT BEACH THAT HAVE BEEN MODIFIED BY ARMORING WITH HARD EROSION CONTROL STRUCTURES, 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 navigation channels each year, not including inland waterways. Most inlets and harbors used for commercial navigation in the United States 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). Five of the 12 New Jersey inlets that now exist are stabilized by jetties, and 2 of the unstabilized jetties are maintained by dredging (Nordstrom 2000, p. 20). Table 6 gives the condition of inlets from North Carolina to Texas.

TABLE 6—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

TABLE 6—INLET CONDITION ALONG THE SOUTHEAST ATLANTIC AND GULF COASTS, DECEMBER 2011—Continued
[Rice 2012b, p. 8]

	Existing inlets							Artificially closed
	Number of inlets	Number of modified inlets	Habitat modification type					
			Structures*	Dredged	Relocated	Mined	Artificially opened	
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 3 and 4 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 3 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). 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 3 and 4, about 40 percent of the coast within the U.S. range of the red knot is already developed, and much of this area is 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 due to diminished availability of borrow sediment or accelerated sea level rise.

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 cost and sand availability (Pietrafesa 2012, entire; NJDEP 2010, p. 2; Titus *et al.* 1991, entire; Leatherman 1989, entire). 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).

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).

Table 7 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).

TABLE 7—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

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.

Hurricane Sandy recovery efforts show that retreat is not yet being contemplated as an option on the highly developed coasts of New York and New Jersey (Martin 2012, entire; Regional Plan Association, p. 1), and underscore the looming sand shortage that may preclude the continuation of beach nourishment as it has been practiced over recent decades (Dean 2012, entire).

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. 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. Hard stabilization structures and dredging degrade and often eliminate existing red knot habitats, and in many cases prevent the formation of new shorebird habitats. Beach nourishment may temporarily maintain suboptimal shorebird habitats where they would otherwise be lost as a result of hard structures, but beach nourishment also has adverse effects to red knots and their habitats. Demographic and economic pressures remain strong to continue existing programs of shoreline stabilization, and to develop additional areas, with an estimated 20 to 33

percent of the coast still available for development. However, we expect existing beach nourishment programs will likely face eventual constraints of budget and sediment availability as sea level rises. 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. 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 considered a threat in this part of the subspecies' range. We have little information about coastal development in the red knot's non-U.S. migration and wintering areas, compared to U.S. migration and wintering 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

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 Energy 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, an important shorebird area, prime roosting sites were lost to housing development in the mid-2000s (Niles *et al.* 2008, p. 73). Development is occurring at a rapid rate around Panama Bay, and protections for the bay were recently reduced (Cosier 2012).

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 (bottom-dwelling) prey (WHSRN 2012; Niles *et al.* 2008, p. 97; COSEWIC 2007, p. 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).

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). 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, little is known about development-related threats to the red knot's nonbreeding habitat in other countries. Residential and recreational development is occurring along the Bay of Fundy in Canada, 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. Industrial development is considered a threat to red knot habitat along the north coast of Brazil, but relatively little is known about this region. Urban development is a localized threat to red knot habitats in Panama, 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 begun in many of these areas. However, human population and development continue to grow in many areas. 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), litter, 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). Wrack removal and beach raking both occur on the Gulf beach side of the developed portion of South Padre Island in the Lower Laguna Madre in Texas (USFWS 2012a, p. 28), a well-documented red knot habitat (Newstead *et al.* in press). 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 the ''Wintering and Migration Food'' section of the Rufa Red Knot Ecology and Abundance supplemental document). 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); 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 red knots (see Factor E—Human Disturbance, below). Only minimal disturbance is likely to occur on mid-Atlantic and northern Atlantic beaches because raking in these areas is most prevalent from Memorial Day to Labor Day, when only small numbers of red knots typically occur in this region.

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 roosting and foraging red knots, particularly in the U.S. wintering range. 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. 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. If left uncontrolled, invasive plants can cause a habitat shift from open or sparsely vegetated sand to dense vegetation, resulting in the loss or degradation of red knot roosting habitat, which is especially important during high tides and migration periods. Many invasive species are either affecting or have the potential to affect coastal beaches (USFWS 2012a, p. 27), and thus red knot habitat.

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 new and 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 is 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). Likewise, oyster aquaculture is practiced in Delaware Bay (NJDEP 2011, pp. 1–10), but we have no information to indicate that this activity is affecting red knots.

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 have documented more red knots than were previously known to use this area. 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 red knots at just one site, Cajuais Bank (Carlos *et al.* 2010 pp. 10–11). 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. com. 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) 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. 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.

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. Farming practices around Lagoa do Peixe are degrading habitats at this red knot stopover site, and localized clam farming in Canada could degrade habitat quality and prey availability for transient red knots. Agriculture is contributing to 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 does 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 From Warming Arctic Conditions

For several decades, surface air temperatures in the Arctic have warmed at approximately twice the global rate. Areas above 60 degrees (°) north latitude (around the middle of Hudson Bay) have experienced an average temperature increase of 1.8 to 3.6 degrees Fahrenheit (°F) (1 to 2 degrees Celsius (°C)) since a temperature minimum in the 1960s and 1970s (IPCC 2007c, p. 656). From 1954 to 2003, mean annual temperatures across most of Arctic Canada increased by as much as 3.6 to 5.4 °F (2 to 3 °C), and warming in this region has been pronounced since 1966 (Arctic Climate Impact Assessment (ACIA) 2005, p. 1101). Increased atmospheric concentrations of greenhouse gases are ''very likely'' to have a larger effect on climate in the Arctic than anywhere else on the globe. (The ACIA (2005, pp. 607) report uses likelihood terminology similar, but not identical, to that used by the IPCC; see supplemental document—Climate Change Background—table 1). Under two mid-range emissions scenarios, models predict a mean global temperature increase of 4.5 to 6.3 °F (2.5 to 3.5 °C) by 2100, while the predicted increase in the Arctic is 9 to 12.6 °F (5 to 7 °C). Under both emission scenarios, arctic temperatures are predicted to rise 4.5 °F (2.5 °C) by mid-century. Under the lower of these two emissions scenarios, some of the highest temperature increases in the Arctic (9 °F; 5 °C) in 2100 are predicted to occur in the Canadian Archipelago (ACIA 2005, p. 100), where the red knot breeds.

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 supplemental document—Rufa Red Knot Ecology and Abundance—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).

Arctic Warming—Eco-Regional Changes

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 their migration is not prevented by regional and 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; 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). 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 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).

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 subarctic 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).

Studies have already documented shifts in arctic vegetation. For example, 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). 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).

In addition, 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 ecosystems (e.g., accelerated warming, vegetation changes) immediately adjacent to the coast, but the terrestrial 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). 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. 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). Based on data since 2001, Stroeve *et al.* (2012, p. 1005) suggested that the rate of sea ice loss is accelerating, and the National Aeronautics and Space Administration (NASA 2012) reported that the extent of summer sea ice in 2012 was the smallest on record (during the satellite era). 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.

In addition to changes in plant communities and loss of sea ice, changes in freshwater hydrology of red knot breeding habitats are expected. Arctic freshwater systems, key foraging areas for red knots (Niles *et al.* 2008, p. 27), 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). Rising global temperatures are expected to result in permafrost degradation, possible decline in precipitation, and lowering of water tables, leading to drying of marshes and ponds in the southern parts of the Arctic (ACIA 2005, p. 418; Meltofte *et al.* 2007, p. 35). Conversely, thawing permafrost and increasing precipitation are very likely to increase the occurrence and distribution of shallow wetlands (ACIA 2005, p. 418) in other portions of the Arctic. We cannot predict the likely net changes in wetland availability within the red knot's breeding range over coming decades.

Arctic Warming—Effects on Red Knot Habitat

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, but they note that habitat area may not be the most important factor limiting population size or breeding success.

Potential impacts to shorebirds from changing arctic ecosystems go well beyond the loss of tundra breeding habitat (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). In the southern Arctic, 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). A continuation of warm summers may lead to more and different predators, parasites, and pathogens. 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).

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 °F (1.7 and 5 °C)), 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 prediction 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.

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.

Vegetation changes may go beyond the replacement of tundra by forest and 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 may 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). However, such changes in the Arctic may already be under way, as several studies have found increased shrub abundance, biomass, and cover; increased plant canopy heights; and decreased prevalence of bare ground (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).

Arctic Warming—Summary

Arctic regions are warming much faster than the global average rates, and the Canadian Archipelago is predicted to experience some of the fastest warming in the Arctic. 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. Forests are expected to colonize the southern part of the red knot's current breeding range by 2100, and vegetation throughout the entire breeding range may become taller and denser and with less bare ground, potentially making it unsuitable for red knot nesting. These changes may be accelerated near coastlines, where red knots breed, due to the loss of sea ice that currently cools the adjacent land. Loss of sea ice may also make the central Canadian island habitats more maritime-dominated and, therefore, less suitable for breeding shorebirds. The red knot's breeding range may also experience changes in freshwater wetland foraging habitats, as well as unpredictable but profound ecosystem changes (e.g., interactions among predators, prey, and competitors). 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. 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.* 2013, entire; USACE 2012, entire; Kalasz 2008, entire). 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 the supplemental document ''Factor D: Inadequacies of Existing Regulatory Mechanisms'' 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. 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 inundation rate of existing habitats. 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.

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. Hard stabilization structures and dredging degrade and often eliminate existing red knot habitats, and in many cases prevent the formation of new shorebird habitats. Beach nourishment may temporarily maintain suboptimal shorebird habitats where they would otherwise be lost as a result of hard structures, but beach nourishment also has adverse effects to red knots and their habitats. In those times and places where artificial beach maintenance is abandoned, 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. 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.

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. 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. 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.

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 expected to change, causing the zone of nesting habitat to shift north and perhaps contract. These effects may be exacerbated by loss of sea ice. Arctic freshwater systems, foraging areas for red knots during the nesting season, are particularly sensitive to climate change. Unpredictable but profound ecosystem changes (e.g., interactions among predators, prey, and competitors) may also occur.

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). 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. 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.

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 supplemental document—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). U.S. 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.

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. 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). 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 a partial agreement with the conservation community to lower the annual shorebirds harvest to 22,500 (Eubanks 2011).

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. To date, red knots have been reported only from Guadeloupe (eBird.org 2012).

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 has 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 begun negotiating bag limits and is working on a new regulation that would stop hunting for 5 days following a tropical storm warning, but these measures are not yet in effect (A. Levesque pers. comm. January 8, 2013; Niles 2012c). Significantly, the red knot was recently added to the list of protected species, and hunter education about red knots is in progress (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.

Current Hunting—The Guianas

Band recoveries indicate that red knots are killed commonly for food in some regions of South America, especially in the Guianas (i.e., Suriname, Guyana, and French Guiana). 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).

Rice fields and other impoundments are prevalent in French Guiana and 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 (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, although the magnitude of the harvest is unknown (USFWS 2011e, p. 3). Shorebird hunting is unregulated in French Guiana (A. Levesque pers. comm. January 8, 2013; D. Mizrahi pers. comm. October 16, 2011), which is an overseas region of France.

Harvest of any shorebirds has been illegal in Suriname since 2002, but there is little enforcement. 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. The survey also documented an illegal food trade of shorebirds, including selling to local markets. 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).

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). 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. As described under Species Information above, red knots and other shorebirds 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.

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). 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

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.* in press; 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. Prolonged captivity or excessive handling during these 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).

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 (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. Numbers of birds per catch and total numbers caught over the

season are limited, and careful handling protocols 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. 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. 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. Hunting pressure on red knots and other shorebirds in the northern Caribbean and on Trinidad is unknown. Hunting pressure on shorebirds in the Lesser Antilles (e.g., Barbados, Guadeloupe) is very high, 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. Red knots are no longer being targeted in Barbados or Guadeloupe, and other measures to regulate shorebird hunting on these islands are being negotiated. Much larger numbers (thousands) of red knots occur in the Guianas, where legal and illegal subsistence shorebird hunting is common. About 20 red knot mortalities have been documented in the Guianas, but total red knot hunting mortality in this region cannot be surmised. Subsistence shorebird hunting was also common in northern Brazil, but has decreased in recent decades. We have no evidence that hunting was a driving factor in red knot population declines in the 2000s, or that hunting pressure is increasing. In addition, 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.

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 does occur. While red knot mortality is documented, we have no information to suggest that mortality levels are high enough to affect red knot populations or the subspecies as a whole. 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.

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 (*Calidris alba*) 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 their 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 and environmental contaminants 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 clam *Darina 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 the 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. canutus 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, but direct mortality from these 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).

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 birds 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; Stillman *et al.* 2005, p. 215) and by causing disturbance, thereby potentially affecting red knots' rates of feeding and weight gain.

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). 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. Butler *et al.* (2003, p. 132) demonstrated how recovering raptor populations in North America appear to have led to changes in the migratory strategies of western sandpipers (*C. mauri*), including lower numbers of shorebirds, reduced stopover length, and lower body mass at the more predation-prone sites (as cited in Niles *et al.* 2008, p. 101).

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 are having the greatest impact on shorebirds (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 direct mortality. At key stopover sites, however, localized predation pressures are likely to 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 (*Falcon 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). 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.

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). 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 birds were in low numbers or absent in areas without lemmings because foxes preyed predominately on birds 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. However, 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).

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 lead 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. 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.

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.

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). We have no evidence that such

a link existed 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. The 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. The cyclical nature of this predation on shorebirds is a situation that has probably occurred over many centuries, and under historic conditions likely had no lasting impact on red knot populations. Where and when rodent-predator cycles are operating, we expect red knot reproductive success will also be cyclic. However, these cycles are being interrupted for reasons that are not yet fully clear. The geographic extent and duration of future interruptions to the cycles cannot be forecast but may intensify as the arctic climate changes. 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. 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; 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. However, we have no evidence that parasites have affected red knot populations beyond causing normal, background levels of mortality, 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, infection rates in red knots are low, and health effects are minimal or have not been documented. 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 affecting red knot populations despite some direct mortality. At key stopover sites, however, localized predation pressures are likely to 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. 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. However, these rodent-predator cycles are being interrupted for reasons that are not yet fully clear but may be linked to climate change. 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. Such disruptions have already occurred and may increase due to climate change. The substantial impacts of elevated egg and chick predation on shorebird reproduction are well known, although the red knot's capacity to adapt to long-term changes in predation pressure is unknown. The threat of persistent increases in predation in the Arctic may already be having subspecies-level effects 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, but we cannot forecast how such ecosystem changes are likely to unfold.

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 the other four factors. 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.

A comprehensive discussion of international, Federal, State, and local laws, regulations, policies, and treaties that apply to the red knot is available as a supplemental document (''Factor D: The Inadequacy of Existing Regulatory Mechanisms'') on the Internet at *http://www.regulations.gov* (Docket No. FWS–R5–ES–2013–0097; see **ADDRESSES** section for further access instructions). We provide a brief summary below.

In Canada, the Species at Risk Act provides protections for the red knot and its habitat, both on and off Federal lands. The red knot is afforded additional protections under the Migratory Birds Convention Act and by provincial law in four of Canada's Provinces. In other areas outside of the United States' jurisdiction, 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 (see Factor B—Hunting). 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 (see Factor A—International Coastal Development) and disturbance, oil spills, environmental contaminants, and wind energy development (see Factor E).

Within the United States, the Migratory Bird Treaty Act of 1918 (16 U.S.C. 703 *et seq.*) (MBTA) and state wildlife laws protect the red knot from direct take resulting from scientific study and hunting (see Factor B). The MBTA is the only Federal law in the United States currently 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 previously devastated by market hunting (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 under Factors A, B, C, and E. However, the Sikes Act (16 U.S.C. 670), covering military bases, the National Park Service Organic Act of 1916, as amended (NPSOA), covering national parks and seashores, and the National Wildlife Refuge System Improvement Act of 1997 (NWRSIA), covering national wildlife refuges, do provide protection for the red knot from habitat loss and inappropriate management on Federal lands.

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 (as endangered) and Georgia (as rare, a category of protected species). The New Jersey Endangered and Non Game 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 in New Jersey 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). 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.

As discussed 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. Shoreline stabilization is often federally funded (e.g., through the Water Resources Development Acts) or authorized (e.g., under section 404 of the Clean Water Act (33 U.S.C. 1251 *et seq.*) and sections 9 and 10 of the Rivers and Harbors Act (33 U.S.C. 403 *et seq.*)). Federal funding or authorization for a project triggers several environmental requirements that may afford some protections to red knots or their habitats, but several of these are nonregulatory in nature (e.g., the National Environmental Policy Act 42 U.S.C. 4321 *et seq.* (1969) (NEPA); Executive Order 13186 (Responsibilities of Federal Agencies to Protect Migratory Birds)). One regulatory measure is the Coastal Barrier Resources Act (Pub. L. 97–348) (96 Stat. 1653; 16 U.S.C. 3501 *et seq.*) (CBRA), as amended. The CBRA 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.

The Coastal Zone Management Act of 1972 (Pub. 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. All eligible States in the red knot's U.S. range (including the Great Lakes) have approved Coastal Zone Management Plans (National Oceanic and Atmospheric Administration (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). Thirteen of 18 Atlantic or Gulf coast States (72 percent) range allow for new hard structures along the oceanfront beach, and 16 of these 18 States allow armoring of bays and sounds (Rice 2012a, p. 7; Titus 2000, p. 743). As of 2000, 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). Due to the CZMA's Federal consistency provision, Federal agencies also generally follow each State's policies in determining if coastal projects may be federally funded or authorized.

Other threats to habitat and food supplies and from disturbance are partially, but not fully, abated by various State and Federal regulations. First, State regulations provide varying levels of protection from impacts

associated with beach grooming (i.e., mechanical raking or cleaning), but we do not have comprehensive information for each State. Above the high tide line, beach grooming 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. Second, several Federal and State regulatory and nonregulatory measures are in effect to stem the introductions and effects of invasive and harmful species (e.g., Executive Order 13112; the Plant Protection Act of 2000 (Pub. L. 106–224); the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990 (Pub. L. 101–646); the National Invasive Species Act of 1996 (Pub. L. 104–332); the U.S. Coast Guard's (USCG) ballast water regulations (77 FR 17254); the Lacey Act (18 U.S.C. 42, 50 CFR part 16); the Clean Water Act; and the Harmful Algal Bloom and Hypoxia Amendments Act of 2004 (Pub. L. 108–456)), but collectively these measures 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. Third, although threats to the horseshoe crab egg resource remain (see Factor E—Reduced Food Supplies), the current regulatory management of the horseshoe crab fishery (e.g., the Adaptive Resource Management (ARM) framework adopted by the ASMFC, a governing body established by the Atlantic Coastal Fisheries Cooperative Management Act of 1993) is adequately addressing threats to the knot's Delaware Bay food supply from direct harvest of horseshoe crabs. Fourth, 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 through the MBTA, Sikes Act, NPSOA, NWRSIA, and State or local laws and policies. And fifth, relatively strong Federal laws likely reduce risks to red knots from oil spills (e.g., the Oil Pollution Act of 1990 (OPA) (33 U.S.C. 2701 *et seq.*)) and pesticides (e.g., the Federal Insecticide, Fungicide, and Rodenticide Act (7 U.S.C. 136 *et seq.*)). The OPA requires contingency planning by Federal, state, and local governments and industry groups, and includes penalties for regulatory noncompliance. Under the OPA, the EPA regulates above ground storage facilities and the USCG regulates oil tankers, which have been transitioning to double hulls since 1992 under international agreements. In addition, oil and gas operations on the Outer Continental Shelf (OCS) are regulated (50 CFR parts 203–291) by the Bureau of Safety and Environmental Enforcement (BSEE) within the Department of the Interior (DOI). Despite the relatively robust oil spill and pesticide regulations in place, these laws have not been sufficient to prevent documented shorebird mortalities and other impacts in recent decades.

In addition to above-mentioned regulatory mechanisms addressing threats to habitat, food resources, and from disturbance, there are Federal laws and policies to reduce the red knot's collision risks from new terrestrial and offshore wind turbine development (e.g., construction and operation). The MBTA applies to all Federal and non-Federal activities that result in the ''take'' of migratory birds. To assist wind developers comply with MBTA, the Service's voluntary Land-Based Wind Energy Guidelines provide a structured, scientific process for addressing wildlife conservation concerns at all stages of land-based wind energy development (USFWS 2012d, p. vi). In addition to the MBTA, other Federal regulatory mechanisms and nonregulatory policies (e.g., NEPA, Executive Order 13186, NSPOA, NWRSIA, and section 10 of the Endangered Species Act) may apply to terrestrial wind energy development, depending on the nature of the Federal nexus, if any, in turbine construction and operation. 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 OSC for wind and other types of renewable energy development. Under NEPA, DOI 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). In addition to these Federal provisions, some states have policies in place to address risks to red knots from wind energy development (see supplemental document—Factor D). However, as described below in Factor E, despite these state and Federal laws, policies, and voluntary guidelines, we expect some level of red knot mortality to occur from the 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 (International Wader Study Group 2003, p. 10). The life history of long-distance, long-hop migrant shorebirds indicates that the availability of abundant food resources at temperate stopovers is critical for completing their annual cycle (USFWS 2003, p. 4). 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 red knot 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—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).

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 lines of evidence suggest that the rufa red knot also faces threats to its food resources throughout its range. The following discussion addresses known or likely threats to the abundance or quality of red knot prey. Potential food shortages caused by asynchronies (''mismatches'') in the red knot's annual cycle are discussed in the next section.

Also see Factor A—Agriculture and Aquaculture, above, regarding clam farming practices in Canada that impact red knot prey resources by modifying suitable foraging habitat via sediment sifting. Although threats to food quality and quantity are widespread, 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 to cope with this threat.

Food Availability—Ocean Acidification

During most of the year, bivalves and other mollusks are the primary prey for the red knot (see the ''Migration and Wintering Food'' section of the Rufa Red Knot Ecology and Abundance supplemental document). Mollusks in general are at risk from climate change-induced ocean acidification (Fabry *et al.* 2008, pp. 419–420). Oceans become more acidic as carbon dioxide emitted into the atmosphere dissolves in the ocean. The pH (percent hydrogen, a measure of acidity or alkalinity) level of the oceans has decreased by approximately 0.1 pH units since preindustrial times, which is equivalent to a 25 percent increase in acidity. By 2100, the pH level of the oceans is projected to decrease by an additional 0.3 to 0.4 units under the highest emissions scenarios (NRC 2010, pp. 285–286). As ocean acidification increases, the availability of calcium carbonate declines. Calcium carbonate is a key building block for the shells of many marine organisms, including bivalves and other mollusks (USEPA 2012; NRC 2010, p. 286). Vulnerability to ocean acidification has been shown in bivalve species similar to those favored by red knots, including mussels (Gaylord *et al.* 2011, p. 2586; Bibby *et al.* 2008, p. 67) and clams (Green *et al.* 2009, p. 1037). 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), the 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).

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 ocean 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). With climate change, interactions between temperature and pH may cause detrimental ecological changes to red knot prey species at both wintering and migration stopover areas.

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 °F (1.1 to 6.4 °C) by 2100 (USEPA 2012). Coastal waters are ''very likely'' to continue to warm by as much as 4 to 8 °F (2.2 to 4.4 °C) 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).

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).

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 aspects of climate change. For example, freshwater inputs, tidal prisms (the volume of water in an estuary between high and low tide), and salinity regimes may be much altered, which could significantly alter the composition of estuarine communities. Furthermore, rising sea levels are expected to affect the physical shape (e.g., dimensions, configuration) of estuaries, changing their sediment compositions. This habitat change in

turn would change invertebrate densities and community composition, thus affecting shorebirds (Rehfisch and Crick 2003, p. 88; Najjar *et al.* 2000, p. 225), such as the red knot.

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 (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).

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 on the Delaware side of Delaware Bay and can affect spawning habitat for horseshoe crabs. Although beach nourishment generally preserves habitat value 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. Avissar (2006, p. 427) modeled nourished versus control beaches and found that nourishment may compromise egg development and viability. Despite possible drawbacks, beach nourishment has been recommended to prevent the 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) and is being pursued as a means of restoring shorebird habitat in Delaware Bay following Hurricane Sandy (Niles *et al.* 2013, entire; USACE 2012, entire). In areas of Delaware Bay with hard stabilization structures or high erosion rates, beach nourishment may be the only option for maintaining habitat.

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 used 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 the ''Migration and Wintering Habitat'' section of the Rufa Red Knot Ecology and Abundance supplemental document, and Factor A—Beach Cleaning). Kluft and Ginsberg (2009, p. vi) found that ORVs killed and displaced 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 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 (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 (known as Adaptive Resource Management, or 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 populations over the next 5 to 10 years. 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 the ''Wintering and Migration Food'' section of the Rufa Red Knot Ecology and Abundance supplemental document). 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 weight of evidence supporting each of these linkages is discussed 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 2009, 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* New Jersey Audubon Society 2007, p. 25); see Factor B, above.

During the 1990s, reported commercial harvest of horseshoe crabs on the Atlantic coast of the United States increased dramatically. 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 2009, 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. The increase in harvest of horseshoe crabs during the 1990s was largely due to increased use as conch bait (ASMFC 2009, 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 2009, p. 2). The ''LAL test'' is currently the worldwide standard for screening medical equipment and injectable drugs for bacterial contamination (ASMFC 2009, p. 2; ASMFC 1998, p. 12). Horseshoe crab blood is obtained from adult crabs that are released alive after extraction is complete (ASMFC 2009, p. 2) or that are sold into the bait market (ASMFC 2009, p. 18). The ASMFC previously assumed a constant 15 percent mortality rate for bled crabs that are not turned over to the bait fishery (ASMFC 2009, p. 3) but now considers a range from 5 to 30 percent mortality (ASMFC 2012a, p. 6) more appropriate. 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 considers implementing action to reduce mortality if estimated mortality exceeds a threshold of 57,500 crabs. This threshold has been exceeded several times, but thus far the ASMFC has opted only to issue voluntary guidelines to the biomedical industry (ASMFC 2009, p. 18). The ASMFC implemented key reductions in the bait harvest in 2000, 2004, and 2006 (ASMFC 2009, p. 3), and several member States have voluntarily restricted harvests below their allotted quotas (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 2009, p. 1) from 1998 to 2011 by over 75 percent (table 9). Further, a growing number of horseshoe crabs are being biomedically bled first before being used as bait; because such crabs count against harvest quotas (ASMFC 2012a, p. 6), this practice helps reduce total mortality rates. 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 8 and 9) 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 8 and 9 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 2009, 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 2009, p. 17).

Table 9 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 2009, 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 8 and 9 do not reflect the changing sex ratio of crabs harvested in the Delaware Bay Region (S. Michels pers. comm. February 15, 2013), which has shifted away from the harvest of females since management began. In 2013, the first year that the harvest level was determined using the ARM, the quota in the Delaware Bay Region is set at 500,000 males and 0 females (ASMFC 2012b, p. 1); however, we do not yet have access to the actual number of crabs removed in 2013 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 2012a, p. 13). From 2006 to 2011, New Jersey opted not to use its 100,000-crab quota by imposing a moratorium, which the State is now considering lifting amid considerable controversy between environmental and fishing groups (Augenstein 2013, entire; 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 2009, pp. 18, 41) to 545,164 in 2011 (ASMFC 2012a, p. 6). Total estimated mortality of biomedical crabs for 2011 was 80,827 crabs (using a 15 percent post-release estimated mortality; see table 9), with a range of 31,554 to 154,737 crabs (using 5 to 30 percent estimated mortality) (ASMFC 2012a, p. 6). 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.

To put the reported harvest numbers in context, two recent assessments using different methods both estimated the population of horseshoe crabs in the Delaware Bay Region at about 20 million adults, with approximately twice as many males as females (Sweka pers. comm. May 30, 2013; Smith *et al.* 2006, p. 461). Therefore, recent annual harvests of roughly 200,000 horseshoe crabs from the Delaware Bay Region represent about 1 percent of the adult population.

TABLE 8—REPORTED ATLANTIC COAST HORSESHOE CRAB LANDINGS (POUNDS), 1970 TO 2011

[NOAA 2012d]

Year	Total pounds reported to NMFS	Year	Total pounds reported to NMFS
1970	15,900	1991	385,487
1971	11,900	1992	321,995
1972	42,000	1993	821,205
1973	88,700	1994	1,171,571
1974	16,700	1995	2,416,168
1975	62,800	1996	5,159,326
1976	2,043,100	1997	5,983,033
1977	473,000	1998	6,835,305
1978	728,500	1999	5,246,598
1979	1,215,630	2000	3,756,475
1980	566,447	2001	2,336,645
1981	326,695	2002	2,772,010
1982	526,700	2003	2,624,248
1983	468,600	2004	974,425
1984	225,112	2005	1,421,957
1985	614,939	2006	1,548,900
1986	635,823	2007	1,804,968
1987	511,758	2008	1,315,963
1988	688,839	2009	1,830,506
1989	1,106,645	2010	869,630
1990	519,057	2011	1,497,462

TABLE 9—REPORTED ATLANTIC COAST HORSESHOE CRAB LANDINGS (POUNDS AND CRABS), 1998 TO 2011

[(A. Nelson Pers. Comm. February 22, 2013 and November 27, 2012; ASMFC 2012a, pp. 6, 13; NOAA 2012d; ASMFC 2009, pp. 38–41); ND = No Data Available]

Year	Total pounds reported to NMFS (from Table 8)	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	5,246,598	2,600,914	1,013,996	ND
2000	3,756,475	1,903,415	767,988	ND

TABLE 9—REPORTED ATLANTIC COAST HORSESHOE CRAB LANDINGS (POUNDS AND CRABS), 1998 TO 2011—Continued

[(A. Nelson Pers. Comm. February 22, 2013 and November 27, 2012; ASMFC 2012a, pp. 6, 13; NOAA 2012d; ASMFC 2009, pp. 38–41); ND = No Data Available]

Year	Total pounds reported to NMFS (from Table 8)	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
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	1,548,900	840,944	270,241	49,182
2007	1,804,968	827,554	169,255	63,432
2008	1,315,963	660,794	190,828	63,285
2009	1,830,506	756,484	250,699	60,642
2010	869,630	604,548	165,852	75,428
2011	1,497,462	650,539	195,153	80,827

Link A, Part 2—Horseshoe Crab Population Levels

Through stock assessments, ASMFC analyzes horseshoe crab data from many different independent surveys and models (ASMFC 2004, pp. 14–24; ASMFC 2009, 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 2009, p. 23). 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.

Though no formal horseshoe crab stock assessment has been conducted since 2009, the ASMFC's Delaware Bay Ecosystem Technical Committee recently reviewed current 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 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 several years of reduced female harvests) (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 2009, 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. Leschen and Correia (2010a, p. 135) reported mortality rates of nearly 30 percent, although this result has been disputed (Dawson 2010, pp. 2–3; Leschen and Correia 2010b, pp. 8–10). The ASMFC now considers a range from 5 to 30 percent mortality (ASMFC 2012a, p. 6).
  - 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). The New Jersey incident was small, and no other violations are known to have occurred in New Jersey (D. Fresco pers. comm. November 9, 2012). Although the poaching in New York involved substantial numbers of crabs, New York waters are outside the Delaware Bay Region and should not affect population

trends in this Region. Together, though, these incidents hint that illegal harvest may be a factor, although the ASMFC law enforcement committee reported very few problems or issues in the past few years (M. Hawk pers. comm. April 29, 2013).

○ The harvest of horseshoe crabs from Federal waters that are not landed in any state, but exchanged directly to a dependent fishery, is unregulated, and, therefore, the magnitude of any such harvest is unknown (ASMFC 1998, p. 27). However, there is no evidence that such boat-to-boat transfers are occurring, and the level of any such unreported harvest 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).

○ The extent of horseshoe crab mortality due to bycatch from other fisheries is unknown (ASMFC 1998, pp. 22, 26); however, at least one State does regulate and limit such bycatch (Virginia Marine Resources Commission Chapter 4 VAC 20–900–10 *et. seq.*), and horseshoe crabs caught as bycatch in the Carl N. Shuster Jr. Horseshoe Crab Reserve must be returned to the water (NOAA 2001, p. 8906).

• 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.

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 recently emerged—the proposed importation of nonnative horseshoe crab species for use as bait. 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 on the Federal level (i.e., under the Lacey Act, see supplemental document—Factor D), although Congress is deliberating legislation to expand that authority (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 2013, entire), although no such State bans have yet gone into effect.

Asian horseshoe crab species are themselves in decline (ASMFC 2013, 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 (Bauers 2013b; 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. Based on data through 2011, numbers of spawning females have not increased or decreased, while numbers of spawning males showed a statistically significant increase. Though not statistically significant, female crab trends were negative in Delaware and positive in New Jersey (Zimmerman *et al.* 2012, pp. 1–2). 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).

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 horseshoe crab abundance. 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 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 recently reviewed the most current egg density data from both States. The committee 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 a negative trend from 2005 to 2012 (A. Dey pers. comm. October 12, 2012).

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 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). 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).

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, due to poor foraging and weather conditions, red knot fueling (temporal patterns and rates of weight gain) proceeded as normal from 1997 to 2002, except in 2000, but not in 2003 or 2005.

Dey *et al.* (2011a, p. 6) 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 2011). 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. This relationship holds with the addition of 2012 data and 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 returned to 1990s levels (A. Dey pers. comm. October 12, 2012).

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. 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 populations 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 the 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.

Baker *et al.* (2004, pp. 878–897) found that average 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 supplemental document—Rufa Red Knot Ecology and Abundance—tables 2 and 10). 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 Conserve Wildlife Foundation 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. 122). The ARM uses competing models to test hypotheses and eventually reduce uncertainty about the influence that 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 recovery targets starting in 2013 (ASMFC 2012e, p. 4).

As long as the ARM is in place and functioning as intended, ongoing horseshoe crab harvests should not be a threat to the red knot. However, the harvest regulations recommended by the ARM require data from two annual, baywide monitoring programs—the trawl survey conducted by the Virginia Polytechnic Institute (Virginia Tech) and the Delaware Bay Shorebird Monitoring Program. No secure funding is in place for either of these programs. For example, in fall 2012, the trawl survey had to be scaled back due to lack of funds (ASMFC 2012d, p. 8). Reduced survey efforts may impact the ability of the ASMFC to implement the ARM as intended (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.

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). 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.

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. 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 (the ARM), the present horseshoe crab harvest is not considered a threat to the red knot. However, it is not yet known if the horseshoe crab egg resource will continue to adequately support red knot populations over the next 5 to 10 years. In addition, implementation of the ARM could be impeded by insufficient funding.

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. 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, 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), ocean acidification to which mollusks are particularly vulnerable, physical habitat changes from climate change affecting invertebrate communities, possibly increasing rates of mollusk diseases due to climate change, invasive marine species from ballast water and aquaculture, and the burial and crushing of invertebrate prey from sand placement and recreational activities. 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

For 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 increases reproductive success. 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). 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). 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).

Particularly for species like the red knot that show fidelity to sites with ephemeral food and habitat resources used to fuel long-distance migration, migrating animals may incur fitness consequences if their migration timing and the availability of resources do not coincide (i.e., are asynchronous or ''mismatched''). 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 (periodic life-cycle events) related to global climate change (McGowan *et al.* 2011a, p. 2; Smith *et al.* 2011a, p. 575; Meltofte *et al.* 2007, p. 36).

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 prey species (McGowan *et al.* 2011a, p. 16); thus, shorebirds are vulnerable to worsening asynchronies due to climate change. 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 (caused by factors inside the animal) 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 these endogenous rhythms to the outside world (Cadée *et al.* 1996, p. 82). Photoperiod 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 *C.c. 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). While 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 environmental factors); based on the experiments with captive *C.c. canutus,* it is reasonable to conclude that these factors will constrain the knot's ability to adapt to the shifting temporal and geographic

patterns of favorable food and weather conditions that are expected to occur with global climate change.

Looking at data from Northern Europe from 1923 to 2008 for 43 taxonomically diverse birds (including shorebirds but not *Calidris canutus*), Petersen *et al.* (2012, p. 65) found that short-distance migrants arrived an average of 0.38 days earlier per year, while the spring arrival of long-distance migrants had advanced an average of 0.17 days per year. Pooling both groups, spring arrival had shifted an average of 3 weeks earlier over the 80-year study period. Changes in environmental conditions (e.g., temperature, precipitation) during winter and spring explained much of the change in phenology. These findings suggest that 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 that 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) caused by the formation of new and novel assemblages of bird species that did not previously occur together in space and time.

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 observed, 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.

Delaware Bay—Timing of Horseshoe Crab Spawning

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.* 2011, 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 species migrations (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 supplemental document—Climate Change Background). 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 °F (14 °C) during May, and daily water temperatures were not consistently above 59 °F (15 °C) until late May. In the other years, daily water temperatures were consistently above 59 °F (15 °C) 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 1.8 °F (1 °C) 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 supplemental document—Climate Change Background) 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 °F (1.1 °C) 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 °F (1.4 to 2.2 °C) in winter and 1.5 to 3.5 °F (0.8 to 1.9 °C) in summer (USGCRP 2009, p. 107). Coastal waters are ''very likely'' to continue to warm by as much 4 to 8 °F (2.2 to 4.4 °C) 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 °F (2.2 to 4.4 °C).

Asynchronies—Other Spring Stopover Areas

Outside of Delaware Bay, migrating red knots feed primarily on bivalves and other mollusks. 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. 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. Thus, red knots are vulnerable to changes in the reproductive timing and the geographic ranges of their prey, such as could be precipitated by climate change (see examples of blue mussel spat in Virginia and horseshoe crab eggs in Delaware Bay discussed above).

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), but the cause is unknown.

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 other species is likely to shift earlier in response to warming water temperatures (Philippart *et al.* 2003, p. 2171).

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* International Wader Study Group 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.

Asynchronies—Fall Migration

Preliminary results of efforts to track red knot migration routes using geolocators found that two of three birds likely detoured from normal migration paths to avoid adverse weather during the fall migration (Niles *et al.* 2010a, p. 129). These birds travelled an extra 640 to 870 mi (1,030 to 1,400 km) to avoid storms. 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 in the supplemental document ''Climate Change Background,'' 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).

Asynchronies—Breeding Grounds

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). Insects 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 Loss from Warming Arctic Conditions, above, for recent rates and predictions of arctic warming and the eco-regional classification of the red knot's current breeding range.) However, forecasting the effects of changing arctic weather patterns on shorebirds is associated with high uncertainty. 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. 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).

Breeding Grounds—Insect Prey

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. 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 decreased prey abundance 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 best available scientific data indicate that red knots will likely be negatively affected by increased asynchronies between the 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 these predicted ecosystem changes in the Arctic.

Breeding Grounds—Snowmelt

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). These findings 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.

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 red knots. Canadian researchers reported that most Arctic-breeding birds failed to breed successfully in 2004 (Niles *et al.* 2005, p. 4).

Trends toward earlier snowmelt dates have been documented in North America in recent years (IPCC 2007b, p. 891). Earlier snowmelts in the Arctic from 2020 to 2080 are ''very likely'' (ACIA 2005, p. 470). As years of late snowmelt have typically had an adverse effect on shorebird breeding, reduced frequency of late-melt years may have a short-term benefit to red knots. Warming trends may benefit arctic shorebirds in the short term by increasing both survival and productivity (Meltofte *et al.* 2007, p. 7). However, it is unknown how red knots would be affected if snowmelts become substantially earlier than the start of the breeding season (see Ims and Fuglei 2005 for consideration of the complex ways tundra ecosystems may respond to climate change).

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. 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. 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. However, the factors that caused delays in the spring migrations of red knots from Argentina and Chile are still unknown, and we have no information to indicate if this delay will reverse, persist, or intensify.

Superimposed on this existing threat of late arrivals in Delaware Bay are new threats of asynchronies emerging due to climate change. 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. 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. Climate change may also cause shifts in the period of optimal arctic conditions relative to the time period when red knots currently breed.

The red knot's adaptive capacity to deal with numerous changes in the timing of resource availability across its geographic range is largely unknown. A few examples suggest some flexibility in migration strategies. However, available information suggests that the timing of the red knot's annual cycle is controlled at least partly by celestial and endogenous cues, while the reproductive seasons of prey species, including horseshoe crabs and mollusks, are largely driven by environmental cues such as water temperature. 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 Reduced Food Availability, above). Effects to red knots from vehicle and pedestrian disturbance can also occur during construction of shoreline stabilization projects including beach nourishment. 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, p. 17; Burger 1986, p. 124) and by beach raking (also called grooming or cleaning, see Factor A above). In Delaware Bay, red knots could also potentially be disturbed by hand-harvest of horseshoe crabs (see Reduced Food Availability, above) 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 in press, entire; Forys 2011, entire; Burger *et al.* 2004, entire), but such management is not occurring throughout the red knot's range.

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). 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).

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. 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 recently 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).

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, 2012; 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 (in press) 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 eight percent for monospecific red knot flocks. 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. 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 (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. 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 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. This finding would apply to red knots, as they forage in the intertidal zone. At two Atlantic coast sites in New Jersey, Burger and Niles (in press) found that 70 percent of shorebird flocks with red knots flew when disturbed, 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. 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 that flew away and did not return in response to disturbance increased between 1982 and 2002 (Burger *et al.* 2004, p. 286).

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). 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). 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 is 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. Excessive disturbance has been shown to preclude shorebird use of otherwise preferred habitats and can impact energy budgets. 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 decline (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 availably 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 shorebirds are. 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. 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. 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., Koch pers. comm. March 5, 2013; 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, tolerance of human disturbance, and generally stable or increasing populations. However, outside of Delaware Bay, there is typically less overlap between the diets of red knots (specializing in small, buried, intertidal mollusks) 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).

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.

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). *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).

A red tide event occurred in October 2009 along the Gulf coast of Texas during the period that 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 that had been 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, when it was 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 just 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 *et al.* in press). These levels are extremely high (Newstead *et al.* in press; 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 *et al.* in press). 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 (Newstead *et al.* in press).

Although no 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). Brevitoxins 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).

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).

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 impacts to red knots from HABs have been documented only in Texas, although a large die-off in Uruguay may have also been linked to an HAB. 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. Combined with other threats, ongoing and possibly increasing mortality from HABs may affect the red knot at the population level.

Factor E—Oil Spills and Leaks

The red knot has the potential to be exposed to oil spills and leaks throughout its migration and wintering range. Oil, as well as spill response activities, can directly and indirectly affect both the bird and its habitat through several pathways. Red knots can be exposed to petroleum products via spills from shipping vessels, leaks or spills from offshore oil rigs or undersea pipelines, leaks or spills from onshore facilities such as petroleum refineries and petrochemical plants, and beach-stranded barrels and containers that can fall from moving cargo ships or offshore rigs. Several key red knot wintering or stopover areas also contain large-scale petroleum extraction, transportation, or both activities. With regard to potential effects on red knot 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.

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 Asynchronies, above) (Henkel *et al.* 2012, p. 681; Gunnarsson *et al.* 2005, p. 2320; Myers *et al.* 1987, pp. 21–22).

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

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).

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 10 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 been documented (Philadelphia Area Committee 1998, Annex E).

TABLE 10—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 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 11 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 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 11—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

TABLE 11—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—Continued

[BSEE 2012]

Year	Number of incidents
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 the ''International Laws and Regulations'' section of the Factor D supplemental document). 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 12.

TABLE 12—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).

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).

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). At the end of July 2010, approximately 625 mi (1,006 km) of Gulf of Mexico shoreline were oiled. By the end of October, 93 mi (150 km) were still affected by moderate to heavy oil, and 483 mi (777 km) of shoreline were affected by light to trace amounts of oil (USFWS 2012a, p. 36; Unified Area Command 2010). These numbers reflect weekly snapshots of shorelines experiencing impacts from oil and do not include cumulative impacts or shorelines that had already been cleaned (M. Bimbi pers. comm. November 1, 2012; USFWS 2012a, p. 36). Limited cleanup operations were still ongoing throughout the spill area in November 2012 (USFWS 2012a, p. 36). A Natural Resources Damage Assessment (NRDA) to assess injury to wildlife resources is in progress (Natural Resource Trustees 2012, pp. 8–9), but due to the legal requirements of the NRDA process, avian injury information, including any impacts to red knots, has not been released (P. Tuttle pers. comm. November 8, 2012).

Oil Spills—South America

South America—Brazil and Patgonia

Threats to red knot habitat in Maranhão, Brazil include oil pollution as well as habitat loss (see Factor A above) from offshore petroleum exploration on the continental shelf (WHSRN 2012; Niles *et al.* 2008, p. 97; COSEWIC 2007, p. 37).

Oil pollution is also 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 percent of red knots were polluted with oil during a study in 1979 (Niles *et al.* 2008, p. 98). 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. In recent years, oil operations have been decreasing in Chile around Bahía Lomas, but increasing 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 10 of the 12 years since 2000 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. 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. No direct evidence exists of red knots being affected by oil pollution, but it remains a risk (Niles *et al.* 2008, pp. 98–99).

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. 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. 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.

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. Contaminant levels have generally declined in the eggs of horseshoe crabs from 1993 to 2001, suggesting that contaminants are 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 (2007b, 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. 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—Florida

A piping plover was found among dead shorebirds discovered on a sandbar near Marco Island, Florida, following the county's aerial application of the organophosphate pesticide Fenthion for mosquito control in 1997 (Pittman 2001; Williams 2001). The USEPA has subsequently banned the use of Fenthion (American Bird Conservancy 2012b). Marco Island also supports an important concentration of red knots, but it is unknown if any red knots were affected by Fenthion at this or other sites.

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 2012).

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). 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. 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. One preliminary study suggests organochlorines and trace metals are not elevated in Delaware Bay shorebirds, although this finding cannot be confirmed without updated testing. 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.

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. Shorebirds have been impacted by pesticide exposure, but use of the specific chemical that caused a piping plover death in Florida has subsequently been banned in the United States. 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. Finally, localized urban pollution has been shown to impact South American red knot habitats, 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

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). Wind turbines can have a direct (e.g., collision mortality) and indirect (e.g., migration disruption, displacement from habitat) impact on shorebirds. We have no information on wind energy development trends in other countries, but risks of red knot collisions would likely be similar wherever large numbers of turbines are constructed along migratory pathways, either on land or offshore.

Wind Energy—Offshore

In 2007, the DOI's Bureau of Ocean Energy Management (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 (Department of Energy (DOE) and BOEMRE 2011, p. iii). In 2011, the U.S. Department of Energy (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, 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 the United States. However in 2010, BOEM issued the first lease to build a wind facility in Federal waters, authorizing the Cape Wind Energy Project off the southeast coast of Massachusetts (DOE and BOEMRE 2011, p. 41). Mapping from BOEM (2013) shows additional leases have been executed for two smaller areas about 10 and 16 mi (16 and 26 km) southeast of Atlantic City, New Jersey and for a larger area about 14 mi (22 km) southeast of the mouth of the Delaware Bay. Offshore wind projects have been proposed off the coasts of Texas and Northern Mexico (Newstead *et al.* in press), and five States recently entered an agreement with the Federal Government to facilitate wind energy development in the Great Lakes (Council on Environmental Quality 2012, p. 1).

Analysis by the DOE shows the potential for wind energy, and offshore wind in particular, to reduce greenhouse gas emissions in a rapid and cost-effective manner (DOE and BOEMRE 2011, p. 5). However, large-scale installation of offshore 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), and 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). Avian collision risks are related to both the total number of turbines and the height of the turbines (Kuvlesky *et al.* 2007, p. 2488; NRC 2007, p. 138; 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). As approved, the Cape Wind Energy facility will include 130, 3.6-megawatt (MW) wind turbines, each with a maximum blade height of 440 ft (134 m) above sea level (BOEM 2012, 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 along 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

increase with the use of updated estimates of population size (see the ''Population Surveys and Estimates'' section of the Rufa Red Knot Ecology and Abundance supplemental document) 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), we have only preliminary information on the actual red knot collision risk posed by offshore wind turbines (e.g., based on collision rates in other countries, the effects of weather and artificial lighting, 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 upon buildout of offshore wind infrastructure.

Research from Europe, where several offshore wind facilities are in operation, suggests that bird collision rates with offshore turbines may be higher than for turbines on land. For various waterbird species, annual collision rates from 6.7 to 19.1 birds per turbine have been reported (Kuvlesky *et al.* 2007, p. 2489). Collision risks depend on turbine design and configuration, geography, 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 (Burger *et al.* 2011, p. 340; Kuvlesky *et al.* 2007, p. 2488; NRC 2007, p. 138).

A number of studies from Europe also suggest that wind facilities could displace migrating waterfowl and shorebirds, create barriers to migration, and alter flight paths between foraging and roosting habitats (Kuvlesky *et al.* 2007, p. 2489). Such effects are thought to extend at least 1,969 ft (600 m) from the wind facility, but could extend 1.2 to 4.5 mi (2 to 4 km) for some species (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 survival as well as the timing of arrival at stopover areas (see Asynchronies, above). However, displacement of birds from habitats around wind facilities somewhat reduces the risks of turbine collisions.

Although little shorebird-specific information is available, the effect of weather on migrating bird flight altitudes has been well documented through the use of radar and thermal imagery. Numerous studies indicate that the risk of bird collisions with wind turbines (including offshore turbines) increases as weather conditions worsen and visibility decreases (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). Avoidance behavior is likely to vary according to conditions. It is reasonable to expect that avoidance rates would be much reduced at times of poor visibility, in poor weather, at night (Chamberlain *et al.* 2006, p. 199), and under varying structure illumination conditions (Drewitt and Langston 2006, p. 31; Hüppop *et al.* 2006, p. 105). 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).

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). 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 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).

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 rounds 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).

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 geolocator results suggest that short-distance migrants may actually face greater collision hazards from wind development in this region. The 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 the 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 2012d, p. 3) that may prompt lower migration altitudes and hinder avoidance behaviors.

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 13). 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). 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), a scenario that would substantially reduce U.S. carbon dioxide emissions (DOE 2008, p. 107). 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 the 2003 voluntary guidelines to provide a structured, scientific process for addressing wildlife conservation concerns at all stages of land-based wind energy development (USFWS 2012e, p. vi).

TABLE 13—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.]

State	1999		2012	
	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

TABLE 13—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).—Continued

[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.]

State	1999		2012	
	Megawatts	Estimated number of turbines	Megawatts	Estimated number of turbines
Montana	0.100	1	645	327
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

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) calculated an average of 2.19 avian fatalities per turbine per year for all bird species combined, and found that shorebirds constituted only 0.2 percent of the total. 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).

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 New Jersey Audubon Society (NJAS (also known as New Jersey Audubon) 2009, entire; NJAS 2008a, entire; NJAS 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 (Newstead *et al.* in press). 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 (Newstead *et al.* in press). Shorebirds departing the coast for destinations along the central flyway (see the ''Migration—Northwest Gulf of Mexico'' section of the Rufa Red Knot Ecology and Abundance supplemental document) 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 (Newstead *et al.* in press).

Wind Energy—Summary

We analyzed shorebird mortality at land-based wind turbines in the United States, 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. We have no 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. 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. Shorebirds as a group have constituted only a small percentage of collisions with U.S. turbines in studies conducted to date, but wind development along the coasts (where shorebirds might be at greater risk) did not begin until 2005.

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 13) 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, as buildout of offshore, coastal, and inland wind energy infrastructure progresses, increasing mortality from turbine collisions may contribute to a subspecies-level effect due to the red knot's vulnerability to direct human-caused mortality. 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 Reduced Food Availability, above, and supplemental document—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. Under the current management framework (the ARM), the present horseshoe crab harvest is not considered a threat to the red knot, but it is not yet known if the horseshoe crab egg resource will continue to adequately support red knot populations over the next 5 to 10 years. 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, warming water and air temperatures, ocean acidification, physical habitat changes, possibly increased prevalence of disease and parasites, marine invasive species, and burial and crushing of invertebrate prey from sand placement and recreational activities.

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. 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. The factors that caused delays in the spring migrations of red knots from Argentina and Chile are still unknown, and we have no information to indicate if this delay will reverse, persist, or intensify. Superimposed on the existing threat of late arrivals in Delaware Bay are new threats emerging due to climate change, such as changes in the timing of reproduction for both horseshoe crabs and mollusks. 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. The red knot's adaptive capacity to deal with numerous changes in the timing of resource availability across its geographic range is largely unknown. A few examples suggest some flexibility in red knot migration strategies, but differences between the annual timing cues of red knots (at least partly celestial and endogenous) and their prey (primarily environmental) suggest there are limitations on the adaptive capacity of red knots to cope with increasing frequency or severity of asynchronies.

Other threats are likely to exacerbate the effects of reduced prey availability and asynchronies, including human disturbance, competition with gulls, and behavioral changes from wind energy development. Additional threats are likely to increase the levels of direct red knot mortality, such as HABs, oil spills and other contaminants, and collisions with wind turbines. In addition to elevating background mortality rates, these three threats 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 scientific and commercial data available, 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. 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 several hundred to a thousand birds now represent a greater proportion of the total red knot population than in the past. Natural or anthropogenic stochastic events affecting these flocks 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, 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.

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).

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 red knots 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 this finding suggests 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, but the impacts are potentially profound (Fraser *et al.* 2013, entire; Schmidt *et al.* 2012, p. 4421; 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). 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.

Determination

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 (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. Listing actions may be warranted based on any of the above threat factors, singly or in combination.

We have carefully assessed the best scientific and commercial data available regarding the past, present, and future threats to the rufa red knot. We have identified threats to the red knot attributable to Factors A, B, C, and E. The primary driving threats to the red knot are from habitat loss and degradation due to sea level rise, shoreline stabilization, and Arctic warming (Factor A), and reduced food availability and asynchronies in the annual cycle (Factor E). Other threats are moderate in comparison to the primary threats; however, cumulatively, they could become significant when working in concert with the primary threats if they further reduce the species' resiliency. These secondary threats include hunting (Factor B); predation (Factor C); and human disturbance, harmful algal blooms, oil spills, and wind energy development (Factor E). All of these factors affect red knots across their current range.

Conservation efforts are being implemented in many areas of the red knot's range (see Factors A, B, C, and E). For example, in 2012, the ASMFC adopted the ARM for the management of the horseshoe crab population in the Delaware Bay Region to meet the dual objectives of maximizing crab harvest and meeting red knot population targets (ASMFC 2012e, p. 1). In addition, regulatory mechanisms exist that provide protections for the red knot directly (e.g., MBTA protections against take for scientific study or by hunting) or through regulation of activities that threaten red knot habitat (e.g., section 404 of the Clean Water Act, Rivers and Harbors Act, Coastal Barrier Resources Act, and Coastal Zone Management Act, and State regulation of shoreline stabilization and coastal development) (see supplemental document—Factor D). While these conservation efforts and existing regulatory mechanisms reduce some threats to the red knot, significant risks to the subspecies remain.

Red knots migrate annually between their breeding grounds in the Canadian Arctic and several wintering regions, including the Southeast United States, the Northeast Gulf of Mexico, northern Brazil, and Tierra del Fuego at the southern tip of South America. During both the spring and fall migrations, red knots use key staging and stopover areas to rest and feed. This life-history strategy makes this species inherently vulnerable to numerous changes in the timing of quality food and habitat resource availability across its geographic range. While a few examples suggest the species has some flexibility in migration strategies, the full scope of

the species' adaptability to changes in its annual cycle is unknown.

The Act defines an endangered species as any species that is ''in danger of extinction throughout all or a significant portion of its range'' and a threatened species as any species ''that is likely to become endangered throughout all or a significant portion of its range within the foreseeable future.'' We find that the rufa red knot meets the definition of a threatened species due to the likelihood of habitat loss driven by climate change and human response to climate change and reduced food resources and further asynchronies in its annual cycle that result in the species' reduced redundancy, resiliency, and representation. While there is uncertainty as to how long it may take some of the climate-induced changes to manifest in population-level effects to the rufa red knot, we find that the best available data suggests the rufa red knot is not at a high risk of a significant decline in the near term. However, should the reduction in redundancy, resiliency, and representation culminate in an abrupt and large loss, or initiation of a steep rate of decline, of reproductive capability or we subsequently find that the species does not have the adaptive capacity to adjust to actual shifts in its food and habitat resources, then the red knot would be at higher risk of a significant decline in the near term, and thus would meet the definition of an endangered species under the Act. We base this determination on the immediacy, severity, and scope of the threats described above. Therefore, on the basis of the best available scientific and commercial data, we propose listing the rufa red knot as a threatened species in accordance with sections 3(6) and 4(a)(1) of the Act.

Under the Act and our implementing regulations, a species may warrant listing if it meets the definition of an endangered or threatened species throughout all or a significant portion of its range. The rufa red knot proposed for listing in this rule is wide-ranging and the threats occur throughout its range. Therefore, we assessed the status of the subspecies throughout its entire range. The threats to the survival of the subspecies are not restricted to any particular significant portion of that range. Accordingly, our assessment and proposed determination applies to the subspecies throughout its entire range.

Available Conservation Measures

Conservation measures provided to species listed as endangered or threatened under the Act include recognition, recovery actions, requirements for Federal protection, and prohibitions against certain practices. Recognition through listing results in public awareness and conservation by Federal, State, Tribal, and local agencies, private organizations, and individuals. The Act encourages cooperation with the States and requires that recovery actions be carried out for all listed species. The protection required by Federal agencies and the prohibitions against certain activities are discussed, in part, below.

The primary purpose of the Act is the conservation of endangered and threatened species and the ecosystems upon which they depend. The ultimate goal of such conservation efforts is the recovery of these listed species, so that they no longer need the protective measures of the Act. Subsection 4(f) of the Act requires the Service to develop and implement recovery plans for the conservation of endangered and threatened species. The recovery planning process involves the identification of actions that are necessary to halt or reverse the species' decline by addressing the threats to its survival and recovery. The goal of this process is to restore listed species to a point where they are secure, self-sustaining, and functioning components of their ecosystems.

Recovery planning includes the development of a recovery outline shortly after a species is listed and preparation of a draft and final recovery plan. The recovery outline guides the immediate implementation of urgent recovery actions and describes the process to be used to develop a recovery plan. Revisions of the plan may be done to address continuing or new threats to the species, as new substantive information becomes available. The recovery plan identifies site-specific management actions that set a trigger for review of the five factors that control whether a species remains endangered or may be downlisted or delisted, and methods for monitoring recovery progress. Recovery plans also establish a framework for agencies to coordinate their recovery efforts and provide estimates of the cost of implementing recovery tasks. Recovery teams (composed of species experts, Federal and State agencies, nongovernmental organizations, and stakeholders) are often established to develop recovery plans. When completed, the recovery outline, draft recovery plan, and final recovery plan will be available on our Web site (*http://www.fws.gov/endangered*), or from our New Jersey Fish and Wildlife Office (see **FOR FURTHER INFORMATION CONTACT**).

Implementation of recovery actions generally requires the participation of a broad range of partners, including other Federal agencies, States, Tribes, nongovernmental organizations, businesses, and private landowners. Examples of recovery actions include habitat restoration (e.g., restoration of native vegetation), research, captive propagation and reintroduction, and outreach and education. The recovery of many listed species cannot be accomplished solely on Federal lands because their ranges may occur primarily or solely on non-Federal lands. Recovery of these species requires cooperative conservation efforts on private, State, and Tribal lands.

If this species is listed, funding for recovery actions will be available from a variety of sources, including Federal budgets, State programs, and cost-share grants for non-Federal landowners, the academic community, and nongovernmental organizations. In addition, pursuant to section 6 of the Act, States regularly inhabited by rufa red knots during the wintering or stopover periods would be eligible for Federal funds to implement management actions that promote the protection or recovery of the rufa red knot. Information on our grant programs that are available to aid species recovery can be found at: *http://www.fws.gov/grants*.

Although the rufa red knot is only proposed for listing under the Act at this time, please let us know if you are interested in participating in recovery efforts for this species. Additionally, we invite you to submit any new information on this species whenever it becomes available and any information you may have for recovery planning purposes (see **FOR FURTHER INFORMATION CONTACT**).

Section 7(a) of the Act requires Federal agencies to evaluate their actions with respect to any species that is proposed or listed as an endangered or threatened species and with respect to its critical habitat, if any is designated. Regulations implementing this interagency cooperation provision of the Act are codified at 50 CFR part 402. Section 7(a)(4) of the Act requires Federal agencies to confer with the Service on any action that is likely to jeopardize the continued existence of a species proposed for listing or result in destruction or adverse modification of proposed critical habitat. If a species is listed subsequently, section 7(a)(2) of the Act requires Federal agencies to ensure that activities they authorize, fund, or carry out are not likely to jeopardize the continued existence of the species or destroy or adversely

modify its critical habitat. If a Federal action may affect a listed species or its critical habitat, the responsible Federal agency must enter into formal consultation with the Service.

Federal agency actions within the species habitat that may require conference or consultation or both as described in the preceding paragraph include management and landscape altering activities on Federal lands administered by the Department of Defense, the Service, and NPS; issuance of section 404 Clean Water Act permits and shoreline stabilization projects implemented by the USACE; construction and management of gas pipeline rights-of-way by the Federal Energy Regulatory Commission; leasing of Federal waters by the BOEM for the construction of wind turbines; and construction and maintenance of roads or highways by the Federal Highway Administration.

The Act and its implementing regulations set forth a series of general prohibitions and exceptions that apply to all endangered wildlife. The prohibitions of section 9(a)(2) of the Act, codified at 50 CFR 17.21 for endangered wildlife, in part, make it illegal for any person subject to the jurisdiction of the United States to take (includes harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect; or to attempt any of these), import, export, ship in interstate commerce in the course of commercial activity, or sell or offer for sale in interstate or foreign commerce any listed species. Under the Lacey Act (18 U.S.C. 42–43; 16 U.S.C. 3371–3378), it is also illegal to possess, sell, deliver, carry, transport, or ship any such wildlife that has been taken illegally. Certain exceptions apply to agents of the Service and State conservation agencies.

We may issue permits to carry out otherwise prohibited activities involving endangered and threatened wildlife species under certain circumstances. Regulations governing permits are codified at 50 CFR 17.22 for endangered species, and at 17.32 for threatened species. With regard to endangered wildlife, a permit must be issued for the following purposes: For scientific purposes, to enhance the propagation or survival of the species, and for incidental take in connection with otherwise lawful activities.

Our policy, as published in the **Federal Register** on July 1, 1994 (59 FR 34272), is to identify to the maximum extent practicable at the time a species is listed, those activities that would or would not constitute a violation of section 9 of the Act. The intent of this policy is to increase public awareness of the potential effect of a listing on proposed and ongoing activities within the range of species proposed for listing. The following activities could potentially result in a violation of section 9 of the Act; this list is not comprehensive:

(1) Unauthorized collecting, handling, possessing, selling, delivering, carrying, or transporting of the species, including import or export across State lines and international boundaries, except for properly documented antique specimens of these taxa at least 100 years old, as defined by section 10(h)(1) of the Act;

(2) Introduction of nonnative species that compete with or prey upon the rufa red knot, or that cause declines of the red knot's prey species;

(3) Unauthorized modification of intertidal habitat that regularly support concentrations of rufa red knots during the wintering or stopover periods; and

(4) Unauthorized discharge of chemicals or fill material into any waters along which the rufa red knot is known to occur.

(1) The following activities are not likely to result in a violation of section 9 of the Act; this list is not comprehensive: Harvest of horseshoe crabs in accordance with the ARM, provided the ARM is implemented as intended (e.g., including implementation of necessary monitoring programs), and enforced.

Questions regarding whether specific activities would constitute a violation of section 9 of the Act should be directed to the New Jersey Fish and Wildlife Office (see **FOR FURTHER INFORMATION CONTACT**). Requests for copies of the regulations concerning listed animals and general inquiries regarding prohibitions and permits may be addressed to the U.S. Fish and Wildlife Service, Endangered Species Permits, 300 Westgate Center Drive, Hadley, MA, 01035 (telephone 413–253–8615; facsimile 413–253–8482).

Under section 4(d) of the Act, the Secretary has discretion to issue such regulations as he deems necessary and advisable to provide for the conservation of threatened species. Our implementing regulations (50 CFR 17.31) for threatened wildlife generally incorporate the prohibitions of section 9 of the Act for endangered wildlife, except when a ''special rule'' promulgated pursuant to section 4(d) of the Act has been issued with respect to a particular threatened species. In such a case, the general prohibitions in 50 CFR 17.31 would not apply to that species, and instead, the special rule would define the specific take prohibitions and exceptions that would apply for that particular threatened species, which we consider necessary and advisable to conserve the species. The Secretary also has the discretion to prohibit by regulation with respect to a threatened species any act prohibited by section 9(a)(1) of the Act. Exercising this discretion, which has been delegated to the Service by the Secretary, the Service has developed general prohibitions that are appropriate for most threatened species in 50 CFR 17.31 and exceptions to those prohibitions in 50 CFR 17.32. We are not proposing to promulgate a special section 4(d) rule, and as a result, all of the section 9 prohibitions, including the ''take'' prohibitions, will apply to the rufa red knot. (As described above, harvest of horseshoe crabs in accordance with the ARM is not likely to result in take under section 9 of the Act.)

Listing the rufa red knot under the Act would invoke provisions under various State laws that would prohibit take and encourage conservation by State government agencies. Further, States may enter into agreements with Federal agencies to administer and manage areas required for the conservation, management, enhancement, or protection of endangered species. Funds for these activities could be made available under section 6 of the Act (Cooperation with the States). Thus, the Federal protection afforded to these species by listing them as endangered species will be reinforced and supplemented by protection under State law.

A determination to list the rufa red knot as a threatened species under the Act, if we ultimately determine that listing is warranted, will not regulate greenhouse gas emissions. Rather, it will reflect a determination that the rufa red knot meets the definition of a threatened species under the Act, thereby establishing certain protections for it under the Act. While we acknowledge that listing will not have a direct impact on those aspects of climate change impacting the rufa red knot (e.g., sea level rise, ocean acidification, warming coastal waters, changing patterns of coastal storm activity, warming of the Arctic), we expect that listing will indirectly enhance national and international cooperation and coordination of conservation efforts, enhance research programs, and encourage the development of mitigation measures that could help slow habitat loss and population declines. In addition, the development of a recovery plan will guide efforts intended to ensure the long-term survival and eventual recovery of the rufa red knot.

Required Determinations

Clarity of the Rule

We are required by Executive Orders 12866 and 12988 and by the Presidential Memorandum of June 1, 1998, to write all rules in plain language. This means that each rule we publish must:

(1) Be logically organized;

(2) Use the active voice to address readers directly;

(3) Use clear language rather than jargon;

(4) Be divided into short sections and sentences; and

(5) Use lists and tables wherever possible.

If you feel that we have not met these requirements, send us comments by one of the methods listed in the **ADDRESSES** section. To better help us revise the rule, your comments should be as specific as possible. For example, you should tell us the numbers of the sections or paragraphs that are unclearly written, which sections or sentences are too long, the sections where you feel lists or tables would be useful, etc.

National Environmental Policy Act (42 U.S.C. 4321 et seq.)

We have determined that environmental assessments and environmental impact statements, as defined under the authority of the National Environmental Policy Act of 1969, need not be prepared in connection with listing a species as an endangered or threatened species under the Endangered Species Act. We published a notice outlining our reasons for this determination in the **Federal Register** on October 25, 1983 (48 FR 49244).

References Cited

A complete list of all references cited in this proposed rule is available on the Internet at *http://www.regulations.gov* or upon request from the Field Supervisor, New Jersey Field Office (see **FOR FURTHER INFORMATION CONTACT** section).

Authors

The primary authors of this proposed rule are the staff members of the New Jersey Field Office (see **FOR FURTHER INFORMATION CONTACT**).

List of Subjects in 50 CFR Part 17

Endangered and threatened species, Exports, Imports, Reporting and recordkeeping requirements, and Transportation.

Proposed Regulation Promulgation

Accordingly, we propose to amend part 17, subchapter B of chapter I, title 50 of the Code of Federal Regulations, as set forth below:

PART 17—[AMENDED]

■ 1. The authority citation for part 17 continues to read as follows:

Authority: 16 U.S.C. 1361–1407; 1531–1544; 4201–4245; unless otherwise noted.

■ 2. In § 17.11(h) add an entry for ''Knot, rufa red'' to the List of Endangered and Threatened Wildlife in alphabetical order under Birds to read as set forth below:

§ 17.11 Endangered and threatened wildlife.

* * * * *

(h) * * *

Species		Historic range	Vertebrate population where endangered or threatened	Status	When listed	Critical habitat	Special rules
Common name	Scientific name						
*	*	*	*	*	*		*
BIRDS							
*	*	*	*	*	*		*
Knot, rufa red ...	*Calidris canutus ssp. rufa.*	Argentina, Aruba, Bahamas, Barbados, Belize, Brazil, British Virgin Islands, Canada, Cayman Islands, Chile, Colombia, Costa Rica, Cuba, Dominican Republic, El Salvador, France (Guadeloupe, French Guiana), Guatemala, Guyana, Haiti, Jamaica, Mexico, Panama, Paraguay, Suriname, Trinidad and Tobago, Uruguay, Venezuela, U.S.A. (AL, AR, CT, CO, DE, FL, GA, IA, IL, IN, KS, KY, LA, MA, MD, ME, MI, MN, MO, MS, MT, NE, NC, ND, NH, NJ, NY, OH, OK, PA, RI, SC, SD, TN, TX, VA, VT, WI, WV, WY, Puerto Rico, U.S. Virgin Islands).	Entire	T	.....	N/A	N/A
*	*	*	*	*	*		*

Dated: September 6, 2013.

Rowan W. Gould,

Acting Director, U.S. Fish and Wildlife Service.

[FR Doc. 2013–22700 Filed 9–27–13; 8:45 am]

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