Nancy Hilding
Mixed Species

1. USFWS Fed. Reg. on Northern Long Eared Bat Oct 2013
2. SD GFP Report on Black Hills Bats 2003 Tigner & Stukel (see page 27)
3. USFWS Fed. Reg. on Red knot listing Sept 2013
4. USFWS Status Review Sturgeon Chub 2001
5. A USFWS Report on Selenium Contamination from In-situ leach waste water Sept. 2000
6. USFWS map Sturgeon chub Range 2001

Northern
Long eared bat

NATIONAL ARCHIVES AND RECORDS ADMINISTRATION
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Department of the Interior

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50 CFR Part 17

Endangered and Threatened Wildlife and Plants; 12-Month Finding on a Petition To List the Eastern Small-Footed Bat and the Northern Long-Eared Bat as Endangered or Threatened Species; Listing the Northern Long-Eared Bat as an Endangered Species; Proposed Rule

DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

50 CFR Part 17

[Docket No. FWS–R5–ES–2011–0024; 4500030113]

RIN 1018–AY98

Endangered and Threatened Wildlife and Plants; 12-Month Finding on a Petition To List the Eastern Small-Footed Bat and the Northern Long-Eared Bat as Endangered or Threatened Species; Listing the Northern Long-Eared Bat as an Endangered Species

AGENCY: Fish and Wildlife Service, Interior.

ACTION: Proposed rule; 12-month finding.

SUMMARY: We, the U.S. Fish and Wildlife Service (Service), announce a 12-month finding on a petition to list the eastern small-footed bat (*Myotis leibii*) and the northern long-eared bat (*Myotis septentrionalis*) as endangered or threatened under the Endangered Species Act of 1973, as amended (Act) and to designate critical habitat. After review of the best available scientific and commercial information, we find that listing the eastern small-footed bat is not warranted but listing the northern long-eared bat is warranted. Accordingly, we propose to list the northern long-eared bat as an endangered species throughout its range under the Act. We also determine that critical habitat for the northern long-eared bat is not determinable at this time. This proposed rule, if finalized, would extend the Act's protections to the northern long-eared bat. The Service seeks data and comments from the public on this proposed listing rule for the northern long-eared bat.

DATES: We will consider comments received or postmarked on or before December 2, 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 a public hearing, in writing, at the address shown in the **FOR FURTHER INFORMATION CONTACT** section by November 18, 2013.

ADDRESSES: You may submit comments by one of the following methods:

(1) In the Search box, enter Docket No. FWS–R5–ES–2011–0024, which is the docket number for this rulemaking. Then, in the Search panel on the left side of the screen, under the Document Type heading, click on the Proposed Rules link to locate this document. You may submit a comment by clicking on "Comment Now!" If your comments will fit in the provided comment box, please use this feature of *http://www.regulations.gov,* as it is most compatible with our comment review procedures. If you attach your comments as a separate document, our preferred file format is Microsoft Word. If you attach multiple comments (such as form letters), our preferred format is a spreadsheet in Microsoft Excel.

(2) *By hard copy:* Submit by U.S. mail or hand-delivery to: Public Comments Processing, Attn: FWS–R5–ES–2011–0024; Division of Policy and Directives Management; U.S. Fish and Wildlife Service; 4401 N. Fairfax Drive, MS 2042–PDM; Arlington, VA 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 Information Requested section below for more details).

FOR FURTHER INFORMATION CONTACT: Peter Fasbender, Field Supervisor, U.S. Fish and Wildlife Service, Green Bay Ecological Services Office, 2661 Scott Tower Dr., New Franken, Wisconsin, 54229; by telephone (920) 866–3650 or by facsimile (920) 866–1710. *mailto:* If you use a telecommunications device for the deaf (TDD), please 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 an endangered or threatened species 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 one year. Listing a species as an endangered or threatened species can only be completed by issuing a rule.

This document consists of:

- Our status review and finding that listing is warranted for the northern long-eared bat and not warranted for the eastern small-footed bat.
- A proposed rule to list the northern long-eared bat as an endangered species. This rule assesses best available information regarding the status of and threats to the northern long-eared bat.

The basis for our action. Under the Act, we can 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 northern long-eared bat is in danger of extinction, predominantly due to the threat of white-nose syndrome (Factor C). However, other threats (Factors A, B, E) when combined with white-nose syndrome heighten the level of risk to the species.

We will seek peer review. We are seeking comments from knowledgeable individuals with scientific expertise to review our analysis of the best available science and application of that science and to provide any additional scientific information to improve this proposed rule. Because we will consider all comments and information we receive during the comment period, our final determination may differ from this proposal.

Information Requested

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 other concerned Federal and State agencies, the scientific community, or any other interested party concerning this proposed rule. We particularly seek comments regarding the northern long-eared bat concerning:

(1) The species' biology, range, and population trends, including:

(a) 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) Any information on the biological or ecological requirements of the species, and ongoing conservation measures for the species and its habitat.

(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) Current or planned activities in the areas occupied by the species and possible impacts of these activities on this species.

(5) Additional information regarding the threats to the species under the five listing factors, which are:

(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.

(6) The reasons why areas should or should not be designated as critical habitat as provided by section 4 of the Act (16 U.S.C. 1531 *et seq.*), including the possible risks or benefits of designating critical habitat, including risks associated with publication of maps designating any area on which this species may be located, now or in the future, as critical habitat.

(7) The following specific information on:

(a) The amount and distribution of habitat for northern long-eared bat;

(b) What areas, that are currently occupied and that contain the physical and biological features essential to the conservation of this species, should be included in a critical habitat designation and why;

(c) Special management considerations or protection that may be needed for the essential features in potential critical habitat areas, including managing for the potential effects of climate change;

(d) What areas not occupied at the time of listing are essential for the conservation of this species and why;

(e) The amount of forest removal occurring within known summer habitat for this species;

(f) Information on summer roost habitat requirements that are essential for the conservation of the species and why; and

(g) Information on species winter habitat (hibernacula) features and requirements for the species.

(8) Information on the projected and reasonably likely impacts of changing environmental conditions resulting from climate change on the species and its habitat.

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 **ADDRESSES**. 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, Green Bay, Wisconsin Field Office (see **FOR FURTHER INFORMATION CONTACT**).

Background

Section 4(b)(3)(B) of the Act requires that, for any petition to revise the Federal Lists of Threatened and Endangered Wildlife and Plants that contains substantial scientific or commercial information that listing a species may be warranted, we make a finding within 12 months of the date of receipt of the petition on whether the petitioned action is: (a) Not warranted; (b) warranted; or (3) warranted, but the immediate proposal of a regulation implementing the petitioned action is precluded by other pending proposals to determine whether any species is endangered or threatened, and expeditious progress is being made to add or remove qualified species from the Federal Lists of Endangered and Threatened Wildlife and Plants. In this document, we have determined that the petitioned action to list the eastern small-footed bat is not warranted, but listing the northern long-eared bat is warranted and; therefore, we are publishing a proposed rule to list the northern long-eared bat.

Previous Federal Actions

On September 18, 1985 (50 FR 37958), November 21, 1991 (56 FR 58804), and November 15, 1994 (59 FR 58982), the Service issued notices of review identifying the eastern small-footed bat as a "category-2 candidate" for listing under the Act. However, on December 5, 1996 (50 FR 64481), the Service discontinued the practice of maintaining a list of species regarded as "category-2 candidates," that is, taxa for which the Service had insufficient information to support issuance of a proposed listing rule.

On January 21, 2010, we received a petition from the Center for Biological Diversity, requesting that the eastern small-footed bat and northern long-eared bat be listed as endangered or threatened and that critical habitat be designated under the Act. The petition clearly identified itself as such and included the requisite identification information for the petitioner, as required by 50 CFR 424.14(a). In a February 19, 2010, letter to the petitioner, we acknowledged receipt of the petition and stated that we would review the petitioned request for listing and inform the petitioner of our determination upon completion of our review. On June 23, 2010, we received a notice of intent to sue (NOI) from the petitioner for failing to make a timely 90-day finding. In a letter dated July 20, 2010, we responded to the NOI, stating that we had assigned lead for the two bat species to the Services' Midwest and Northeast Regions, and that although completing the 90-day finding within the 90 days following our receipt of the petition was not practicable, the Regions were recently allocated funding to work on the findings and had begun review of the petition. On June 29, 2011, we published in the **Federal Register** (76 FR 38095) our finding that the petition to list the eastern small-footed bat and northern long-eared bat presented substantial information indicating that the requested action may be warranted, and we initiated a status review of the species. On July 12, 2011, the Service filed a proposed settlement agreement with the Center for Biological Diversity in a consolidated case in the U.S. District Court for the District of Columbia. The settlement agreement was approved by the court on September 9, 2011. As part of this settlement agreement, the Service agreed to complete a status review for the eastern small-footed bat and northern long-eared bat by September 30, 2013, and if warranted for listing, publish a proposed listing rule also by that date.

Species Information

Eastern Small-Footed Bat

Taxonomy and Species Description

The eastern small-footed bat (*Myotis leibii*) belongs to the Order Chiroptera, Suborder Microchiroptera, and Family Vespertilionidae (Best and Jennings 1997, p. 1). The eastern small-footed bat is considered monotypic, whereby no subspecies has been recognized (van Zyll de Jong 1984, p. 2525). This species has been identified by different scientific names: *Vespertilio leibii* (Audubon and Bachman 1842, p. 284) and *Myotis subulatus* (Miller and Allen 1928, p. 164). This species also has been identified by different common names: Leib's bat (Audubon and Bachman 1842, p. 284), least brown bat (Mohr 1936, p. 62), and Leib's masked bat or least bat (Hitchcock 1949, p. 47). The Service agrees with the treatment in Best and Jennings (1997, p. 1) regarding the scientific and common names and will refer to this species as eastern small-footed bat and recognizes it as a listable entity under the Act.

The eastern small-footed bat is one of the smallest North American bats, weighing from 3 to 8 grams (g) (0.1 to 0.3 ounces (oz)) (Merritt 1987, p. 94). Total body length is from 73 to 85 millimeters (mm) (2.9 to 3.4 inches (in)), tail length is from 31 to 34 mm (1.2 to 1.3 in), forearm length is from 30 to 36 mm (1.2 to 1.4 in), and wingspan is from 212 to 248 mm (8.4 to 9.8 in) (Barbour and Davis 1969, p. 103; Merritt 1987, p. 94; Erdle and Hobson 2001, p. 6; Amelon and Burhans 2006, p. 57). Eastern small-footed bats are recognized by their short hind feet (less than 8 mm (0.3 in)), short ears (less than 15 mm (0.6 in)), black facial mask, black ears, keeled calcar (a spur of cartilage that helps spread the wing membrane), and small flattened skull (Barbour and Davis 1969, p. 103; Best and Jennings 1997, p. 1). The wings and interfemoral membrane (the wing membrane between the tail and hind legs) are black. The dorsal fur is black at the roots and tipped with light brown, giving it a dark yellowish-brown appearance. The ventral fur is gray at the roots and tipped with yellowish-white (Audubon and Bachman 1842, pp. 284–285).

Distribution and Abundance

The eastern small-footed bat occurs from eastern Canada and New England south to Alabama and Georgia and west to Oklahoma. The species' range includes 26 states and 2 Canadian provinces, including Alabama, Arkansas, Connecticut, Delaware, Georgia, Illinois, Indiana, Kentucky, Maine, Maryland, Massachusetts, Mississippi, Missouri, New Hampshire, New Jersey, New York, North Carolina, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, Tennessee, Vermont, Virginia, West Virginia, Ontario, and Quebec. Relative to other species of bats in its range, eastern small-footed bats are considered uncommon (Best and Jennings 1997, p. 3). They historically have been considered rare because of their patchy distribution and generally low population numbers (Mohr 1932, p. 160). In areas with abundant summer habitat, however, they have been found to be relatively common (Brack *et al.*, unpublished manuscript). Johnson *et al.* (2011, p. 99) observed that capture success decreased as the distance increased from suitable roosting habitat. Eastern small-footed bats have also been noted for their ability to detect and avoid mist nets, which are typically relied upon for summer bat surveys (Barbour and Davis 1974, p. 84), suggesting their numbers could be underrepresented (Tyburec 2012).

Eastern small-footed bats have most often been detected during winter hibernacula (the areas where the bats hibernate during winter; primarily caves and mines) surveys (Barbour and Davis 1969, p. 103). Two-hundred eighty-nine hibernacula (includes cave and abandoned mine features only) have been identified across the species' range, though most contain just a few individuals. The majority of known hibernacula occur in Pennsylvania (n=55), New York (n=53), West Virginia (n=50), Virginia (n=33), Kentucky (n=26), and North Carolina (n=25), but hibernacula are also known from Tennessee (approximately 12), Arkansas (n=9), Maryland (n=7), Vermont (n=6), Missouri (n=3), Maine (n=2), Massachusetts (n=2), New Hampshire (n=2), New Jersey (n=2), Indiana (n=1), and Oklahoma (n=1). In Vermont, eastern small-footed bats were consistently found in very small numbers and often not detected at all during periodic surveys of hibernacula (Trombulak *et al.* 2001, pp. 53–57). Their propensity for hibernating in cracks and crevices in cave and mine floors and ceilings may also mean they are more often overlooked than other cave-hibernating bat species. The largest number of hibernating individuals ever reported for the species was 2,383, which were found in a mine in Essex County, New York (Herzog 2013, pers. comm.).

In Pennsylvania, eastern small-footed bats were observed at 55 of 480 (12 percent) hibernacula from 1984 to 2011, accounting for only 0.1 percent of the total bats observed during winter hibernacula surveys. The number of eastern small-footed bats observed per site fluctuates annually and ranges from 1 to 46 (mean = 4, median = 1). Summer mist-net surveys also confirm that eastern small-footed bats are observed less frequently than other bat species. From 1995 to 2011, of the 7,007 bat mist-net surveys conducted in Pennsylvania, only 104 surveys (2 percent) include eastern small-footed bat captures, representing only 0.3 percent of the total bats captured (Butchkoski 2011, unpublished data). Of the other states within the species' range, seven states (Alabama, Connecticut, Delaware, Indiana, Massachusetts, Mississippi, and Rhode Island) have no summer records, and of those States with summer records, the most have fewer than 20 capture locations (Service, unpublished data).

Illustrating the potential for under-representation of the species during hibernacula surveys, the following is an example from one state. From 1939 to 1944, over 100 caves were surveyed in Pennsylvania (and a portion of West Virginia), and out of these, eastern small-footed bats were observed at only 7 sites, totaling 363 individuals. In 1978 and 1979, the same seven caves were surveyed again, and no eastern small-footed bats were observed (Felbaum *et al.* 1995, p. 24). However, surveys conducted from 1980 to 1988, found eastern small-footed bats inhabiting 21 hibernacula from an 8-county area in Pennsylvania (Dunn and Hall 1989, p. 169), and by 2011, surveys had confirmed presence at 55 sites in a 14-county area (Pennsylvania Game Commission, unpublished data). This example is typical of the species' potential for fluctuation throughout its range.

Habitat

Winter Habitat

Eastern small-footed bats have been observed most often overwintering in hibernacula that include caves and abandoned mines (*e.g.*, limestone, coal, iron). Because they tolerate colder temperatures more so than other *Myotis* bats, they are most often encountered close to cave or mine entrances where humidity is low and temperature fluctuations may be high relative to more interior areas (Hitchcock 1949, p. 53; Barbour and Davis 1969, p. 104; Best and Jennings 1997, pp. 2–3; Veilleux 2007, p. 502). On occasion, however, they have been observed hibernating deep within cave interiors (Hitchcock 1965, p. 9; Gunier and Elder 1973, p. 490). In Pennsylvania, caves containing wintering populations of eastern small-footed bats have been found in hemlock-dominated forests in the foothills of mountains that rise to 610 meters (m) (2000 feet (ft)) (Mohr 1936, p. 63). Dunn and Hall (1989, p. 169) noted that 52 percent of Pennsylvania hibernacula used by eastern small-footed bats were small caves of less than 150 m (500 ft) in length. Before it was commercialized, the cave in Fourth Chute, Ontario was home to a relatively large number of hibernating eastern small-footed bats (n = 434) and is described in Hitchcock (1949, pp. 47–54) as follows: "the cave is in a limestone outcropping on the north bank of the Bonnechere River, at an elevation of 425 ft (130 m). Sinkholes and large openings to passages make this cave conspicuous. Most of the land immediately surrounding the cave area is open field or pasture, with wooded hills beyond. The part utilized by bats for hibernation lies farthest from the river, and is entered from one of the large, outside passageways through a narrow opening; the main passages are well ventilated by a through draft; the forests near Fourth Chute are mixed, with spruce and white cedar predominating among the conifers." Eastern small-footed bats were found in cold, dry, drafty locations at Fourth Chute, usually in narrow cracks in the cave wall or roof (Hitchcock 1949, p. 53).

Winter habitat used by eastern small-footed bats may also include non-cave or non-mine features, such as rock outcrops and stone highway culverts. In Pennsylvania, eastern small-footed bats were observed hibernating multiple years during the months of January and March in a rock outcrop located high above the Juniata River. The bats were found in small cracks and crevices at the back of a 4.6-m (15-ft) depression in the rock outcrop. Big brown bats (*Eptesicus fuscus*) were also present. Temperatures within the cracks where bats were hibernating ranged from 1.7 to 8.3 °C (35 to 47 °F). Observers noted that it seemed a cold, unstable site for hibernating bats (Pennsylvania Game Commission, unpublished data). In West Virginia, an eastern small-footed bat was observed in a crack in a rock outcrop about 1.5 to 1.8 m (5 to 6 ft) above the ground in February (Stihler 2012, pers. comm.). Sasse *et al.* (in press) reported a single female eastern small-footed bat hibernating inside a stone highway culvert underneath a highway in Arkansas. Mohr (1936, p. 64) noted fluctuations in the number of eastern small-footed bats observed at hibernacula during winter surveys conducted 2 to 3 weeks apart, suggesting bats left caves and mines during warmer winter periods only to return when it became colder. Consequently, eastern small-footed bats may be utilizing non-cave or non-mine rock features during mild or milder portions of winters, but to what extent they may be doing so is largely unknown.

Summer Habitat

In the summer, eastern small-footed bats are dependent on emergent rock habitats for roosting and on the immediately surrounding forests for foraging (Johnson *et al.* 2009, p. 5). Eastern small-footed bats have been observed roosting singly or in small maternity colonies in talus fields and slopes, rock-outcrops, rocky ridges, sandstone boulders, shale rock piles, limestone spoil piles, rocky terrain of strip mine areas, and cliff crevices, but have also been found on humanmade structures such as buildings and expansion joints of bridges (Barbour and Davis 1969, p. 103; McDaniel *et al.* 1982, p. 93; Merritt 1987, p. 95; MacGregor and Kiser 1998, p. 175; Roble 2004, p. 43; Amelon and Burhans 2006, p. 58; Chenger 2008a, p. 10; Chenger 2008b, p. 6; Johnson *et al.* 2011, p. 100; Johnson and Gates 2008, p. 456; Hauser and Chenger 2010; Sanders 2010; Mumma and Capouillez 2011, p. 24; Thomson and O'Keefe 2011; Brack *et al.*, unpublished manuscript). Other humanmade features exploited by eastern small-footed bats include rocky dams, road cuts, rocky mine lands, mines, and rock fields within transmission-line and pipeline clearings (Sanders 2011, pers. comm.; Johnson *et al.* 2011, p. 99; Thomson and O'Keefe 2011). Roost sites are most often located in areas with full solar exposure, but have also been found in areas with moderate to extensive canopy cover (Johnson *et al.* 2011, p. 100; Brack *et al.* unpublished manuscript, pp. 9–15; Thomson and O'Keefe 2012). In New Hampshire, eastern small-footed bats have been observed roosting between boulder crevices along the southern outflow of the Surry Mountain Reservoir (Veilleux and Reynolds 2006, p. 330). In Vermont, one summer colony, containing approximately 30 eastern small-footed bats, was located in a slate roof of a house (Darling and Smith 2011, p. 4). Tuttle (1964, p. 149) reported two individuals found in April in Tennessee under a large flat rock at the edge of a quarry surrounded by woods and cow pastures (elevation 549 m (1,800 ft)). In Ontario, a colony of approximately 12 bats was found in July behind a shed door (Hitchcock 1955, p. 31). In addition, small numbers of adult and juvenile eastern small-footed bats have been observed using caves and mines as roosting habitat during the summer months in Maryland, Pennsylvania, Kentucky, Arkansas, West Virginia, and Virginia (Davis *et al.* 1965, p. 683; Krutzsch 1966, p. 121; Hall and Brenner 1968, p. 779; McDaniel *et al.* 1982, p. 93; Agosta *et al.* 2005, p. 1213; Reynolds, pers. comm.).

Summer foraging habitat used by eastern small-footed bats includes rivers, streams, riparian forests, upland forests, clearings, strip mines, and ridgetops (Chenger 2003, pp. 14–23; Chenger 2008a, pp. 10 and 69–71; Chenger 2008b, p. 6; Hauser and Chenger 2010; Johnson *et al.* 2009, p. 3; Mumma and Capouillez 2011, p. 24; Brack *et al.*, unpublished manuscript).

Biology

Hibernation

Eastern small-footed bats hibernate during the winter months to conserve energy from increased thermoregulatory demands and reduced food resources. To increase energy savings, individuals enter a state of torpor where internal body temperatures approach ambient temperature, metabolic rates are significantly lowered, and immune function declines (Thomas *et al.* 1990, p. 475; Thomas and Geiser 1997, p. 585; Bouma *et al.* 2010, p. 623). Periodic arousal from torpor naturally occurs in all hibernating mammals (Lyman *et al.* 1982, p. 92), although arousals remain among the least understood of hibernation phenomena (Thomas and Geiser 1997, p. 585). Numerous factors (*e.g.*, reduction of metabolic waste, body temperature theories, and water balance theory) have been proposed to account for the occurrence and frequency of arousals (Thomas and Geiser 1997, p. 585). Each time a bat arouses from torpor, it uses a significant amount of energy to warm its body and increase its metabolic rate. The cost and number of arousals are the two key factors that determine energy expenditures of hibernating bats in winter (Thomas *et al.* 1990, p. 475). For example, little brown bats (*Myotis lucifugus*) used as much fat during a typical arousal from hibernation as would be used during 68 days of torpor, and arousals and subsequent activity may constitute 84 percent of the total energy used by hibernating bats during the winter (Thomas *et al.* 1990, pp. 477–478).

Of all hibernating bats, eastern small-footed bats are among the last to enter hibernacula and the first to emerge in the spring (Barbour and Davis 1969, p. 104). Hibernation is approximately mid-November to March (Barbour and Davis 1969, p. 104; Dalton 1987, p. 373); however, there are indications that eastern small-footed bats are active during mild winter weather (Mohr 1936, p. 64; Fenton 1972, p. 5). Fenton (1972, p. 5) observed that when temperatures at hibernation sites rose above 4°

Celsius (C) (39.2 °F (F)), eastern small-footed bats, along with big brown bats, aroused and departed from caves and mines. Whether these bats departed to take advantage of prey availability during mild winter spells or seek out other hibernation sites was never determined. Frequent oscillations in microclimate near cave or mine entrances may contribute to frequent arousals from torpor by eastern small-footed bats (Hitchcock 1965, p. 8). Frequent arousals may deplete energy reserves at a faster rate than would more continuous torpor characteristic of other cave-hibernating bats, contributing to a lower survival rate compared to other *Myotis* bats (Hitchcock *et al.* 1984, p. 129). Eastern small-footed bats lose up to 16 percent of their body weights during hibernation (Fenton 1972, p. 5).

Eastern small-footed bats often hibernate solitarily or in small groups and have been found hibernating in the open, in small cracks in cave walls and ceilings, in rock crevices in cave or mine floors, and beneath rocks (Hitchcock 1949, p. 53; Davis 1955, p. 130; Martin *et al.* 1966, p. 349; Barbour and Davis 1969, p. 104; Banfield 1974, p. 52; Dalton 1987, p. 373). Martin *et al.* (1966, p. 349) observed up to 30 eastern small-footed bats hanging from the ceilings of two mines in New York. From one small fissure, Hitchcock (1949, p. 53) extracted 35 eastern small-footed bats that were packed so tightly that it appeared almost impossible for those farthest in to get air. This propensity for hibernating in narrow cracks and crevices may mean they are sometimes overlooked by surveyors. In Maryland, for example, far fewer eastern small-footed bats were observed by surveyors during internal hibernacula surveys than were caught in traps during spring emergence (Maryland Department of Natural Resources 2011, unpublished data).

Eastern small-footed bats have been observed hibernating in caves that also contain little brown bats, big brown bats, northern long-eared bats (*Myotis septentrionalis*), Indiana bats (*Myotis sodalis*), tri-colored bats (*Perimyotis subflavus*), Virginia big-eared bats (*Corynorhinus townsendii virginianus*), gray bats (*Myotis grisescens*), and Rafinesque's big-eared bats (*Corynorhinus rafinesquii rafinesquii*), and approximately equal numbers of males and females occupy the same areas and cluster together indiscriminately (Hitchcock 1949, pp. 48–49; Hitchcock 1965, pp. 6–8; Fenton 1972, p. 3; Best and Jennings 1997, p. 3; Hemberger 2011, unpublished data; Graeter 2011, unpublished data; Graham 2011, unpublished data). Fenton (1972, p. 5) commonly observed eastern small-footed bats hibernating in physical contact with big brown bats, usually in small clusters of fewer than five bats, but never close to or in contact with little brown or Indiana bats. Eastern small-footed bats often hibernate in a horizontal position, tucked between cracks and crevices, unlike most *Myotis* bats, which hang in the open (Merritt 1987, p. 95). When suspended, however, the position of the forearm is unique in that, instead of hanging parallel to the body, as in other *Myotis* bats, the forearms are somewhat extended (Banfield 1974, p. 52). Like most bat species, eastern small-footed bats exhibit high site fidelity to hibernacula, with individuals returning to the same site year after year (Gates *et al.* 1984, p. 166).

Migration and Homing

Eastern small-footed bats have been observed migrating up to 19 kilometers (km) (12 miles (mi)) (Hitchcock 1955, p. 31) and as little as 0.1 km (0.06 mi) from winter hibernacula to summer roost sites (Johnson and Gates 2008, p. 456). The distance traveled is probably influenced by the availability of hibernacula and roosting sites across the landscape (Johnson and Gates 2008, p. 457). But in general, data suggest that this species hibernates in proximity to its summer range (van Zyll de Jong 1985, p. 119; Divoll *et al.* 2011). Eastern small-footed bats show a definite homing ability (Best and Jennings 1997, p. 4). Marked bats were present in the same cave in consecutive winters, and when moved to a different cave during the winter, they returned to the original cave the following winter (Mohr 1936, p. 64). In the Mammoth Cave region of Kentucky, eastern small-footed bats are fairly common in late summer in the groups of migrating bats, although the whereabouts of these bats at other seasons is unknown (Barbour and Davis 1969, p. 104).

Summer Roosts

Both males and females change summer roost sites often, even daily, although they typically are moving short distances within a general area (Chenger 2003, pp. 14–23; Johnson *et al.* 2011, p. 100; Brack *et al.*, unpublished manuscript). Chenger (2009, p. 7) suggests that eastern small-footed bats roost in low numbers over a wide area, such as talus fields, as a predator-avoidance strategy (Chenger 2009, p. 7). Frequent roost-switching may be another means of avoiding potential predators. Johnson *et al.* 2011 (pp. 98–101) radiotracked five lactating female bats and five nonreproductive males and observed that females and males switched roosts on average every 1.1 days. Males traveled an average of 41 m (135 ft) between consecutive roosts. Females traveled an average of 67 m (218 ft) between consecutive roosts, and roosts were closer to ephemeral water sources than those used by males.

Johnson *et al.* 2011 (p. 103) hypothesized that roost selection is based on either avoiding detection by predators or minimizing energy expenditures. They observed that roosts were located within 15 m (50 ft) from vegetation or forest edge and in areas with low canopy cover, which consequently provided a short distance to protective cover and high solar exposure. It appears eastern small-footed bats exhibit fidelity to their summer roosting areas, as demonstrated by the recapture of banded bats in successive years at the Surry Mountain Reservoir and Acadia National Park (Divoll *et al.* 2013; Veilleux and Moosman, unpublished data).

Reproduction

Available data regarding the eastern small-footed bat suggest that females of this species form small summer colonies, with males roosting singly or in small groups (Erdle and Hobson 2001, p. 10; Johnson *et al.* 2011, p. 100). Small maternity colonies of 12 to 20 individuals occurring in buildings have been reported (Merritt 1987, p. 95). Eastern small-footed bats are thought to be similar to sympatric *Myotis* that breed in the fall; spermatozoa are stored in the uterus of hibernating females until spring ovulation, and a single pup is born in May or June (Barbour and Davis 1969, p. 104; Amelon and Burhans 2006, p. 58). Brack *et al.* (unpublished manuscript) captured two female eastern small-footed bats in the fall that appeared to have recently mated as noted by fluids around the vagina. Two female eastern small-footed bats caught on June 20 and 24 were pregnant, and 16 female bats caught from June 23 to July 15 were lactating (Brack *et al.*, unpublished manuscript).

Adult longevity is estimated to be up to 12 years in the wild (Hitchcock 1965, p. 11). Estimated mean annual survival is low compared to other *Myotis*, and survival rates are significantly lower for females than for males, 42 and 75 percent, respectively (Hitchcock *et al.* 1984, p. 128). The lower rate of survival of females may be a result of a combination of factors: The greater demands of reproduction on females; the higher metabolic rates and less frequent torpor; and the greater exposure to possible disease-carrying parasites in maternity colonies

(Hitchcock *et al.* 1984, p. 127). Low survivorship in combination with low reproductive potential (*i.e.*, one offspring produced per year) (Best and Jennings 1997, p. 2) may explain why eastern small-footed bats are generally uncommon (Hitchcock *et al.* 1984, p. 129).

Foraging Behavior and Home Range

Eastern small-footed bats have low wing loading and high, frequency-modulated echolocation calls, making them capable of foraging efficiently in cluttered forest interiors (Johnson *et al.* 2009, p. 5). Although some accounts state that this species emerges early in the evening (van Zyll de Jong 1985, p. 119), Brack *et al.* (unpublished manuscript) found that activity peaked well after dark, and low post-midnight activities point to the possibility of a bimodal activity period. Most observations indicate that eastern small-footed bats fly slow and close to the ground, usually at heights from 0.6 to 3.5 m (2 to 11.5 ft) (Davis *et al.* 1965, p. 683; Brack *et al.*, unpublished manuscript).

Using ridgelines, streams, and forested roads as travel corridors, eastern small-footed bats have been observed travelling from 0.8 to 13.2 km (0.5 to 8.2 mi) between daytime roost sites and foraging areas (Chenger 2003, pp. 14–23; Chenger 2008b, p. 6; Johnson *et al.* 2009, p. 3; Mumma and Capouillez 2011, p. 24). Considerable declines in eastern small-footed bat capture rates have been observed with increasing distance from available rock habitat; and short distances between roosts and capture sites suggest these bats have small home ranges (Johnson *et al.* 2011, p. 104). Observed home range varies from 10.2 to 1,405 hectares (ha) (25 to 3,472 acres (ac)) (Johnson *et al.* 2009, p. 3; Mumma and Capouillez 2011, p. 25), although core habitat for three male and two female eastern small-footed bats ranged from 4 to 75 ha (10 to 185 ac) (50 percent fixed kernel utilization distribution) (Mumma and Capouillez 2011, p. 25).

Food habits of eastern small-footed bats are those of a generalist, although moths (Lepidoptera), true flies (Diptera), and beetles (Coleoptera) compose most of their diet (Johnson and Gates 2007, p. 319; Moosman *et al.* 2007, p. 355; Brack *et al.*, unpublished manuscript). Presence of spiders (Araneae) and crickets (Gryllidae) in the diet suggest eastern small-footed bats capture some prey via gleaning (Moosman *et al.* 2007, p. 358). Gleaning behavior is characterized by catching prey on surfaces via echolocation; calls are generally short in duration, high frequency, and of low intensity, characteristics that are difficult for some invertebrate prey to detect (Faure *et al.* 1993, p. 174).

Species Information

Northern Long-Eared Bat

Taxonomy and Species Description

The northern long-eared bat belongs to the order Chiroptera, suborder Microchiroptera, family Vespertilionidae, subfamily Vesperitilionae, genus Myotis, subgenus Myotis (Caceres and Barclay 2000, p. 1). The northern long-eared bat was considered a subspecies of Keen's long-eared Myotis (*Myotis keenii*) (Fitch and Schump 1979, p. 1), but was recognized as a distinct species by van Zyll de Jong in 1979 (1979, p. 993) based on geographic separation and difference in morphology (as cited in Caceres and Pybus 1997 p. 1; Caceres and Barclay 2000, p. 1; Nagorsen and Brigham 1993, p. 87; Whitaker and Hamilton 1998, p. 99; Whitaker and Mumford 2009, p. 207; Simmons 2005, p. 516). No subspecies have been described for this species (Nagorsen and Brigham 1993, p. 90; Whitaker and Mumford 2009, p. 214; van Zyll de Jong 1985, p. 94). This species has been recognized by different common names, such as: Keen's bat (Whitaker and Hamilton 1998, p. 99), northern myotis bat (Nagorsen and Brigham 1993, p. 87, Whitaker and Mumford 2009, p. 207), and the northern bat (Foster and Kurta 1999, p. 660). For the purposes of this finding, we refer to this species as the northern long-eared bat, and recognize it as a listable entity under the Act.

A medium-sized bat species, the northern long-eared bat adult body weight averages 5 to 8 g (0.2 to 0.3 ounces), with females tending to be slightly larger than males (Caceres and Pybus 1997, p. 3). Average body length ranges from 77 to 95 mm (3.0 to 3.7 in), tail length between 35 and 42 mm (1.3 to 1.6 in), forearm length between 34 and 38 mm (1.3 to 1.5 in), and wingspread between 228 and 258 mm (8.9 to 10.2 in) (Caceres and Barclay 2000, p. 1; Barbour and Davis 1969, p. 76). Pelage (fur) colors include medium to dark brown on its back, dark brown, but not black, ears and wing membranes, and tawny to pale-brown fur on the ventral side (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207). As indicated by its common name, the northern long-eared bat is distinguished from other *Myotis* species by its long ears (average 17 mm (0.7 in), Whitaker and Mumford 2009, p. 207) that, when laid forward, extend beyond the nose but less than 5 mm (0.2 in) beyond the muzzle (Caceres and Barclay 2000, p. 1). The tragus (projection of skin in front of the external ear) is long (average 9 mm (0.4 in); Whitaker and Mumford 2009, p. 207), pointed, and symmetrical (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207). Within its range, the northern long-eared bat can be confused with the little brown bat or the western long-eared myotis (*Myotis evotis*). The northern long-eared bat can be distinguished from the little brown bat by its longer ears, tragus, slightly longer tail, and less glossy pelage (Caceres and Barclay 2000, p. 1). The northern long-eared bat can be distinguished from the western long-eared myotis by its darker pelage and paler membranes (Caceres and Barclay 2000, p. 1).

Distribution and Abundance

The northern long-eared bat ranges across much of the eastern and north central United States, and all Canadian provinces west to the southern Yukon Territory and eastern British Columbia (Nagorsen and Brigham 1993, p. 89; Caceres and Pybus 1997, p. 1; Environment Yukon 2011, p. 10). In the United States, the species' range reaches from Maine west to Montana, south to eastern Kansas, eastern Oklahoma, Arkansas, and east to the Florida panhandle (Whitaker and Hamilton 1998, p. 99; Caceres and Barclay 2000, p. 2; Wilson and Reeder 2005, p. 516; Amelon and Burhans 2006, pp. 71–72). The species' range includes the following 39 States (including the District of Columbia, which we count as one of the "States"): Alabama, Arkansas, Connecticut, Delaware, the District of Columbia, Florida, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, Montana, Nebraska, New Hampshire, New Jersey, New York, North Carolina, North Dakota, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, South Dakota, Tennessee, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming. Historically, the species has been most frequently observed in the northeastern United States and in Canadian Provinces, Quebec and Ontario, with sightings increasing during swarming and hibernation (Caceres and Barclay 2000, p. 2). However, throughout the majority of the species' range it is patchily distributed, and historically was less common in the southern and western portions of the range than in the northern portion of the range (Amelon and Burhans 2006, p. 71).

Although they are typically found in low numbers in inconspicuous roosts, most records of northern long-eared bats are from winter hibernacula surveys (Caceres and Pybus 1997, p. 2) (for more information on use of hibernacula, see *Biology* below). More than 780 hibernacula have been identified throughout the species' range in the United States, although many hibernacula contain only a few (1 to 3) individuals (Whitaker and Hamilton 1998, p. 100). Known hibernacula (sites with one or more winter records) include: Arkansas (n=20), Connecticut (n=5), Georgia (n=1), Illinois (n=36), Indiana (n=25), Kentucky (n=90), Maine (n=3), Maryland (n=11), Massachusetts (n=7), Michigan (n=94), Minnesota (n=11), Missouri (n=>111), Nebraska (n=2), New Hampshire (n=9), New Jersey (n=8), New York (n=58), North Carolina (n=20), Oklahoma (n=4), Ohio (n=3), Pennsylvania (n=112), South Carolina (n=2), South Dakota (n=7), Tennessee (n=11), Vermont (n=13 (23 historical)), Virginia (n=8), West Virginia (n=104), and Wisconsin (n=45). Other states within the species' range have no known hibernacula (due to no suitable hibernacula present or lack of survey effort). They are typically found roosting in small crevices or cracks on cave or mine walls or ceilings, thus are easily overlooked during surveys and usually observed in small numbers (Griffin 1940, pp. 181–182; Barbour and Davis 1969, p. 77; Caire *et al.* 1979, p. 405; Van Zyll de Jong 1985, p. 9; Caceres and Pybus 1997, p. 2; Whitaker and Mumford 2009, pp. 209–210).

The U.S. portion of the northern long-eared bat's range can be described in four parts, as discussed below: the eastern population, Midwestern population, the southern population, and the western population.

Eastern Population

Historically, the northern long-eared bat was most abundant in the eastern portion its range (Caceres and Barclay 2000, p. 2). Northern long-eared bats have been consistently caught during summer mist nets surveys and detected during acoustic surveys in eastern populations. Large numbers of northern long-eared bats have been found in larger hibernacula in Pennsylvania (*e.g.*, an estimated 881 individuals in a mine in Bucks County, Pennsylvania in 2004). Fall swarm trapping conducted in September–October 1988–1989, 1990–1991, and 1999–2000 at two hibernacula with large historical numbers of northern long-eared bats had total captures ranging from 6 to 30 bats per hour, which demonstrated that the species was abundant at these hibernacula (Pennsylvania Game Commission, unpublished data, 2012).

In Delaware, the species is rare and no hibernacula are documented within the State; however, there is a historical record from Newcastle County in 1970 (Niederriter 2012, pers. comm.). In Connecticut, the northern long-eared bat was historically one of the most commonly encountered bats in the State and had been documented statewide (Dickson 2011, pers. comm.). In Maine, 3 hibernacula are known (all on private land), and the species has also been found in the summer in Acadia National Park (DePue 2012, unpublished data) where northern long-eared bats were found to be fairly common in 2009–2010 (242 northern long-eared bats captured comprising 27 percent of the total captures for the areas surveyed) (NPS 2010).

In Maryland, three of seven known hibernacula for the species are railroad tunnels, and no summer mist net or acoustic surveys have been conducted for the species (Feller 2011, unpublished data). In Massachusetts, there are 7 known hibernacula, 42 percent of which are privately owned. In New Hampshire, northern long-eared bats are known to inhabit at least nine mines and two World War II bunkers and have been found in summer surveys, including at Surry Mountain Dam (Brunkhurst 2012, unpublished data). In the White Mountain National Forest in New Hampshire in 1993–1994, northern long-eared was one of the most common species captured (27 percent) (Sasse and Pekins 1996, pp. 93–95). In New Jersey, one of the seven known hibernacula is a cave, and the remainder are mines (Markuson 2011, unpublished data). Northern long-eared bats consisted of 6 to 14 percent of total number of captures at Wallkill River National Wildlife Refuge in New Jersey from 2006–2010 (Kitchell and Wight 2011).

In Vermont, prior to 2009, the species was found in 23 hibernacula, totaling an estimated 595 animals, which was thought to be an under-estimate due to the species' preference for hibernating in hibernacula cracks and crevices. Summer capture data (2001–2007) indicated that northern long-eared bats comprised 19 percent of bats captured; it was considered the second most common bat species in the State (Smith 2011, unpublished data). In Virginia, they were historically considered "fairly common" during summer mist net surveys; however, they are considered "uncommon" during winter hibernacula surveys (Reynolds 2012, unpublished data).

In West Virginia, northern long-eared bats are found regularly in hibernacula surveys, but typically in small numbers (less than 20 individuals) in caves (Stihler 2012, unpublished data). The species has also been found in 41 abandoned coal mines in winter surveys conducted from 2002 to 2011 in the New River Gorge National River and Gauley River National Recreation Area, both managed by the National Park Service (NPS); the largest number observed was 157 in one of the NPS mines (NPS 2011, unpublished data). Northern long-eared bats are considered common in summer surveys in West Virginia; in summer records from 2006–2011 northern long-eared bat captures comprised 46 to 49 percent of all bat captures (Stihler 2012, pers. comm.).

Northern long-eared bats have been observed in 58 hibernacula in abandoned mines, caves, and tunnels in New York. They have also been observed in summer mist net and acoustic surveys. Summer mist-net surveys in New York from 2003–2008 resulted in a range of 0.21–0.47 bats/net night and declined to 0.012 bats/net night in 2011 (Herzog 2012, unpublished data). They have also been observed on Fort Drum in New York, where acoustic surveys (2003–2010) and mist net surveys (1999, 2007) have monitored the summer population (Dobony 2011, unpublished data). There are no known hibernacula in Rhode Island; however, there were 6 records from 2011 mist-net surveys in Washington County (Brown 2012, unpublished data).

Midwest Population

The northern long-eared bat is commonly encountered in summer mist-net surveys throughout the majority of the Midwest and is considered fairly common throughout much of the region. However, the species is often found infrequently and in small numbers in hibernacula surveys throughout most of the Midwest. In Missouri, northern long-eared bats were listed as a State species of conservation concern until 2007, after which it was decided the species was more common than previously thought because they were commonly captured in mist net surveys (Elliot 2013, pers. comm.). Historically, the northern long-eared bat was considered quite common throughout much of Indiana, and was the fourth or fifth most abundant bat species in the State in 2009. The species has been captured in at least 51 counties, is often captured in mist-nets along streams, and is the most common bat taken by trapping at mine entrances (Whitaker and Mumford 2009, pp. 207–

208). The abundance of northern long-eared bats appears to vary within Indiana during the summer. For example, during 3 summers (1990–1992) of mist-netting surveys in the northern half of Indiana, 37 northern long-eared bats were captured at 22 of 127 survey sites, which represented 4 percent of all bats captured (King 1993, p. 10). In contrast, northern long-eared bats were the most commonly captured bat species (38 percent of all bats captured) during three summers (2006–2008) of mist netting on two State forests in south-central Indiana (Sheets *et al.* 2013, p. 193). Indiana has 25 hibernacula with winter records of one or more northern long-eared bats. However, it is very difficult to find individuals in caves and mines during hibernation in large numbers in Indiana hibernacula (Whitaker and Mumford 2009, p. 208).

In Michigan, the northern long-eared bat is known from 25 counties and is not commonly encountered in the State except in parts of the northern Lower Peninsula and portions of the Upper Peninsula (Kurta 1982, p. 301; Kurta 2013, pers. comm.). The majority of hibernacula in Michigan are in the far northern and western Upper Peninsula; therefore, there are very few cave-hibernating bats in general in the southern half of the Lower Peninsula during the summer because the distance to hibernacula is too great (Kurta 2013, pers. comm.). It is thought that the few bats that do spend the summer in the southern half of the Lower Peninsula may hibernate in caves or mines in neighboring states, such as Indiana (Kurta 1982, pp. 301–302; Kurta 2013, pers. comm.).

In Wisconsin, the species is reported to be uncommon (Amelon and Burhans 2006, pp. 71–72). "Although the northern long-eared bat can be found in many parts of Wisconsin, it is clearly not abundant in any one location. The department has determined that the Northern long-eared bat is one of the least abundant bats in Wisconsin through cave and mine hibernacula counts, acoustic surveys, mist-netting in summer foraging areas and harp trap captures during the fall swarming period" (Redell 2011, pers. comm.). Northern long-eared bats are regularly caught in mist-net surveys in the Shawnee National Forest in southern Illinois (Kath 2013, pers. comm.). Further, the average number of northern long-eared bats caught during surveys between 1999 and 2011 at Oakwood Bottoms in the Shawnee National Forest has been fairly consistent (Carter 2012, pers. comm.). In Iowa, there are only summer mist net records for the species; in 2011 there were eight records (including three lactating females) from west-central Iowa (Howell 2011, unpublished data). In Minnesota, one mine in St. Louis County may contain a large number of individuals, possibly over 3,000; however, this is a very rough estimate since the majority of the mine cannot be safely accessed for surveys (Nordquist 2012, pers. comm.). In Ohio, there are three known hibernacula and the largest population in Preble County has had more than 300 bats. In general, northern long-eared bats are also regularly collected as incidental catches in mist-net surveys for Indiana bats in Ohio (Boyer 2012, pers. comm.).

Southern Population

The northern long-eared bat is less common in the southern portion of its range than in the northern portion of the range (Amelon and Burhans 2006, p. 71) and, in the South, is considered more common in states such as Kentucky and Tennessee, and more rare in the southern extremes of the range (*e.g.*, Alabama, Georgia, South Carolina). In Alabama, the northern long-eared bat is rare, while in Tennessee it is uncommon (Amelon and Burhans 2006, pp. 71–72). In Tennessee, northern long-eared bats were found in summer mist-net surveys conducted through summer of 2010 in addition to hibernacula censuses. Northern long-eared bats were found in 11 caves surveyed in 2011 in Tennessee (Pelren 2011, pers. comm.). In 2000, during sampling of bat populations in the Kisatchie National Forest, Louisiana, three northern long-eared bat specimens were collected; these were the first official records of the species from Louisiana (Crnkovic 2003, p. 715). In Georgia, northern long-eared bats have been found at 1 of 5 known hibernacula in the State and 24 summer records were found between 2007 and 2011. Mist-net surveys were conducted in the Chattahoochee National Forest in 2001–2002 and 2006–2007, with 51 total records for the species (Morris 2012, unpublished data). Northern long-eared bats have been found in 20 hibernacula within North Carolina (Graeter 2011, unpublished data). In the summer of 2007, (Morris *et al.* 2009, p. 356) six northern long-eared bats were captured in Washington County, North Carolina. Both adults and juveniles were captured, suggesting that there is a reproducing resident population (Morris *et al.* 2009, p. 359). In Kentucky, although typically found in small numbers, northern long-eared bats were historically found in the majority of hibernacula in Kentucky and have been a commonly captured species during summer surveys (Hemberger 2012, pers. comm.). The northern long-eared bat can be found throughout the majority of Kentucky, with historical records in 91 of its 120 counties. Eighty-five counties have summer records, and 68 of those include reproductive records (*i.e.*, captures of juveniles or pregnant, lactating, or post-lactating adult females) (Hemberger 2012, pers. comm.). In South Carolina, there are two known hibernacula: one is a cave that had 26 bats present in 1995, but has not been surveyed since, and the other is a tunnel where only one bat was found in 2011 (Bunch 2011, unpublished data). Northern long-eared bats are known from 20 hibernacula in Arkansas, although they are typically found in very low numbers (Sasse 2012, unpublished data). Surveys in the Ouachita Mountains of central Arkansas from 2000–2005 tracked 17 males and 23 females to 43 and 49 day roosts, respectively (Perry and Thill 2007, pp. 221–222). The northern long-eared bat is known to occur in seven counties along the eastern edge of Oklahoma, (Stevenson 1986, p. 41). The species has been recorded in 21 caves (7 of which occur on the Ozark Plateau National Wildlife Refuge) during the summer. The species has regularly been captured in summer mist-net surveys at cave entrances in Adair, Cherokee, Sequoyah, Delaware, and LeFlore counties, and are often one of the most common bats captured during mist-net surveys at cave entrances in the Ozarks of northeastern Oklahoma (Stark 2013, pers. comm.). Small numbers of northern long-eared bats (typical range of 1–17 individuals) also have been captured during mist-net surveys along creeks and riparian zones in eastern Oklahoma.

Western Population

The northern long-eared bat is generally less common in the western portion of its range than in the northern portion of the range (Amelon and Burhans 2006, p. 71) and is considered common in only small portions of the western part of its range (*e.g.*, Black Hills of South Dakota) and uncommon or rare in the western extremes of the range (*e.g.*, Wyoming, Kansas, Nebraska) (Caceres and Barclay 2000, p. 2). The northern long-eared bat has been observed hibernating and residing during the summer and is considered abundant in the Black Hills National Forest in South Dakota. Capture and banding data for survey efforts in the Black Hills of South Dakota and Wyoming showed northern long-eared bats to be the second most common bat banded (159 of 878 total bats) during 3 years of survey effort (Tigner and Aney

1994, p. 4). South Dakota contains seven known hibernacula, five of which are abandoned mines. The largest number of individuals was found in a hibernaculum near Hill City, South Dakota; 40 individuals were found in this mine in the winter of 2002–2003 (Tigner and Stukel 2003, pp. 27–28). A summer population was found on the habitats in Dakota Prairie National Grassland and Custer National Forest in 2005 (Lausen undated, unpublished data). Also, northern long-eared bats have been captured during the summer along the Missouri River in South Dakota (Swier 2006, p. 5; Kiesow and Kiesow 2010, pp. 65–66). Summer surveys in North Dakota (2009–2011) documented the species in the Turtle Mountains, the Missouri River Valley, and in the Badlands (Gillam and Barnhart 2011, pp. 10–12). No hibernacula are known within North Dakota; however, there has been very limited survey effort in the State (Riddle 2012, pers. comm.).

Northern long-eared bats have been observed at two quarries located in east-central Nebraska, but there is no survey data for either of these sites (Geluso 2011, unpublished data). They are also known to summer in the northwestern parts of Nebraska, specifically Pine Ridge in Sheridan County (only males have been documented), and a reproducing population has been documented north of Valentine in Cherry County (Benedict *et al.* 2000, pp. 60–61). During an acoustic survey conducted during the summer of 2012, the species was common in Cass County (east-central Nebraska), but was uncommon or absent from extreme southeastern Nebraska (White *et al.* 2012, p. 2). The occurrence of this species in Cass County, Nebraska is likely attributable to limestone quarries in the region that are used as hibernacula by this species and others (White *et al.* 2012, p. 3).

During acoustic and mist net surveys conducted throughout Wyoming in the summers of 2008–2011, 27 separate observations of northern long-eared bats were made in the northeast part of the State and breeding was confirmed (Wyoming Game and Fish Department 2012, unpublished data). To date, there are no known hibernacula in Wyoming and it is unclear if there are existing hibernacula, although the majority of potential hibernacula (abandoned mines) within the State occur outside of the northern long-eared bat's range (Tigner and Stukel 2003, p. 27; Wyoming Game and Fish Department 2012). Montana has only one known record: a male collected in an abandoned coal mine in 1978 in Richland County (Montana Fish, Wildlife, and Parks 2012). In Kansas, the northern long-eared bat was first found in summer mist-net surveys in 1994 and 1995 in Osborne and Russell counties, before which the species was thought to only migrate through parts of the State (Sparks and Choate 1995, p. 190).

Canada Population

The northern long-eared bat occurs throughout the majority of the forested regions of Canada, although it is found in higher abundance in eastern Canada than in western Canada, similar to in the United States (Caceres Pybus 1997, p. 6). However, the scarcity of records in the western parts of Canada may be due to more limited survey efforts. It has been estimated that approximately 40 percent of the northern long-eared bat's global range is in Canada; however, due to the species being relatively common and widespread, limited effort has been made to determine overall population size within Canada (COSEWIC 2012, p.9). The range of the northern long-eared bat in Canada includes Alberta, British Columbia, Manitoba, New Brunswick, Newfoundland and Labrador, Northwest Territories, Nova Scotia, Prince Edward Island, Ontario, Quebec, Saskatchewan, and Yukon (COSEWIC 2012, p. 4). There are no records of the species overwintering in Yukon and Northwest Territories (COSEWIC 2012, p. 9).

Habitat

Winter Habitat

Northern long-eared bats predominantly overwinter in hibernacula that include caves and abandoned mines. Hibernacula used by northern long-eared bats are typically large, with large passages and entrances (Raesly and Gates 1987, p. 118), relatively constant, cooler temperatures (0 to 9 °C (32 to 48 °F) (Raesly and Gates 1987, p. 18; Caceres and Pybus 1997, p. 2; Brack 2007, p. 744), and with high humidity and no air currents (Fitch and Shump 1979, p. 2; Van Zyll de Jong 1985, p. 94; Raesly and Gates 1987 p. 118; Caceres and Pybus 1997, p. 2). The sites favored by northern long-eared bats are often in very high humidity areas, to such a large degree that droplets of water are often observed on their fur (Hitchcock 1949, p. 52; Barbour and Davis 1969, p. 77). Northern long-eared bats typically prefer cooler and more humid conditions than little brown bats, similar to the eastern small-footed bat and big brown bat, although the latter two species tolerate lower humidity than northern long-eared bats (Hitchcock 1949, p. 52–53; Barbour and Davis 1969, p. 77; Caceres and Pybus 1997, p. 2). Northern long-eared bats are typically found roosting in small crevices or cracks in cave or mine walls or ceilings, often with only the nose and ears visible, thus are easily overlooked during surveys (Griffin 1940, pp. 181–182; Barbour and Davis 1969 p.77; Caire *et al.* 1979, p. 405; Van Zyll de Jong 1985, p.9; Caceres and Pybus 1997, p. 2; Whitaker and Mumford 2009, pp. 209–210). Caire *et al.* (1979, p. 405) and Whitaker and Mumford (2009, p. 208) commonly observed individuals exiting caves with mud and clay on their fur, also suggesting the bats were roosting in tighter recesses of hibernacula. They are also found hanging in the open, although not as frequently as in cracks and crevices (Barbour and Davis 1969, p.77, Whitaker and Mumford 2009, pp. 209–210). In 1968, Whitaker and Mumford (2009, pp. 209–210) observed three northern long-eared bats roosting in the hollow core of stalactites in a small cave in Jennings County, Indiana.

To a lesser extent, northern long-eared bats have been found overwintering in other types of habitat that resemble cave or mine hibernacula, including abandoned railroad tunnels, more frequently in the northeast portion of the range. Also, in 1952 three northern long-eared bats were found hibernating near the entrance of a storm sewer in central Minnesota (Goehring 1954, p. 435). Kurta and Teramino (1994, pp. 410–411) found northern long-eared bats hibernating in a hydro-electric dam facility in Michigan. In Massachusetts, northern long-eared bats have been found hibernating in the Sudbury Aqueduct, a structure created in the late 1800s to transfer water, but that is rarely used for this purpose today (French 2012, unpublished data). Griffin (1945, p. 22) found northern long-eared bats in December in Massachusetts in a dry well, and commented that these bats may regularly hibernate in "unsuspected retreats" in areas where caves or mines are not present.

Summer Habitat

During the summer, northern long-eared bats typically roost singly or in colonies underneath bark or in cavities or crevices of both live trees and snags (Sasse and Perkins 1996, p. 95; Foster and Kurta 1999, p. 662; Owen *et al.* 2002, p. 2; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 222; Timpone *et al.* 2010, p. 119). Males and non-reproductive females' summer roost sites may also include cooler locations, including caves and mines (Barbour and Davis 1969, p. 77; Amelon and Burhans 2006, p. 72). Northern long-eared bats have also been observed roosting in colonies in humanmade structures, such as buildings, barns, a park pavilion, sheds, cabins, under eaves of buildings, behind window shutters, and in bat houses (Mumford and Cope 1964, p. 72; Barbour and Davis 1969, p. 77; Cope and Humphrey 1972, p. 9 ; Amelon and Burhans 2006, p. 72; Whitaker and Mumford 2009, p. 209; Timpone *et al.* 2010, p. 119; Joe Kath 2013, pers. comm.).

The northern long-eared bat appears to be somewhat opportunistic in tree roost selection, selecting varying roost tree species and types of roosts throughout its range, including tree species such as black oak (*Quercus velutina*), northern red oak (*Quercus rubra*), silver maple (*Acer saccharinum*), black locust (*Robinia pseudoacacia*), American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), sourwood (*Oxydendrum arboreum*), and shortleaf pine (*Pinus echinata*) (*e.g.*, Mumford and Cope 1964, p. 72; Clark *et al.* 1987, p. 89; Sasse and Pekins 1996, p. 95; Foster and Kurta 1999, p. 662; Lacki and Schwierjohann 2001, p. 484; Owen *et al.* 2002, p. 2; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 224; Timpone *et al.* 2010, p. 119). Northern long-eared bats most likely are not dependent on a certain species of trees for roosts throughout their range; rather, certain tree species will form suitable cavities or retain bark and the bats will use them opportunistically (Foster and Kurta 1999, p. 668). Carter and Felhamer (2005, p. 265) speculated that structural complexity of habitat or available roosting resources are more important factors than the actual tree species.

Many studies have documented the northern long-eared bat's selection of live trees and snags, with a range of 10 to 53 percent selection of live roosts found (Sasse and Perkins 1996, p. 95; Foster and Kurta 1999, p. 668; Lacki and Schwierjohann 2001, p. 484; Menzel *et al.* 2002, p. 107; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 224; Timpone *et al.* 2010, p. 118). Foster and Kurta (1999, p. 663) found 53 percent of roosts in Michigan were in living trees, whereas in New Hampshire, 34 percent of roosts were in snags (Sasse and Pekins 1996, p. 95). The use of live trees versus snags may reflect the availability of such structures in study areas (Perry and Thill 2007, p. 224) and the flexibility in roost selection when there is a sympatric bat species present (*e.g.*, Indiana bat) (Timpone *et al.* 2010, p. 120). In tree roosts, northern long-eared bats are typically found beneath loose bark or within cavities and have been found to use both exfoliating bark and crevices to a similar degree for summer roosting habitat (Foster and Kurta 1999, p. 662; Lacki and Schwierjohann 2001, p. 484; Menzel *et al.* 2002, p. 110; Owen *et al.* 2002, p. 2; Perry and Thill 2007, p. 222; Timpone *et al.* 2010, p. 119).

Canopy coverage at northern long-eared bat roosts has ranged from 56 percent in Missouri (Timone *et al.* 2010, p. 118), 66 percent in Arkansas (Perry and Thill 2007, p. 223), greater than 75 percent in New Hampshire (Sasse and Pekins 1996, p. 95), to greater than 84 percent in Kentucky (Lacki and Schwierjohann 2001, p. 487). Studies in New Hampshire and British Columbia have found that canopy coverage around roosts is lower than in available stands (Caceres 1998; Sasse and Pekins 1996, p. 95). Females tend to roost in more open areas than males, likely due to the increased solar radiation, which aids pup development (Perry and Thill 2007, p. 224). Fewer trees surrounding maternity roosts may also benefit juvenile bats that are starting to learn to fly (Perry and Thill 2007, p. 224). However, in southern Illinois, northern long-eared bats were observed roosting in areas with greater canopy cover than in random plots (Carter and Feldhamer 2005, p. 263). Roosts are also largely selected below the canopy, which could be due to the species' ability to exploit roosts in cluttered environments; their gleaning behavior suggests an ability to easily maneuver around obstacles (Foster and Kurta 1999, p. 669; Menzel *et al.* 2002, p. 112).

Female northern long-eared bats typically roost in tall, large-diameter trees (Sasse and Pekins 1996, p. 95). Studies have found that the diameter-at-breast height (dbh) of northern long-eared bat roost trees was greater than random trees (Lacki and Schwierjohann 2001, p. 485) and others have found both dbh and height of selected roost trees to be greater than random trees (Sasse and Pekins 1996, p. 97; Owen *et al.* 2002 p. 2). However, other studies have found that roost tree mean dbh and height did not differ from random trees (Menzel *et al.* 2002, p. 111; Carter and Feldhamer 2005, p. 266). Lacki and Schwierjohann (2001, p. 486) have also found that northern long-eared bats roost more often on upper and middle slopes than lower slopes, which suggests a preference for higher elevations due to increased solar heating.

Biology

Hibernation

Similar to the eastern small-footed bat description above, the northern long-eared bats hibernate during the winter months to conserve energy from increased thermoregulatory demands and reduced food resources. In general, northern long-eared bats arrive at hibernacula in August or September, enter hibernation in October and November, and leave the hibernacula in March or April (Caire *et al.* 1979, p. 405; Whitaker and Hamilton 1998, p. 100; Amelon and Burhans 2006, p. 72). However, hibernation may begin as early as August (Whitaker and Rissler 1992, p. 56). In Copperhead Cave in west-central Indiana, the majority of bats enter hibernation during October, and spring emergence occurs mainly from about the second week of March to mid-April (Whitaker and Mumford 2009, p. 210). In Indiana, northern long-eared bats become more active and start feeding outside the hibernaculum in mid-March, evidenced by stomach and intestine contents. This species also showed spring activity earlier than little brown bats and tri-colored bat (Whitaker and Rissler 1992, pp. 56–57). In northern latitudes, such as in upper Michigan's copper-mining district, hibernation for northern long-eared bats and other *myotis* species may begin as early as late August and may last for 8 to 9 months (Stones and Fritz, 1969, p. 81; Fitch and Shump 1979, p. 2). Northern long-eared bats have shown a high degree of philopatry (using the same site multiple years) for a hibernaculum (Pearson 1962, p. 30), although they may not return to the same hibernaculum in successive seasons (Caceres and Barclay 2000, p. 2).

Typically, northern long-eared bats are not abundant and compose a small proportion of the total number of bats hibernating in a hibernaculum (Barbour and Davis 1969, p. 77; Mills 1971, p. 625; Caire *et al.* 1979, p. 405; Caceres and Barclay 2000, pp. 2–3). Although usually found in small numbers, the species typically inhabits the same hibernacula with large numbers of other bat species, and occasionally are found in clusters with these other bat species. Other species that commonly occupy the same habitat include: little brown bat, big brown bat, eastern small-footed bat, tri-colored bat, and Indiana bat (Swanson and Evans 1936, p. 39; Griffin 1940, p. 181; Hitchcock 1949, pp. 47–58; Stones and Fritz 1969, p. 79; Fitch and Shump 1979, p. 2). Whitaker and Mumford (2009, pp. 209–210), however, infrequently found northern long-eared bats hibernating beside little brown bats, Indiana bats, or tri-colored bats, since they found few hanging on side walls or ceilings of cave passages. Barbour and Davis (1969, p. 77) found that the species is never abundant and rarely recorded in concentrations of over 100 in a single hibernaculum.

Northern long-eared bats often move between hibernacula throughout the winter, which may further decrease population estimates (Griffin 1940, p. 185; Whitaker and Rissler 1992b, p. 131; Caceres and Barclay 2000 pp. 2–3). Whitaker and Mumford (2009, p. 210) found that this species flies in and out of some of the mines and caves in southern Indiana throughout the winter. In particular, the bats were active at Copperhead Cave periodically all winter, with northern long-eared bats being more active than other species (such as little brown bat and tri-colored bat) hibernating in the cave. Though northern long-eared bats fly outside of the hibernacula during the winter, they do not feed; hence the function of this behavior is not well understood (Whitaker and Hamilton 1998, p. 101). However, it has been suggested that bat activity during winter could be due in part to disturbance by researchers (Whitaker and Mumford 2009, pp. 210–211).

Northern long-eared bats exhibited significant weight loss during hibernation. In southern Illinois, weight loss during hibernation was found in male northern long-eared bats, with individuals weighing an average of 6.6 g (0.2 ounces) prior to 10 January, and those collected after that date weighing an average of 5.3 g (0.2 ounces) (Pearson 1962, p. 30). Whitaker and Hamilton (1998, p. 101) reported a weight loss of 41–43 percent over the hibernation period for northern long-eared bats in Indiana. In eastern Missouri, male northern long-eared bats lost an average of 3 g (0.1 ounces) during the hibernation period (late October through March), and females lost an average of 2.7 g (0.1 ounces) (Caire *et al.* 1979, p. 406).

Migration and Homing

While the northern long-eared bat is not considered a long-distance migratory species, short migratory movements between summer roost and winter hibernacula between 56 km (35 mi) and 89 km (55 mi) have been documented (Nagorsen and Brigham 1993 p. 88; Griffith 1945, p. 53). However, movements from hibernacula to summer colonies may range from 8 to 270 km (5 to 168 mi) (Griffin 1945, p. 22).

Several studies show a strong homing ability of northern long-eared bats in terms of return rates to a specific hibernaculum, although bats may not return to the same hibernaculum in successive winters (Caceres and Barclay 2000, p. 2). Banding studies in Ohio, Missouri, and Connecticut show return rates to hibernacula of 5.0 percent (Mills 1971, p. 625), 4.6 percent (Caire *et al.* 1979, p. 404), and 36 percent (Griffin 1940, p. 185), respectively. An experiment showed an individual bat returned to its home cave up to 32 km (20 mi) away after being removed 3 days prior (Stones and Branick 1969, p. 158). Individuals have been known to travel between 56 and 97 km (35 and 60 mi) between caves during the spring (Caire *et al.* 1979, p. 404; Griffin 1945, p. 20).

Summer Roosts

Northern long-eared bats switch roosts often (Sasse and Perkins 1996, p. 95), typically every 2–3 days (Foster and Kurta 1999, p. 665; Owen *et al.* 2002, p. 2; Carter and Feldhamer 2005, p. 261; Timpone *et al.* 2010, p. 119). In Missouri, the longest time spent roosting in one tree was 3 nights; however, the up to 11 nights spent roosting in a humanmade structure has been documented (Timpone *et al.* 2010, p. 118). Similarly, Carter and Feldhamer (2005, p. 261) found that the longest a northern long-eared bat used the same tree was 3 days; in West Virginia, the average time spent at one roost was 5.3 days (Menzel *et al.* 2002, p. 110). Bats switch roosts for a variety of reasons, including, temperature, precipitation, predation, parasitism, and ephemeral roost sites (Carter and Feldhamer 2005, p. 264). Ephemeral roost sites, with the need to proactively investigate new potential roost trees prior to their current roost tree becoming uninhabitable (*e.g.*, tree falls over), may be the most likely scenario (Kurta *et al.* 2002, p. 127; Carter and Feldhamer 2005, p. 264; Timpone *et al.* 2010, p. 119). In Missouri, Timpone *et al.* (2010, p. 118) radiotracked 13 northern long-eared bats to 39 roosts and found the mean distance between the location where captured and roost tree was 1.7 km (1.1 mi) (range 0.07–4.8 km (0.04–3.0 mi), and the mean distance traveled between roost trees was 0.67 km (0.42 mi) (range 0.05–3.9 km (0.03–2.4 mi)). In Michigan, the longest distance the same bat moved between roosts was 2 km (1.2 mi) and the shortest was 6 m (20 ft) (Foster and Kurta 1999, p. 665). In New Hampshire, the mean distance between foraging areas and roost trees was 602 m (1975 ft) (Sasse and Pekins 1996, p. 95). In the Ouachita Mountains of Arkansas, Perry and Thill (2007, p. 22) found that individuals moved among snags that were within less than 2 ha (5 ac).

Some studies have found tree roost selection to differ slightly between male and female northern long-eared bats. Male northern long-eared bats have been found to more readily use smaller diameter trees for roosting than females, suggesting males are more flexible in roost selection than females (Lacki and Schwierjohann 2001, p. 487; Broders and Forbes 2004, p. 606; Perry and Thill 2007, p. 224). In the Ouachita Mountains of Arkansas, both sexes primarily roosted in snags, although females roosted in snags surrounded by fewer midstory trees than did males (Perry and Thill 2007, p. 224). In New Brunswick, Canada, Broders and Forbes (2004, pp. 606–607) found that there was spatial segregation between male and female roosts, with female maternity colonies typically occupying more mature, shade-tolerant deciduous tree stands and males occupying more conifer-dominated stands. In northeastern Kentucky, males do not use colony roosting sites and are typically found occupying cavities in live hardwood trees, while females form colonies more often in both hardwood and softwood snags (Lacki and Schwierjohann 2001, p. 486).

The northern long-eared bat is comparable to the Indiana bat in terms of summer roost selection, but appears to be more opportunistic (Carter and Feldhamer 2005, pp. 265–266; Timpone *et al.* 2010, p. 120–121). In southern Michigan, northern long-eared bats used cavities within roost trees, living trees, and roosts with greater canopy cover more often than does the Indiana bat, which occurred in the same area (Foster and Kurta 1999, p. 670). Similarly, in northeastern Missouri, Indiana bats typically roosted in snags with exfoliating bark and low canopy cover, whereas northern long-eared bats used the same habitat in addition to live trees, shorter trees, and trees with higher canopy cover (Timpone *et al.* 2010 pp. 118–120). Although northern long-eared bats are more opportunistic than Indiana bats, there may be a small amount of roost selection overlap between the two species (Foster and Kurta 1999, p. 670; Timpone *et al.* 2010, pp. 120–121).

Reproduction

Breeding occurs from late July in northern regions to early October in southern regions and commences when males begin to swarm hibernacula and initiate copulation activity (Whitaker and Hamilton 1998, p. 101; Whitaker and Mumford 2009, p. 210; Caceres and Barclay 2000, p. 2; Amelon and Burhans 2006, p. 69). Copulation occasionally occurs again in the spring (Racey 1982, p. 73). Hibernating females store sperm until spring, exhibiting a delayed fertilization strategy (Racey 1979, p.

392; Caceres and Pybus 1997, p. 4). Ovulation takes place at the time of emergence from the hibernaculum, followed by fertilization of a single egg, resulting in a single embryo (Cope and Humphrey 1972, p. 9; Caceres and Pybus 1997, p. 4; Caceres and Barclay 2000, p. 2); gestation is approximately 60 days (Kurta 1994, p. 71). Males are reproductively inactive until late July, with testes descending in most males during August and September (Caire *et al.* 1979, p. 407; Amelon and Burhans 2006, p. 69).

Maternity colonies, consisting of females and young, are generally small, numbering from about 30 (Whitaker and Mumford 2009, p. 212) to 60 individuals (Caceres and Barclay 2000, p. 3); however, one group of 100 adult females was observed in Vermilion County, Indiana (Whitaker and Mumford 2009, p. 212). In West Virginia, maternity colonies in two studies had a range of 7–88 individuals (Owen *et al.* 2002, p. 2) and 11–65 individuals, with a mean size of 31 (Menzel *et al.* 2002, p. 110). Lacki and Schwierjohann (2001, p. 485) found that the population size of colony roosts declined as the summer progressed with pregnant females using the largest colonies (mean=26) and post-lactating females using the smallest colonies (mean=4), with the largest overall reported colony size of 65 bats. Other studies have also found that the number of individuals within a maternity colony typically decreases from pregnancy to post-lactation (Foster and Kurta 1999, p. 667; Lacki and Schwierjohann 2001, p. 485; Garroway and Broders 2007, p. 962; Perry and Thill 2007, p. 224; Johnson *et al.* 2012, p. 227). Female roost site selection, in terms of canopy cover and tree height, changes depending on reproductive stage; relative to pre- and post-lactation periods, lactating northern long-eared bats have been shown to roost higher in tall trees situated in areas of relatively less canopy cover and tree density (Garroway and Broders 2008, p. 91).

Adult females give birth to a single pup (Barbour and Davis 1969). Birthing within the colony tends to be synchronous, with the majority of births occurring around the same time (Krochmal and Sparks 2007, p. 654). Parturition (birth) likely occurs in late May or early June (Caire *et al.* 1979, p. 406; Easterla 1968, p. 770; Whitaker and Mumford 2009, p. 213), but may occur as late as July (Whitaker and Mumford 2009, p. 213). Broders *et al.* (2006, p. 1177) estimated a parturition date of July 20 in New Brunswick. Lactating and post-lactating females were observed in mid-June in Missouri (Caire *et al.* 1979, p. 407), July in New Hampshire and Indiana (Sasse and Pekins 1996, p. 95; Whitaker and Mumford 2009, p. 213), and August in Nebraska (Benedict 2004, p. 235). Juvenile volancy (flight) occurs by 21 days after parturition (Krochmal and Sparks 2007, p. 651, Kunz 1971, p. 480) and as early as 18 days after parturition (Krochmal and Sparks 2007, p. 651). Subadults were captured in late June in Missouri (Caire *et al.* 1979, p. 407), early July in Iowa (Sasse and Pekins 1996, p. 95), and early August in Ohio (Mills 1971, p. 625).

Adult longevity is estimated to be up to 18.5 years (Hall 1957, p. 407), with the greatest recorded age of 19 years (Kurta 1995, p. 71). Most mortality for northern long-eared and many other species of bats occurs during the juvenile stage (Caceres and Pybus 1997, p. 4).

Foraging Behavior and Home Range

The northern long-eared bat has a diverse diet including moths, flies, leafhoppers, caddisflies, and beetles (Nagorsen and Brigham 1993, p. 88; Brack and Whitaker 2001, p. 207; Griffith and Gates 1985, p. 452), with diet composition differing geographically and seasonally (Brack and Whitaker 2001, p. 208). Feldhamer *et al.* (2009, p. 49) noted close similarities of all *Myotis* diets in southern Illinois, while Griffith and Gates (1985, p. 454) found significant differences in the diets of northern long-eared bat and little brown bat. The most common insects found in the diets of northern long-eared bats are lepidopterans (moths) and coleopterans (beetles) (Feldhamer *et al.* 2009, p. 45; Brack and Whitaker 2001, p. 207) with arachnids (spiders) also being a common prey item (Feldhamer *et al.* 2009, p. 45).

Foraging techniques include hawking (catching insects in flight) and gleaning in conjunction with passive acoustic cues (Nagorsen and Brigham 1993, p. 88; Ratcliffe and Dawson 2003, p. 851). Observations of northern long-eared bats foraging on arachnids (Feldhamer *et al.* 2009, p. 49), presence of green plant material in their feces (Griffith and Gates 1985, p. 456), and non-flying prey in their stomach contents (Brack and Whitaker 2001, p. 207) suggest considerable gleaning behavior. Northern long-eared bats have the highest frequency call of any bat species in the Great Lakes area (Kurta 1995, p. 71). Gleaning allows this species to gain a foraging advantage for preying upon moths because moths are less able to detect these high frequency echolocation calls (Faure *et al.* 1993, p. 185). Emerging at dusk, most hunting occurs above the understory, 1 to 3 m (3 to 10 ft) above the ground, but under the canopy (Nagorsen and Brigham 1993, p. 88) on forested hillsides and ridges, rather than along riparian areas (Brack and Whitaker 2001, p. 207; LaVal *et al.* 1977, p. 594). This coincides with data indicating that mature forests are an important habitat type for foraging northern long-eared bats (Caceres and Pybus 1998, p. 2). Occasional foraging also takes place over forest clearings and water, and along roads (Van Zyll de Jong 1985, p. 94). Foraging patterns indicate a peak activity period within 5 hours after sunset followed by a secondary peak within 8 hours after sunset (Kunz 1973, p. 18–19). Brack and Whitaker (2001, p. 207) did not find significant differences in the overall diet of northern long-eared bats between morning (3 a.m. to dawn) and evening (dusk to midnight) feedings; however there were some differences in the consumption of particular prey orders between morning and evening feedings. Additionally, no significant differences existed in dietary diversity values between age classes or sex groups (Brack and Whitaker 2001, p. 208).

Female home range size may range from 19 to 172 ha (47–425 acres) (Lacki *et al.* 2009, p. 5). Owen *et al.* (2003, p. 353) estimated average maternal home range size to be 65 ha (161 ac). Home range size of northern long-eared bats in this study site was small relative to other bat species, but this may be due to the study's timing (during the maternity period) and the small body size of *M. septentrionalis* (Owen *et al.* 2003, pp. 354–355). The mean distance between roost trees and foraging areas of radio-tagged individuals in New Hampshire was 620 m (2034 ft) (Sasse and Pekins 1996, p. 95).

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.

We have carefully assessed the best scientific and commercial information available regarding the past, present, and future threats to the eastern small-footed and northern long-eared bats. Effects to both the eastern small-footed bat and northern long-eared bat from these factors are discussed together where the species are affected similarly.

There are several factors presented below that affect both the eastern small-footed and the northern long-eared bats to a greater or lesser degree; however, we have found that no other threat is as severe and immediate to the northern long-eared bat's persistence as the disease, white-nose syndrome (WNS), discussed below in Factor C. WNS is currently the predominant threat to the species, and if WNS had not emerged or was not affecting the northern long-eared bat populations to the level that it has, we presume the species' would not be experiencing the dramatic declines that it has since WNS emerged. Therefore, although we have included brief discussions of other factors affecting both species, the focus of the discussion below is on WNS.

Factor A. The Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

Hibernation Habitat

Modifications to bat hibernacula by erecting physical barriers (*e.g.*, doors, gates) to control cave access and mining can affect the thermal regime of the habitat, and thus the ability of the cave or mine to support hibernating bats, including the northern long-eared and, in some cases, the eastern small-footed bat. For example, the Service's Indiana Bat Draft Recovery Plan (2007, pp. 71–74) presents a discussion of well-documented examples of these type of effectss to cave-hibernating species that are also applicable to our discussion here. Modifications to cave and mine entrances, such as the addition of gates or other structures intended to exclude humans, not only restricts flight and movement (Hemberger 2011, unpublished data), but also changes airflow and alters internal microclimates of the caves and mines and eliminating their utility as hibernacula. For example, Richter *et al.* (1993, p. 409) attributed the decline in the number of Indiana bats at Wyandotte Cave, Indiana (which harbors one of the largest known population of hibernating Indiana bats), to an increase in the cave's temperature resulting from restricted airflow caused by a stone wall erected at the cave's entrance. After the wall was removed, the number of Indiana bats increased markedly over the next 14 years (Richter *et al.* 1993, p. 412; Brack *et al.* 2003, p. 67). In an eastern small-footed bat example, the construction associated with commercializing the Fourth Chute Cave in Ontario, Canada, eliminated the circulation of cold air in one of the unvisited passages where a relatively large number of eastern small-footed bats hibernated. These bats were completely displaced as a result of the warmer microclimate produced (Mohr 1972, p. 36). Correctly installed gates, however, at other locations (*e.g.*, Aitkin Cave, Pennsylvania) have led to increases in eastern small-footed bat populations (Butchkoski 2012, pers. comm.). An example of northern long-eared bats likely being affected occurred when John Friend Cave in Maryland was filled with large rocks in 1981, which closed the only known entrance to the cave (Gates *et al.* 1984, p. 166).

In addition to the direct access modifications to caves discussed above, debris buildup at entrances or on cave gates can also significantly modify the cave or mine site characteristics through restricting airflow, altering the temperature of hibernacula, and restricting water flow. Water flow restriction could lead to flooding, thus drowning hibernating bats (Amelon and Burhans 2006, p. 72; Hemberger 2011, unpublished data). In Minnesota, 5 of 11 known northern long-eared bat hibernacula are known to flood, presenting a threat to hibernating bats (Nordquist 2012, pers. comm.). In Massachusetts, one of the known hibernacula for northern long-eared bats is a now unused aqueduct that on very rare occasions may fill up with water and make the hibernaculum unusable (French 2012, unpublished data). Flooding has been noted in hibernacula in other States within the range of the northern long-eared bat, but to a lesser degree. Although modifications to hibernacula can lead to mortality of both species, it has not had population-level effects.

Mining operations, mine passage collapse (subsidence), and mine reclamation activities can also affect bats and their hibernacula. Internal and external collapse of abandoned coal mines was identified as one of the primary threats to eastern small-footed and northern long-eared bat hibernacula at sites located within the New River Gorge National River and Gauley River National Recreation Area in West Virginia (Graham 2011, unpublished data). Collapse of hibernacula entrances or areas within the hibernacula, as well as quarry and mining operations that may alter known hibernacula, are considered threats to northern long-eared bats within Kentucky (Hemberger 2011, unpublished data). In States surveyed for effects to northern long-eared bats by hibernacula collapse, responses varied, with the following number of hibernacula in each State reported as susceptible to collapse: 1 (of 7) in Maryland, 3 (of 11) in Minnesota, 1 (of 5) in New Hampshire, 4 (of 15) in North Carolina, 1 (of 2) in South Carolina, and 1 (of 13) in Vermont (Service 2011, unpublished data).

Before current cave protection laws, there were several reported instances where mines were closed while bats were hibernating and entombing entire colonies (Tuttle and Taylor 1998, p. 8). Several caves were historically sealed or mined in Maryland prior to cave protection laws, although bat populations were undocumented (Feller 2011, unpublished data). For both the eastern small-footed and northern long-eared bats, loss of potential winter habitat through mine closures has been noted as a concern in Virginia, although visual inspections of openings are typically conducted to determine whether gating is warranted (Reynolds 2011, unpublished data). In Nebraska, closing quarries, and specifically sealing quarries in Cass and Sapry Counties, is considered a potential threat to northern long-eared bats (Geluso 2011, unpublished data).

In general, threats to the integrity of bat hibernacula have decreased since the Indiana bat was listed as endangered in 1967, and since the implementation of Federal and State cave protection laws. Increasing awareness about the importance of cave and mine microclimates to hibernating bats and regulation under the Act have helped to alleviate the destruction or modification of hibernation habitat, at least where the Indiana bat is present (Service 2007, p. 74). The eastern small-footed bat and northern long-eared bat have likely benefitted from the protections given to the Indiana bat and its winter habitat, as both species' ranges overlap significantly with the Indiana bat's range.

Disturbance of Hibernating Bats

Human disturbance of hibernating bats has long been considered a threat to cave-hibernating bat species like the eastern small-footed and northern long-eared bats, and is discussed in detail in the Service's Indiana Bat Draft Recovery Plan (2007, pp. 80–85). The primary forms of human disturbance to hibernating bats result from cave commercialization (cave tours and other commercial uses of caves), recreational caving, vandalism, and research-related activities (Service 2007, p. 80). Arousal during hibernation causes the greatest amount of energy depletion in hibernating bats (Thomas *et al.* 1990, p. 477). Human disturbance at hibernacula, specifically non-tactile disturbance such as changes in light and sound, can cause bats to arouse more frequently, causing premature energy store depletion and starvation, as well as increased tactile disturbance of bats to other individuals (Thomas *et al.* 1995, p. 944; Speakman *et al.* 1991, p. 1103), leading to marked reductions in bat populations (Tuttle 1979, p. 3). Prior to the outbreak of WNS, Amelon and Burhans (2006, p. 73) indicated that "the widespread recreational use of caves and indirect or direct disturbance by humans during the hibernation period pose the greatest known threat to this species (northern long-eared bat)." Olson *et al.* (2011, p. 228), hypothesized that decreased visits by recreational users and researchers were related to an increase in the hibernating bat population (including northern long-eared bats) at Cadomin Cave in Alberta, Canada. Disturbance during hibernation could cause movements within or between caves (Beer 1955, p. 244).

Human disturbance is a potential threat at approximately half of the known eastern small-footed bat hibernacula in the States of Kentucky, Maryland, North Carolina, Vermont, and West Virginia (Service, unpublished data). Of the States in the northern long-eared bat's range that assessed the possibility of human disturbance at bat hibernacula, 93 percent (13 of 14) identified potential effects from human disturbance for at least 1 of the known hibernacula for this species in their state (Service, unpublished data). Eight of these 14 States (Arkansas, Kentucky, Maine, Minnesota, New Hampshire, North Carolina, South Carolina, and Vermont) indicated the potential for human disturbance at over 50 percent of the known hibernacula in that State. Nearly all States without WNS identified human disturbance as the primary threat to hibernating bats, and all others (including WNS-positive States) noted human disturbance as a secondary threat (WNS was predominantly the primary threat in these States) or of significant concern (Service, unpublished data).

The threat of commercial use of caves and mines during the hibernation period has decreased at many sites known to harbor Indiana bats, and we believe that this also applies to eastern small-footed and northern long-eared bats. However, effects from recreational caving are more difficult to assess. In addition to unintended effects of commercial and recreational caving, intentional killing of bats in caves by shooting, burning, and clubbing has been documented, although there are no data suggesting that eastern small-footed bats have been killed by these activities (Tuttle 1979, pp. 4, 8). Intentional killing of northern long-eared bats has been documented at a small percentage of hibernacula (*e.g.,* several cases of vandalism at hibernacula in Kentucky, one case of shooting disturbance in Maryland, one case of bat torching in Massachusetts where approximately 100 bats (northern long-eared bats and other species) were killed) (Service, unpublished data), but we do not have evidence that this is happening on a large enough scale to have population-level effects.

In summary, while there are isolated incidents of previous disturbance to both bat species due to recreational use of caves in both species, we conclude that there is no evidence suggesting that this threat in itself has led to population declines in either species.

Summer Habitat

Eastern small-footed bats roost in a variety of natural and manmade rock features, whereas northern long-eared bats roost predominantly in trees and to a lesser extent in manmade structures, as discussed in detail in the *Species Information* section above. We know of only one documented account where vandals were responsible for destroying a portion of an eastern small-footed bat roost located in Maryland (Feller 2011, unpublished data). More commonly, roost habitat for both the eastern small-footed bat and northern long-eared bat is at risk of modification or destruction. In Pennsylvania, for example, highway construction, commercial development, and several wind-energy projects may remove eastern small-footed bat roosting habitat (Librandi-Mumma 2011, pers. comm.). Some of the highest rates of development in the conterminous United States are occurring within the range of eastern small-footed and northern long-eared bats (Brown *et al.* 2005, p. 1856) and contribute to loss of forest habitat.

Wind-energy development is rapidly increasing throughout the eastern small-footed bat and northern long-eared bats' ranges, particularly in the States of New Hampshire, New York, Pennsylvania, and Massachusetts. As well, Iowa, Illinois, Minnesota, Oklahoma, and North Dakota are within the top 10 States for wind power capacity (in megawatts) (installed projects) in the United States (American Wind Energy Association 2012, p. 6). If projects are sited in forested habitats, effects from wind-energy development may include forest-clearings associated with turbine placement, road construction, turbine lay-down areas, transmission lines, and substations. In Maryland, wind power development has been proposed in areas with documented eastern small-footed bat and northern long-eared bat summer habitat (Feller 2011, unpublished data). In Pennsylvania, the majority of wind-energy projects are located in habitats characterized as mountain ridge-top, cliffs, steep slopes, or isolated hills with steep, often vertical sides (Mumma and Capouillez 2011, pp. 11–12). Eastern small-footed bats were confirmed through bat mist-net surveys at 7 of 34 proposed wind-energy project sites in Pennsylvania, and northern long-eared bats were confirmed at all 34 proposed wind project sites (Mumma and Capouillez 2011, pp. 62–63). See *Factor E. Other Natural or Manmade Factors Affecting Its Continued Existence* for a discussion on effects to bats from the operation of wind turbines.

Another activity that may modify or destroy eastern small-footed bat roosting habitat is mined-land reclamation, whereby rock habitats (*e.g.,* rock piles, cliffs, spoil piles) are removed from previously mined lands. The Office of Surface Mining Reclamation and Enforcement and its partners are responsible for reclaiming and restoring lands degraded by mining operations. Mining sites eligible for restoration are numerous in the States of Pennsylvania, Ohio, West Virginia, and Kentucky. Reclaiming these sites often involves the removal of exposed rock habitats that may be used as eastern small-footed bat roost habitat (Sanders 2011, pers. comm.). The number of potential roost sites that have been destroyed or that may be destroyed in the future and the potential effect of this destruction on eastern small-footed bat populations are largely unknown. Despite the potential negative effects of this activity, there are no data available suggesting a decrease in the number of eastern small-footed bats from mined-land reclamation activities. Since northern long-eared bats are not known to use exposed rock habitat for roost sites, mined-land reclamation does not affect this species.

Surface coal mining is also common in the central Appalachian region, which includes portions of Pennsylvania, West Virginia, Virginia, Kentucky, and Tennessee, and is one of the major drivers of land cover change in the region (Sayler 2008, unpaginated). Surface coal mining also may destroy forest habitat in parts of the Illinois Basin in southwest Indiana, western Kentucky, and Illinois (King

2013, pers. comm.). One major form of surface mining is mountaintop mining, which is widespread throughout eastern Kentucky, West Virginia, and southwestern Virginia (Palmer *et al.* 2010, p. 148). Mountaintop mining involves the clearing of upper elevation forests, stripping of topsoil, and use of explosives to break up rocks to access buried coal. The excess rock is sometimes pushed into adjacent valleys, where it buries existing streams (Palmer *et al.* 2010, p. 148). Hartman *et al.* (2005, p. 96) reported significant reductions in insect densities in streams affected with fill material, including lower densities of coleopterans, a primary food source of eastern small-footed and northern long-eared bats (Griffith and Gates 1985, p. 452; Johnson and Gates 2007, p. 319; Moosman *et al.* 2007, p. 355; Feldhamer *et al.* 2009, p. 45). The effect of mountaintop mining on eastern small-footed bat and northern long-eared bat populations is largely unknown.

The effect of forest removal related to the eastern small-footed bat is poorly understood. Forest management can influence the availability and characteristics of non-tree roost sites, such as those used by eastern small-footed bats, although the resulting effects on bats and bat populations are poorly known (Hayes and Loeb 2007, p. 215). Since eastern small-footed bats often forage in forests immediately surrounding roost sites, forest management may affect the quality of foraging habitat (Johnson *et al.* 2009, p. 5). Scientific evidence and anecdotal observations support the hypotheses that bats respond to prey availability, that prey availability is influenced by forest management, and that influences of forest management on prey populations affect bat populations (Hayes and Loeb 2007, p. 219). In addition, forest management activities that influence tree density directly alter the amount of vegetative clutter (*e.g.*, tree density) in an area. As a result, forest management can directly influence habitat suitability for bats through changes in the amount of vegetative clutter (Hayes and Loeb 2007, p. 217). Eastern small-footed bats are capable of foraging in cluttered forest interiors, but as discussed in the *Species Information* section above, they have also been found foraging in clearings, in strip mine areas, and over water. Johnson and Gates (2008, p. 459) suggest that a better understanding of the required spatial extent and structure of forest cover along ridgelines and rock outcrops, as well as additional foraging activity requirements, is needed to aid conservation efforts for the eastern small-footed bat.

Although there is still much to learn about the effects of forest removal on northern long-eared bats and their associated summer habitat, studies to date have found that the northern long-eared bat shows a varied degree of sensitivity to timber harvesting practices. Several studies (as discussed in the *Species Information* section above) have found that the species uses a wide range of tree species for roosting, suggesting that forest succession may play a larger role in roost selection (than tree species) (Silvis *et al.* 2012, p. 6). Studies have found that female bat roosts are more often (*i.e.*, greater than what would be expected from random chance) located in areas with partial harvesting than in random sites, which may be due to trees located in more open habitat receiving greater solar radiation and therefore speeding development of young (Menzel *et al.* 2002, p. 112; Perry and Thill 2007, pp. 224–225). In the Appalachians of West Virginia, diameter-limit harvests (70–90 year-old stands, with 30–40 percent of the basal area removed in the past 10 years) rather than intact forest was the habitat type most selected by northern long-eared bats (Owen *et al.* 2003, p. 356). Cryan *et al.* (2001, p. 49) found several northern long-eared bat roost areas in recently harvested (less than 5 years) stands in the Black Hills of South Dakota, although the largest colony (n=41) was found in a mature forest stand that had not been harvested in over 50 years. In intensively managed forests in the central Appalachians, Owen *et al.* (2002, p. 4) found roost availability was not a limiting factor for the northern long-eared bat, since bats often chose black locust and black cherry as roost trees, which were quite abundant since these trees often regenerate quickly after disturbance (*e.g.*, timber harvest).

It is possible that this flexibility in roosting habits allows northern long-eared bats to be adaptable in managed forests, which allows them to avoid competition for roosting habitat with more specialized species, such as the Indiana bat (Timpone *et al.* 2010, p. 121). However, the northern long-eared bat has shown a preference for contiguous tracts of forest cover for foraging (Owen *et al.* 2003, p. 356; Yates and Muzika 2006, p. 1245). Jung *et al.* (2004, p. 333) found that it is important to retain snags and provide for recruitment of roost trees during selective harvesting in forest stands that harbor bats. If roost networks are disturbed through timber harvesting, there may be more dispersal and fewer shared roost trees, which may lead to less communication between bats in addition to less disease transmission (Johnson *et al.* 2012, p. 230). In the Appalachians, Ford *et al.* (2006, p. 20) assessed that northern long-eared bats may be a suitable management indicator species for assessing mature forest ecosystem integrity, since they found male bats using roosts in mature forest stands of mostly second growth or regenerated forests.

There is conflicting information on sensitivities of male versus female northern long-eared bats to forestry practices and resulting fragmentation. In Arkansas, Perry and Thill (2007, p. 225) found that male northern long-eared bats seem to prefer more dense stands for summer roosting, with 67 percent of male roosts occurring in unharvested sites versus 45 percent of female roosts. The greater tendency of females to roost in more open forested areas than males may be due to greater solar radiation experienced in these openings, which could speed growth of young in maternity colonies (Perry and Thill 2007, p. 224). Lacki and Schwierjohann (2001, p. 487) stated that silvicultural practices could meet both male and female roosting requirements by maintaining large-diameter snags, while allowing for regeneration of forests. However, Broders and Forbes (2004, p. 608) found that timber harvest may have negative effects on female bats since they use forest interiors at small scales (less than 2 km (1.2 mi) from roost sites). They also found that males are not as limited in roost selection and they do not have the energetic cost of raising young; therefore males may be less affected than females (Broders and Forbes 2004, p. 608). Henderson *et al.* (2008, p. 1825) also found that forest fragmentation effects northern long-eared bats at different scales based on sex; females require a larger unfragmented area with a large number of suitable roost trees to support a colony, whereas males are able to use smaller areas (more fragmented). Henderson and Broders (2008, pp. 959–960) examined how female northern long-eared bats use the forest-agricultural landscape on Prince Edward Island, Canada, and found that bats were limited in their mobility and activities are constrained where suitable forest is limited. However, they also found that bats in relatively fragmented areas used a building for colony roosting, which suggests an alternative for a colony to persist in an area with fewer available roost trees. Although we are still learning about the effect of forest removal on northern long-eared bats and their associated summer habitat, studies to date have found that the northern long-eared bat shows a varied degree of sensitivity to timber harvesting practices and the amount of forest removal occurring varies by State.

Natural gas development from shale is expanding across the United States, particularly throughout the range of the northern long-eared and eastern small-footed bat. Natural gas extraction involves fracturing rock formations and uses highly pressurized fluids consisting of water and various chemicals to do so (Hein 2012, p. 1). Natural gas extraction, particularly across the Marcellus Shale region, which includes large portions of New York, Pennsylvania, Ohio, and West Virginia, is expected to expand over the coming years. In Pennsylvania, for example, nearly 2,000 Marcellus natural gas wells have already been drilled or permitted, and as many as 60,000 more could be built by 2030, if development trends continue (Johnson 2010, pp. 8, 13). Habitat loss and degradation due to this practice could occur in the form of forest clearing for well pads and associated infrastructure (*e.g.*, roads, pipelines, and water impoundments), which would decrease the amount of suitable interior forest habitat available to northern long-eared and eastern small-footed bats for establishing maternity colonies and for foraging, in addition to further isolating populations and, therefore, potentially decreasing genetic diversity (Johnson 2010, p. 10; Hein 2012, p. 6). Since northern long-eared bats and eastern small-footed bats have philopatric tendencies, loss or alteration of forest habitat for natural gas development may also put additional stress on females when returning to summer roost or foraging areas after hibernation if females were forced to find new roosting or foraging areas (expend additional energy) (Hein 2012, pp. 11–12).

Conservation Efforts To Reduce Habitat Destruction, Modification, or Curtailment of Its Range

Although there are various forms of habitat destruction and disturbance that present potential adverse effects to the northern long-eared bat, this is not considered the predominant threat to the species. Even if all habitat-related stressors were eliminated or minimized, the significant effects of WNS on the northern long-eared bat would still be present. Therefore, below we present a few examples, but not a comprehensive list, of conservation efforts that have been undertaken to lessen effects from habitat destruction or disturbance to northern long-eared and eastern small-footed bats. One of the threats to bats in Michigan is the closure of unsafe mines in such a way that bats are trapped within or excluded; however, there have been efforts by the Michigan Department of Natural Resources and others to work with landowners who have open mines to encourage them to install bat-friendly gates to close mines to humans, but allow access to bats (Hoving 2011, unpublished data). The NPS has proactively taken efforts to minimize effects to bat habitat resulting from vandalism, recreational activities, and abandoned mine closures (Plumb and Budde 2011, unpublished data). In addition, the NPS is properly gating, using a "bat-friendly design, abandoned coal mine entrances as funding permits (Graham 2011, unpublished data). All known hibernacula within national grasslands and forestlands of the Rocky Mountain Region of the U.S. Forest Service are closed during the winter hibernation period, primarily due to the threat of white-nose syndrome, although this will reduce disturbance to bats in general inhabiting these hibernacula (U.S. Forest Service 2013, unpaginated). Concern over the importance of bat roosts, including hibernacula, fueled efforts by the American Society of Mammalogists to develop guidelines for protection of roosts, many of which have been adopted by government agencies and special interest groups (Sheffield *et al.* 1992, p. 707).

Summary of the Present or Threatened Destruction, Modification, or Curtailment of Its Habitat or Range

We have identified several activities, such as constructing physical barriers at cave accesses, mining, flooding, vandalism, development, and timber harvest, that may modify or destroy habitat for the eastern small-footed bat and northern long-eared bat. Although such activities occur, these activities alone do not have significant, population-level effects on either species.

Factor B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

There are very few records of either species being collected specifically for commercial, recreational, scientific, or educational purposes, and thus we do not consider such collection activities to pose a threat to either species. Disturbance of hibernating bats as a result of recreational use and scientific research activities in hibernacula is discussed under Factor A.

Factor C. Disease or Predation

Disease

White-Nose Syndrome

White-nose syndrome is an emerging infectious disease responsible for unprecedented mortality in some hibernating insectivorous bats of the northeastern United States (Blehert *et al.* 2009, p. 227), and poses a considerable threat to several hibernating bat species throughout North America (Service 2010, p. 1). Since its first documented appearance in New York in 2006, WNS has spread rapidly throughout the Northeast and is expanding through the Midwest. As of August 2013, WNS has been confirmed in 22 States (Alabama, Connecticut, Delaware, Georgia, Illinois, Indiana, Kentucky, Maine, Maryland, Massachusetts, Missouri, New Hampshire, New Jersey, New York, North Carolina, Ohio, Pennsylvania, South Carolina, Tennessee, Vermont, Virginia, and West Virginia) and 5 Canadian provinces (New Brunswick, Nova Scotia, Ontario, Prince Edward Island, and Quebec). Four additional States (Arkansas, Iowa, Minnesota, and Oklahoma) are considered suspect for WNS based on the detection of the causative fungus on bats within those States, but with no associated disease to date. Service biologists and partners estimate that at least 5.7 million to 6.7 million bats of several species have now died from WNS (Service 2012, p. 1). Dzal *et al.* (2011, p. 393) documented a 78-percent decline in the summer activity of little brown bats in New York State, coinciding with the arrival and spread of WNS, suggesting large-scale population effects. Turner *et al.* (2011, p. 22) reported an 88-percent decline in the number of hibernating bats at 42 sites from the States of New York, Pennsylvania, Vermont, Virginia, and West Virginia. Furthermore, Frick *et al.* (2010, p. 681) predicted that the little brown bat, formerly the most common bat in the northeastern United States, will likely become extinct in the region by 2026 (potential loss of some 6.5 million bats) if current trends continue. Similarly, Thogmartin *et al.* (2013, p. 171) predicted that WNS is likely to extirpate the federally endangered Indiana bat over large parts of its range. These predicted trends in little brown bats and Indiana bats may or may not also be indicative of population trends in other bat species like the eastern small-footed and northern long-eared bats.

The first evidence of WNS was documented in a photograph taken from Howes Cavern, 52 km (32 mi) west of

Albany, New York, on February16, 2006 (Blehert *et al.* 2009, p. 227). Prior to the arrival of WNS, surveys of six species of hibernating bats in New York State revealed that populations had been stable or increasing in recent decades (Service 2010, p. 1). Decreases in some species of bats at WNS-infected hibernacula have ranged from 30 to 99 percent (Frick *et al.* 2010, p. 680).

The pattern of spread has generally followed predictable trajectories along recognized migratory pathways and overlapping summer ranges of hibernating bat species. Therefore, Kunz and Reichard (2010, p. 12) assert that WNS is spread mainly through bat-to-bat contact; however, evidence suggests that fungal spores can be transmitted by humans (United States Geologic Survey (USGS) National Wildlife Health Center, Wildlife Health Bulletin 2011–05), and bats can also become infected by coming into contact with contaminated cave substrate (Darling 2012, pers. comm.). Six North American hibernating bat species (little brown bat, Indiana bat, northern long-eared bat, eastern small-footed bat, big brown bat, and tri-colored bat), are known to be affected by WNS; however, the effect of WNS varies by species. The fungus that causes WNS has been detected on three additional species; the southeastern bat (*Myotis austroriparius*), and gray bat (*Myotis grisescens*), and cave bat (*Myotis velifer*). White-nose syndrome is caused by the recently described psychrophilic (cold-loving) fungus, currently known as *Geomyces destructans*. *Geomyces destructans* may be nonnative to North America, and only recently arrived on the continent (Puechmaille *et al.* 2011, p. 8). The fungus grows on and within exposed tissues of hibernating bats (Lorch *et al.* 2011, p. 376; Gargas *et al.* 2009, pp. 147–154)), and the diagnostic feature is the white fungal growth on muzzles, ears, or wing membranes of affected bats, along with epidermal (skin) erosions that are filled with fungal hyphae (branching, filamentous structures of fungi) (Blehert *et al.* 2009, p. 227; Meteyer 2009, p. 412). *Geomyces destructans* grows optimally at temperatures from 5 to 10 °C (41 to 50 °F), the same temperatures at which bats typically hibernate (Blehert *et al.* 2009, p. 227). Temperatures in WNS-affected hibernacula seasonally range from 2 to 14 °C (36 to 57 °F), permitting year-round growth, and may act as a reservoir maintaining the fungus (Blehert *et al.* 2009, p. 227). Growth is slow, and no growth occurs at temperatures above 24 °C (75 °F) (Gargas *et al.* 2009, p. 152). Bats that are found in more humid regions of hibernacula may be more susceptible to WNS, but further research is needed to confirm this hypothesis. Declines in Indiana bats have been greater under more humid conditions, suggesting that growth of the fungus and either intensity or prevalence of infections are higher in more humid conditions (Langwig *et al.* 2012a, p. 1055). Although *G. destructans* has been isolated from five bat species from Europe, research suggests that bat species in Europe may be immunologically or behaviorally resistant, having coevolved with the fungus (Wibbelt *et al.* 2010, p. 1241). Pikula *et al.* (2012, p. 210), however, confirmed that bats found dead in the Czech Republic exhibited lesions consistent with WNS infection.

In addition to the presence of the white fungus, initial observations showed that bats affected by WNS were characterized by some or all of the following: (1) Depleted fat reserves by mid-winter; (2) a general unresponsiveness to human disturbance; (3) an apparent lack of immune response during hibernation; (4) ulcerated, necrotic, and scarred wing membranes; and (5) aberrant behaviors, including shifts of large numbers of bats in hibernacula to roosts near the entrances or unusually cold areas, large numbers of bats dispersing during the day from hibernacula during mid-winter, and large numbers of fatalities, either inside the hibernacula, near the entrance, or in the immediate vicinity of the entrance (WNS Science Strategy Report 2008, p. 2; Service 2010, p. 2). Although the exact process by which WNS leads to death remains undetermined, it is likely that the immune function during torpor compromises the ability of hibernating bats to combat the infection (Bouma *et al.* 2010, p. 623; Moore *et al.* 2011, p. 10).

Early hypotheses suggested that WNS may affect bats before the hibernation season begins, causing bats to arrive at hibernacula with insufficient fat to survive the winter. Alternatively, a second hypothesis suggests that bats arrive at hibernacula unaffected and enter hibernation with sufficient fat stores, but then become affected and use fat stores too quickly as a result of disruption to hibernation physiology (WNS Science Strategy Group 2008, p. 7). More recent observations, however, suggest that bats are arriving to hibernacula with sufficient or only slightly lower fat stores (Turner 2011, pers. comm.), and that although body weights of WNS-infected bats were consistently at the lower end of the normal range, in one study 12 of 14 bats (10 little brown bats, 1 big-brown bat, and 1 tri-colored bat) had an appreciable degree of fat stores (Courtin *et al.* 2010, p. 4).

Boyles and Willis (2010, pp. 92–98) hypothesized that infection by *Geomyces destructans* alters the normal arousal cycles of hibernating bats, particularly by increasing arousal frequency, duration, or both. In fact, Reeder *et al.* (2012, p. 5) and Warnecke *et al.* (2012, p. 2) did observe an increase in arousal frequency in laboratory studies of hibernating bats infected with *G. destructans.* A disruption of this torpor-arousal cycle could easily cause bats to metabolize fat reserves too quickly, thereby leading to starvation. For example, skin irritation from the fungus might cause bats to remain out of torpor for longer than normal to groom, thereby exhausting their fat reserves prematurely (Boyles and Willis 2010, p. 93).

Due to the unique physiological importance of wings to hibernating bats in relation to the damage caused by *Geomyces destructans,* Cryan *et al.* (2010, pp. 1–8) suggests that mortality may be caused by catastrophic disruption of wing-dependent physiological functions. The authors hypothesize that *G. destructans* may cause unsustainable dehydration in water-dependent bats, trigger thirst-associated arousals, cause significant circulatory and thermoregulatory disturbance, disrupt respiratory gas exchange, and destroy wing structures necessary for flight control (Cryan *et al.* 2010, p. 7). The wings of winter-collected WNS-affected bats often reveal signs of infection, whereby the degree of damage observed suggests functional impairment. Emaciation is a common finding in bats that have died from WNS (Cryan *et al.* 2010, p. 3). Cryan *et al.* (2010, p. 3) hypothesized that disruption of physiological homeostasis, potentially caused by *G. destructans* infection, may be sufficient to result in emaciation and mortality. The authors hypothesized that wing damage caused by *G. destructans* infections could sufficiently disrupt water balance to trigger frequent thirst-associated arousals with excessive winter flight, and subsequent premature depletion of fat stores. In related research, Cryan *et al.* (2013, p. 398) found, after analyzing blood from hibernating bats infected with WNS, that electrolytes, sodium and chloride, tended to decrease as wing damage increased in severity. Proper concentrations of electrolytes are necessary for maintaining physiologic homeostasis, and any imbalance could be life-threatening (Cryan *et al.* 2013, p. 398). Although the exact mechanism by which WNS affects bats is still in question, the effect it has on many hibernating bat species is well documented as well as the high levels of mortality it causes in some susceptible bat species.

Effects of White-Nose Syndrome on the Eastern Small-Footed Bat

Eastern small-footed bats are known to be susceptible to WNS. As of 2011, of the 283 documented eastern small-footed bat hibernacula, 86 (31 percent) were WNS-positive (Service 2011, unpublished data). Only three eastern small-footed bats have been collected, tested, and confirmed positive for WNS by histology: One bat collected and euthanized from New York in 2009, one bat found dead in Pennsylvania in 2011, and one bat found dead from South Carolina in 2013 (Ballmann 2011, pers. comm.; Last 2013a, pers. comm.). An additional eastern small-footed bat collected in winter 2011–2012 from the Mammoth Cave Visitor Center in Kentucky, was submitted to the Southeastern Cooperative Wildlife Disease Study; however, this bat tested negative for WNS. Biologists also observed approximately five dead eastern small-footed bats with obvious signs of fungal infection in Virginia (Reynolds 2011, pers. comm.).

To determine whether WNS is causing a population-level effect to eastern small-footed bats, the Service began by reviewing winter hibernacula survey data. By comparing the most recent pre-WNS count to the most recent post-WNS count, Turner *et al.* (2011, p. 22) reported a 12-percent decline in the number of hibernating eastern small-footed bats at 25 hibernacula in New York, Pennsylvania, Vermont, Virginia, and West Virginia. Data analyzed in this study were limited to sites with confirmed WNS mortality for at least 2 years and sites with comparable survey effort across pre- and post-WNS years. Based on a review of pre-WNS hibernacula count data over multiple years at 12 of these sites, the number of eastern small-footed bats fluctuated between years.

When we compared the most recent post-WNS eastern small-footed bat count to pre-WNS observations, we found that post-WNS counts were within the normal observed range at nine sites (75 percent), higher at two sites (17 percent), and lower at only one site (8 percent). In addition, although Langwig *et al.* (2012a, p. 1052) reported a significantly lower population growth rate compared to pre-WNS population growth rates for eastern small-footed bat, they found that the species was not declining significantly at hibernacula in New York, Vermont, Connecticut, and Massachusetts. Langwig *et al.* (2012b, p. 15) also observed lower prevalence of *Geomyces destructans* on eastern small-footed bat wing and muzzle tissue during late hibernation, compared to other bat species (*e.g.,* little brown bats). Lastly, biologists did not observe fungal growth (although the fungus may not be visible after the first couple of years) on eastern small-footed bats during 2013 hibernacula surveys in New York, Pennsylvania, and North Carolina, even though it was observed on other bat species (*e.g.,* little brown bats) within the same sites (although a few, not all, eastern small-footed bats viewed under ultraviolet light did show signs of mild infections), nor did they observe reduced numbers of eastern small-footed bats compared to pre-WNS years (Graeter 2013, pers. comm.; Herzog 2013, pers. comm.; Turner 2013, unpublished data). In fact, biologists in New York observed the largest number of hibernating eastern small-footed bats ever reported (2,383) during surveys conducted in 2013, up from 1,727 reported in 1993 using roughly comparable survey effort (Herzog 2013, pers. comm.). In summary, WNS does not appear to have caused a significant population decline in hibernating eastern small-footed bats.

Summer survey data are limited for the eastern small-footed bat. We know of only three studies that have attempted to quantify changes in the number of non-hibernating eastern small-footed bats since the spread of WNS (Francl *et al.* 2012; Nagel and Gates 2012; Moosman *et al.* in press). At one study location, Surry Mountain Reservoir, New Hampshire, bats were mist-netted over multiple years before and after the emergence of WNS (Moosman *et al.* in press). Researchers observed a significant decline in the relative abundance of eastern small-footed bats between 2005 and 2011, based on reductions in capture rates. However, they found that the probability of capturing greater than or equal to one eastern small-footed bat on any given visit during the 7 years of study was similar across years, although the probability of capturing other species (*e.g.,* northern long-eared and little brown bats) declined over time. Moosman *et al.* (unpublished data) also noted that the observed decline in relative abundance of eastern small-footed bats at their site should not be solely attributed to WNS because of the potential for bats to become trap-shy due to repeated sampling efforts.

Eastern small-footed bats are noted for their ability to detect and avoid mist-nets, perhaps more so than other bat species within their range (Tyburec 2012, unpaginated). In addition, Francl *et al.* (2012, p. 34) compared bat mist-net data collected from 31 counties in West Virginia prior to the detection of WNS (1997 to 2008) to 8 West Virginia and 1 extreme southwestern Pennsylvania counties surveyed in 2010. Researchers reported a 16-percent decline in the post-WNS capture rate for eastern small-footed bats, although they acknowledge the small sample size may have inherently higher variation and bias compared to more common species that showed consistently negative trends (*e.g.,* northern long-eared, little brown, and tri-colored bats) (Francl *et al.* 2012, p. 40). Lastly, during acoustic surveys for bats, Nagel and Gates (2012, p. 5) reported a 63-percent increase in the number of eastern small-footed bat passes during acoustic surveys from 2010 to 2012 in western Maryland, although large declines in bat passes were observed for other species (*e.g.,* northern long-eared, little brown/Indiana, and tri-colored bats).

Several factors may influence why eastern small-footed bats are potentially less susceptible to WNS than other *Myotis* bats. First, during mild winters, eastern small-footed bats may not enter caves and mines or, if they do, may leave during mild periods. Although there are few winter observations of this species outside of cave and mine habitat, it was first speculated in 1945 as a possibility. In trying to explain why so many bats banded in the summer were unaccounted for during winter hibernacula surveys, Griffin (1945, p. 22) suggested that bats may be using alternate hibernacula such as small, deep crevices in rocks, which he suggested would provide a bat with adequate protection from freezing. Neubaum *et al.* (2006, p. 476) observed many big brown bats choosing hibernation sites in rock crevices and speculated that this pattern of roost selection could be common for other species. Time spent outside of cave and mine habitat by eastern small-footed bats means less time for the fungus to grow because environmental conditions (*e.g.,* temperature and humidity) are suboptimal for fungus growth.

A second factor that may influence lower susceptibility of eastern small-footed bats to WNS is that this bat species tends to enter cave or mine habitat later (mid-November) and leave earlier (mid-March) compared to other *Myotis* bats, again providing less time for the fungus to grow, and less energy expenditure than other species that hibernate longer. Third, when eastern small-footed bats are present at caves and mines, they are most frequently observed at the entrances, where humidity is low and temperature fluctuations are high, which consequently does not provide ideal environmental conditions for fungal growth. Cryan *et al.* (2010, p. 4) suggest that eastern small-footed bats may be less susceptible to evaporative water loss, since they often select drier areas of hibernacula, and therefore may be less susceptible to succumbing to WNS. Big brown bats also tend to select drier, more ventilated areas for hibernation, and consequently, Blehert *et al.* (2009, p. 227) and Courtin *et al.* (2010, p. 4) did not observe the fungus in big brown bat specimens. Lastly, unlike some other gregarious bats (*e.g.,* little brown bats), eastern small-footed bats frequently roost solitarily or deep within cracks, possibly further reducing their exposure to the fungus.

Fenton (1972, p. 5) never observed eastern small-footed bats close to or in contact with little brown or Indiana bats, both highly gregarious species experiencing severe population declines. Solitary hibernating habits have also been suggested as one of the reasons why big brown bats appear to have been only moderately affected by WNS (Ford *et al.* 2011, p. 130). Laboratory studies conducted by Blehert *et al.* (2011) further support this hypothesis. In their study, only healthy bats that came into direct contact with infected bats or were inoculated with pure cultures of *Geomyces destructans* developed lesions consistent with WNS. Healthy bats housed with infected bats in such a way as to prohibit animal-to-animal contact but still allow for potential aerosols to be transmitted from sick bats did not develop any detectable signs of WNS.

In conclusion, there are several factors that may explain why eastern small-footed bats appear to be less susceptible to WNS than other cave bat species. These factors include hibernacula selection (cave versus non-cave), total time spent hibernating in hibernacula, location within the hibernacula (areas with lower humidity and higher temperature fluctuation), and solitary roosting behavior.

Effects of White-Nose Syndrome on the Northern Long-Eared Bat

The northern long-eared bat is known to be susceptible to WNS, and mortalities due to the disease have been confirmed. The USGS National Wildlife Health Center in Madison, Wisconsin, received 79 northern long-eared bat submissions since 2007, of which 65 were tested for WNS. Twenty-eight of the 65 northern long-eared bats tested were confirmed as positive for WNS by histopathology and another 10 were suspect (Ballmann 2013, pers. comm.). In addition, 9 of 14 northern long-eared bats in 2012–2013 were positive, and 1 was suspect (Last 2013b, pers. comm.); all the WNS-positive submissions were from Tennessee, Kentucky, and Ohio. The New York Department of Environmental Conservation has confirmed 29 northern long-eared bats submitted with signs of WNS, at minimum (there are still bat carcasses that have not been analyzed yet), since 2007 in New York (Okonieski 2012, pers. comm.).

Due to WNS, the northern long-eared bat has experienced a sharp decline in the northeastern part of its range, as evidenced in hibernacula surveys. The northeastern United States is very close to saturation (WNS found in majority of hibernacula) for the disease, with the northern long-eared bat being one of the species most severely affected by the disease (Herzog and Reynolds 2012, p. 10). Turner *et al.* (2011, p. 22) compared the most recent pre-WNS count to the most recent post-WNS count for 6 cave bat species; they reported a 98-percent decline between pre- and post-WNS in the number of hibernating northern long-eared bats at 30 hibernacula in New York, Pennsylvania, Vermont, Virginia, and West Virginia. Data analyzed in this study were limited to sites with confirmed WNS mortality for at least 2 years and sites with comparable survey effort across pre and post-WNS years. In addition to the Turner *et al.* (2011) data, the Service conducted an additional analysis that included data from Connecticut (n=3), Massachusetts (n=4), and New Hampshire (n=4), and added one additional site to the previous Vermont data. We used a similar protocol for analyses as used in Turner *et al.* (2011); our analysis was limited to sites where WNS has been present for at least 2 years. The combined overall rate of decline seen in hibernacula count data for the 8 States is approximately 99 percent.

In hibernacula surveys in New York, Vermont, Connecticut, and Massachusetts, hibernacula with larger populations of northern long-eared bats experienced greater declines, suggesting a density-dependent decline due to WNS (Langwig *et al.* 2012a, p. 1053). Also, although some species' populations (*e.g.,* tri-colored bat, Indiana bat) stabilized at drastically reduced levels compared to pre-WNS, each of the 14 populations of northern long-eared bats became locally extinct within 2 years due to disease, and no population was remaining 5 years post-WNS (Langwig *et al.* 2012, p. 1054). During 2013 hibernacula surveys at 34 sites where northern long-eared bats were also observed prior to WNS in Pennsylvania, researchers found a 99-percent decline (from 637 to 5 bats) (Turner 2013, unpublished data).

Due to favoring small cracks or crevices in cave ceilings, making them more challenging to locate during hibernacula surveys, data in some States (particularly those with a greater number of caves with more cracks or crevices) may not give an entirely clear picture of the level of decline the species is experiencing (Turner *et al.* 2011, p. 21). When dramatic declines due to WNS occur, the overall rate of decline appears to vary by site; some sites experience the progression from the detection of a few bats with visible fungus to widespread mortality after a few weeks, while at other sites this may take a year or more (Turner *et al.* 2011, pp. 20–21). For example, in Massachusetts, WNS was first confirmed in February of 2008, and by 2009, "the population (northern long-eared bat) was knocked down, and the second year the population was finished" (French 2012, pers. comm.). Further, in Virginia, Reynolds (2012, pers. comm.) reported that "not all sites are on the same 'WNS time frame,' but it appears the effects will be similar, suggesting that all hibernacula in the mountains of Virginia will succumb to WNS at one time or another." We have not yet seen the same level of decline in the Midwestern and southern parts of the species' range, although we expect similar rates of decline once the disease arrives or becomes more established.

Although the disease has not yet spread throughout the species' entire range (WNS is currently found in 22 of 39 States where the northern long-eared bat occurs), it continues to spread, and we have no reason not to expect that where it spreads, it will have the same impact to the affected species (Coleman 2013, pers. comm.). The current rate of spread has been rapid, spreading from the first documented occurrence in New York in February 2006, to 22 states and 5 Canadian provinces by July 2013. There is some uncertainty as to the timeframe when the disease will spread throughout the species' range and when resulting mortalities as witnessed in the currently affected area will occur in the rest of the range. Researchers have suggested that there may be a 'slow down' in the spread of the disease in the Great Plains (Frick and Kilpatrick 2013, pers. comm.); however, this is on the western edge of the northern long-eared bat's range where the species is naturally less common and, therefore, offers little respite to the species. A few models have attempted to project the spread of *Geomyces destructans* and WNS, and although they have differed in the timing of the disease spreading throughout the continental United States, all were in agreement that WNS will indeed spread throughout the United States (Hallam *et al.* 2011, p. 8; Maher *et al.* 2012, pp. 4–5). One of these models suggests that there may be a temperature-dependent boundary in southern latitudes that may offer refuge to WNS-susceptible bats. However, this would likely provide little relief to the northern long-eared bat, since the species' range only slightly enters these southern states (Hallam *et al.* 2011, pp. 9–11). In addition, human transmission could introduce the spread of the fungus to new locations that are far removed from the current known locations (e.g., spread the fungus farther than an infected bat could transmit it within their natural movement patterns) (Coleman 2013, pers. comm.).

Long-term (including pre- and post-WNS) summer data for the northern long-eared bat are somewhat limited; however, the available data parallel the population decline exhibited in hibernacula surveys. Summer data can corroborate and confirm the decline to the species seen in hibernacula data. Summer surveys from 2005–2011 near Surry Mountain Lake in New Hampshire showed a 99-percent decline in capture success of northern long-eared bats post-WNS, which is similar to the hibernacula data for the State (a 95-percent decline) (Brunkhurst 2012, unpublished data).

The northern long-eared bat is becoming less common on the Vermont landscape as well. Pre-WNS, the species was the second most common bat species in the State; however, it is now one of the least likely to be encountered, with the change in effort to capture one bat increasing by nearly 13 times, and approximately a 94-percent overall reduction in captures in mist-net surveys (Darling and Smith 2011, unpublished data). In eastern New York, captures of northern long-eared bats have declined dramatically, approximately 93 percent, for the species from pre-WNS (Herzog 2012, unpublished data). Prior to discovery of WNS in West Virginia, northern long-eared bat mist-net captures comprised 41 percent of all captures and 24 percent post-WNS (2010) and at a rate of 23 percent of historical rates (Francl *et al.* 2012, pp. 35–36). In addition, pregnancy peaked more than 2 weeks earlier post-WNS than pre-WNS (May 20 versus June 7, respectively) and the proportion of juveniles declined by more than half in mid-August; it is unclear if this change will have population-level effects on the species at this time (Francl *et al.* 2012, p. 36). Ford *et al.* (2011, p. 127) conducted summer acoustic surveys on Fort Drum, New York, from 2003–2010, including pre-WNS (2003–2008) and post-WNS (2008–2010). Although activity still rose from early summer to late summer for northern long-eared bats, the overall activity levels for the species declined from pre- to post-WNS (Ford *et al.* 2011, pp. 129–130). Similarly, Nagel and Gates (2012, p. 5) reported a 78-percent decrease in northern long-eared bat passes (as compared to a 63-percent increase in the number of eastern small-footed bats mentioned above) during acoustic surveys between 2010 and 2012 in western Maryland. "Due to the greatest recorded decline in regional hibernacula counts (Turner *et al.* 2011), the northern long-eared bat is of particular concern (to researchers in Pennsylvania)" (Turner 2013, unpublished data). Therefore, researchers in Pennsylvania selected two sites to study in 2010 and 2011, where pre-WNS swarm trapping had previously been conducted. The capture rates at the first site declined by 95 percent and at the second site by 97 percent, which corroborates documented interior hibernacula declines (Turner 2013 unpublished data; Turner *et al.* 2011, p. 18).

Although northern long-eared bats are known to awaken from a state of torpor sporadically throughout the winter and move between hibernacula (Griffin 1940, p. 185; Whitaker and Rissler 1992b, p. 131; Caceres and Barclay 2000 pp. 2–3), they have not been observed roosting regularly outside of caves and mines during the winter, as species that are less susceptible to WNS (*e.g.*, big brown bat) have. Northern long-eared bats may be more susceptible to evaporative water loss (and therefore more susceptible to WNS) due to their propensity to roost in the most humid parts of the hibernacula (Cryan *et al.* 2010, p. 4). As described in the *Hibernation* section above, northern long-eared bats roost in areas within hibernacula that have higher humidity, possibly leading to higher rates of infection, as Langwig *et al.* (2012a, p. 1055) found with Indiana bats. Also, northern long-eared bats prefer cooler temperatures within hibernacula: 0 to 9 °C (32 to 48 °F) (Raesly and Gates 1987, p. 18; Caceres and Pybus 1997, p. 2; Brack 2007, p. 744), which are within the optimal growth limits of *Gyomyces destructans* (5 to 10 °C (41 to 50 °F)) (Blehert *et al.* 2009, p. 227).

The northern long-eared bat may also spend more time in hibernacula than other species that are less susceptible (*e.g.*, eastern small-footed bat (see *Effects of White-nose Syndrome on the Eastern Small-footed Bat* section, above)), which allows more time for the fungus to infect bats and grow; northern long-eared bats enter the cave or mine in October or November (although they may enter as early as August) and leave the hibernaculum in March or April (Caire *et al.* 1979, p. 405; Whitaker and Hamilton 1998, p. 100; Amelon and Burhans 2006, p. 72). Furthermore, the northern long-eared bat occasionally roosts in clusters or in the same hibernacula as other bat species that are also susceptible to WNS (see *Hibernation* section, above); therefore, northern long-eared bats may have increased susceptibility to bat-to-bat transmission of WNS.

Given the observed dramatic population declines attributed to WNS, as described above, we are greatly concerned about this species' persistence where WNS has already spread. The area currently affected by WNS constitutes the core of the northern long-eared bat's range, where the species was most common prior to WNS; the species is less common in the southern and western parts of its range and is considered to be rare in the northwestern part of its range (Caceres and Barclay 2000, p. 2; Harvey 1992, p. 35), the areas where WNS has not yet been detected. Furthermore, the rate at which WNS has spread has been rapid; it was first detected in New York in 2006, and has spread west at least as far as Illinois and Missouri, south as far as Georgia and South Carolina, and north as far as southern Quebec and Ontario as of 2013. Although this spread rate may slow or have reduced effects in the more southern and western parts of the species' range (Frick and Kilpatrick 2013, pers. comm.), general agreement is that WNS will indeed spread throughout the United States (Hallam *et al.* 2011, p. 8; Maher *et al.* 2012, pp. 4–5). WNS has already had a substantial effect on northern long-eared bats in the core of its range and is likely to spread throughout the species' entire range within a short time; thus we consider it to be the predominant threat to the species rangewide.

Other Diseases

Infectious diseases observed in North American bat populations include rabies, histoplasmosis, St. Louis encephalitis, and Venezuelan equine encephalitis (Burek 2001, p. 519; Rupprecht *et al.* 2001, p. 14; Yuill and Seymour 2001, pp. 100, 108). Rabies is the most studied disease of bats, and can lead to mortality, although antibody evidence suggests that some bats may recover from the disease (Messenger *et al.* 2003, p. 645) and retain immunological memory to respond to subsequent exposures (Turmelle *et al.* 2010, p. 2364). Bats are hosts of rabies in North America (Rupprecht *et al.* 2001, p. 14), accounting for 24 percent of all wild animal cases reported during 2009 (Centers for Disease Control and Prevention 2011). Although rabies is detected in up to 25 percent of bats submitted to diagnostic labs for testing, less than 1 percent of bats sampled randomly from wild populations test positive for the virus (Messenger *et al.* 2002, p. 741). Eastern small-footed and northern long-eared bats are among the species reported positive for rabies virus infection (Constantine 1979, p. 347; Burnett 1989, p. 12; Main 1979, p. 458); however, rabies is not known to have appreciable effects to either species.

Histoplasmosis has not been associated with eastern small-footed bats or northern long-eared bats and may be limited in these species compared to other bats that form larger aggregations with greater exposure to guano-rich substrate (Hoff and Bigler 1981, p. 192). St. Louis encephalitis antibody and high concentrations of Venezuelan equine encephalitis virus have been observed in big brown bats and little brown bats (Yuill and Seymour 2001, pp. 100, 108), although data are lacking on the prevalence of these viruses in eastern small-footed bats. Eastern equine encephalitis has been detected in northern long-eared bats (Main 1979, p. 459), although no known population declines have been found due to presence of the virus. Northern long-eared bats are also known to carry a variety of pests including chiggers, mites, bat bugs, and internal helminthes (Caceres and Barclay 2000, p. 3). None of these diseases or pests, however, has caused the record level of bat mortality like that observed since the emergence of WNS.

Predation

Typically, animals such as owls, hawks, raccoons, skunks, and snakes prey upon bats, although a limited number of animals consume bats as a regular part of their diet (Harvey *et al.* 1999, p. 13). Eastern small-footed and northern long-eared bats experience a very small amount of predation; therefore, predation does not appear to be a major cause of mortality (Caceres and Pybus 1997, p. 4; Whitaker and Hamilton 1998, p. 101).

Predation has been observed at a limited number of hibernacula within the range of the northern long-eared and eastern small-footed bats. Of the State and Federal agency responses received pertaining to eastern small-footed bat hibernacula and the threat of predation, only 8 out of 80 responses (10 percent) reported hibernacula as being prone to predation. For northern long-eared bats, 1 hibernacula in Maine, 3 in Maryland (2 of which were due to feral cats), 1 in Minnesota, and 10 in Vermont were reported as being prone to predation. In one instance, domestic cats were observed killing bats at a hibernaculum used by northern long-eared bat and eastern small-footed bat in Maryland, although the species of bat killed was not identified (Feller 2011, unpublished data). Turner (1999, personal observation) observed a snake (species unknown) capture an emerging Virginia big-eared bat (*Corynorhinus townsendii virginianus*) in West Virginia. The bat was captured in flight while the snake was perched along the top of a bat gate at the cave's entrance. Tuttle (1979, p. 11) observed (eastern) screech owls (*Otus asio*) capturing emerging gray bats.

Northern long-eared bats are known to be affected to a small degree by predators at summer roosts. Avian predators, such as owls and magpies, are known to successfully take individual bats as they roost in more open sites, although this most likely does not have an effect on the overall population size (Caceres and Pybus 1997, p. 4). In addition, Perry and Thill (2007, p. 224) observed a black rat snake (*Elaphe obsoleta obsoleta*) descending from a known maternity colony snag in the Ouachita Mountains of Arkansas. In summary, since bats are not a primary prey source for any known natural predators, it is unlikely that predation has substantial effects on either species at this time.

Conservation Efforts To Reduce Disease or Predation

As mentioned above, WNS is a disease that is responsible for unprecedented mortality in some hibernating bats in the northeast, like the northern long-eared bat, and it continues to spread throughout the range of the northern long-eared bat and eastern small-footed bat. Although conservation efforts have been undertaken to help reduce the spread of the disease through human-aided transmission, these efforts have only been in place for a few years and it is too early to determine how effective they are in decreasing the rate of spread. In 2008, the Service, along with several other State and Federal agencies, initiated a national plan (A National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-Nose Syndrome in Bats (WNS National Plan, *http://static.whitenosesyndrome.org/sites/default/files/white-nose_syndrome_national_plan_may_2011.pdf*)) that details the elements critical to investigating and managing WNS, along with identifying actions and roles for agencies and entities involved with the effort (Service 2011, p. 1). In addition to bat-to-bat transmission of the disease, fungal spores can be transmitted by humans (USGS National Wildlife Health Center, Wildlife Health Bulletin 2011–05). Therefore, the WNS Decontamination Team (a sub-group under the WNS National Plan), created a decontamination protocol (Service 2012, p. 2) that provides specific procedures to ensure human transmission risk to bats is minimized.

The Service also issued an advisory calling for a voluntary moratorium on all caving activity in States known to have hibernacula affected by WNS, and all adjoining States, unless conducted as part of an agency-sanctioned research or monitoring project (Service 2009). The Western Bat Working Group has also developed a White-nose Syndrome Action Plan, a comprehensive strategy to prevent the spread of WNS, that covers States currently outside the range of WNS (Western Bat Working Group 2010, p. 1–11). Although the majority of State and Federal agencies and tribes within the northern long-eared bat's and eastern small-footed bat's ranges have adopted the recommendations and protocols in the WNS National Plan, these are not mandatory or required. For example, in Virginia, the decontamination procedures are recommended for cavers; however, although the Virginia Department of Game and Inland Fisheries currently has closed the caves on the agencies' properties, they are reviewing this policy in light of the extensive spread of WNS throughout the State.

The NPS is currently updating their cave management plans (for parks with caves) to include actions to minimize the risk of WNS spreading to uninfected caves. These actions include WNS education, screening visitors for disinfection, and closure of caves if necessary (NPS 2013, *http://www.nature.nps.gov/biology/WNS*). In April 2009, all caves and mines on U.S. Forest Service lands in the Eastern Region were closed on an emergency basis in response to the spread of WNS. Eight National Forests in the Eastern Region contain caves or mines that are used by bats; caves and mines on seven of these National Forests (Allegheny, Hoosier, Ottawa, Mark Twain, Mononqahela, Shawnee, and Wayne) are currently closed, and no closure is needed for the one mine on the eighth National Forest (Green Mountain) because it is already gated with a bat-friendly structure. Forest supervisors continue to evaluate the most recent information on WNS to inform decisions regarding extending cave and mine closures for the purpose of limiting the spread of WNS (U.S. Forest Service 2013, *http://www.fs.fed.us/r9/wildlife/wildlife/bats.php*). Caves and mines on U.S. Forest Service lands in the Rocky Mountain Region were closed on an emergency basis in 2010, in response to WNS, but since then have been reopened, with some exceptions (U.S. Forest Service 2013, *http://www.fs.usda.gov/detail/r2/home/?cid=stelprdb5319926*). In place of the emergency closures, the Rocky Mountain Region will implement an adaptive management strategy that will require registration to access an open cave, prohibit use of clothing or equipment used in areas where WNS is found, require decontamination procedures prior to entering any and all caves, and close all known cave hibernacula during the winter hibernation period. Although the above mentioned WNS-related conservation measures may help reduce or slow the spread of the disease, these efforts are not currently enough to ameliorate the population-level effect to the northern long-eared bat.

Summary of Disease and Predation

In summary, while populations of several species of hibernating bats (*e.g.*, little brown bat, Indiana bat, northern long-eared bat, tri-colored bat) have experienced mass mortality due to WNS, populations of the eastern small-footed bat appear to be stable, and if they are in decline, the level of impact is not discernible at this time. Summer monitoring data are scarce, and the little data we have are inconclusive. However, based on the best available scientific information, we conclude that disease does not have an appreciable effect on the eastern small-footed bat.

Unlike the eastern small-footed bat, the northern long-eared bat has experienced a sharp decline, estimated at approximately 99 percent (from hibernacula data), in the northeastern portion of its range, due to the emergence of WNS. Summer survey data have confirmed rates of decline observed in northern long-eared bat hibernacula data post-WNS. The species is highly susceptible to WNS where the disease currently occurs in the East, and there is no reason to expect that western populations will be resistant to the disease. Thus, we expect that similar declines as seen in the East will be experienced in the future throughout the majority of the species' range. This is currently viewed as the predominant threat to the species, and if WNS had not emerged or was not affecting northern long-eared bat populations to the level that it has, we presume the species would not be declining to the degree observed.

As bats are not a primary prey source for any known natural predators, it is unlikely that predation is significantly affecting either species at this time.

Factor D. The Inadequacy of Existing Regulatory Mechanisms

Under this factor, we examine whether existing regulatory mechanisms are inadequate to address the threats to the species discussed under the other 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 minimize any of the threats we describe in threat analyses under the other four factors, or otherwise enhance conservation of the species. 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 constitution, or Federal action under statute.

Having evaluated the significance of the threat as mitigated by any such conservation efforts, we analyze under Factor D the extent to which existing regulatory mechanisms are inadequate to address the specific threats to the species. Regulatory mechanisms, if they exist, may reduce or eliminate the effects from one or more identified threats. In this section, we review existing State, Federal, and local regulatory mechanisms to determine whether they effectively reduce or remove threats to the eastern small-footed bat or northern long-eared bat.

No existing regulatory mechanisms have been designed to protect the species against WNS, the primary threat to the northern long-eared bat; thus, despite regulatory mechanisms that are currently in place, the species is still at risk. There are, however, some mechanisms in place to provide some protection from other factors that may act cumulatively with WNS. As such, the discussion below provides a few examples of such existing regulatory mechanisms, but is not a comprehensive list.

Federal

Several laws and regulations help Federal agencies protect bats on their lands, such as the Federal Cave Resources Protection Act (16 U.S.C. 4301 *et seq.*) that protects caves on Federal lands and National Environmental Policy Act (42 U.S.C. 4321 *et seq.*) review, which serves to mitigate effects to bats due to construction activities on federally owned lands. The NPS has additional laws, policies, and regulations that protect bats on NPS units, including the NPS Organic Act od 1916 (16 U.S.C. 1 *et seq.*), NPS management policies (related to exotic species and protection of native species), and NPS policies related to caves and karst systems (provides guidance on placement of gates on caves not only to address human safety concerns but also for the preservation of sensitive bat habitat) (Plumb and Budde 2011, unpublished data). Even if a bat species is not listed under the Endangered Species Act, the NPS works to minimize effects to the species. In addition, the NPS Research Permitting and Reporting System tracks research permit applications and investigator annual reports, and NPS Management Policies require non-NPS studies conducted in parks to conform to NPS policies and guidelines regarding the collection of bat data (Plumb and Budde 2011, unpublished data).

The northern long-eared bat is considered a "sensitive species" throughout U.S. Forest Service's Eastern Region (USDA Forest Service 2012). As such, the northern long-eared bat must receive, "special management emphasis to ensure its viability and to preclude trends toward endangerment that would result in the need for Federal listing. There must be no effects to sensitive species without an analysis of the significance of adverse effects on the populations, its habitat, and on the viability of the species as a whole. It is essential to establish population viability objectives when making decisions that would significantly reduce sensitive species numbers" (Forest Service Manual (FSM) 2672.1).

State

The eastern small-footed bat is State-listed as endangered in Maryland and New Hampshire; State-listed as threatened in Kentucky, Pennsylvania, South Carolina, and Vermont; and considered as a species of special concern in Connecticut, Delaware, Georgia, Indiana, Massachusetts, Missouri, New Jersey, New York, North Carolina, Ohio, Oklahoma, Tennessee, Virginia, and West Virginia. The level of protection provided under these laws varies by State, but most prohibit take, possession, or transport of listed species. For example, in Maryland, a person may not take, possess, transport, export, process, sell, offer for sale, or ship nongame wildlife (MD Code, Natural Resources, sec. 10–2A–01–09); however, effects to summer roosting habitat and direct mortality from wind energy development projects under 70 Megawatts (MW) are currently exempted from protections offered to the eastern small-footed bat (Feller 2011, unpublished data). In Pennsylvania, however, a House Bill proposed in the General Assembly, if passed, would not allow any "commonwealth agency to take action to classify or consider wildlife, flora or fauna as threatened or endangered unless the wildlife, flora or fauna is protected under the Endangered Species Act of 1973" (General Assembly of Pennsylvania 2013, p. 2).

The northern long-eared bat is listed in very few of the States within the species' range. The northern long-eared bat is listed as endangered under the Massachusetts endangered species act, under which all listed species are, "protected from killing, collecting, possessing, or sale and from activities that would destroy habitat and thus directly or indirectly cause mortality or disrupt critical behaviors." In addition, listed animals are specifically protected from activities that disrupt nesting, breeding, feeding, or migration (Massachusetts Division of Fisheries and Wildlife 2012, unpublished document). In Wisconsin, all cave bats, including the northern long-eared bat, were listed as threatened in the State in 2011, due to previously existing threats and the impending threat of WNS (Redell 2011, pers. comm.). Certain development projects (*e.g.*, wind energy), however, are excluded from regulations in place to protect the species in Wisconsin (Wisconsin Department of Natural Resources, unpublished document, 2011, p. 4). The northern long-eared bat is considered as some form of species of concern in 17 States: "Species of Greatest Concern" in Alabama and Rhode Island; "Species of Greatest Conservation Need" in Delaware, Iowa, and Vermont; "Species of Concern" in Ohio and Wyoming; "Rare Species of Concern" in South Carolina; "Imperiled" in Oklahoma; "Critically Imperiled" in Louisiana; and "Species of Special Concern" in Indiana, Maine, Minnesota, New Hampshire, North Carolina, Pennsylvania, and South Carolina.

In the following States, there is either no State protection law or the northern long-eared bat is not protected under the existing law: Arkansas, Connecticut, Florida, Georgia, Illinois, Kansas, Kentucky, Maryland, Mississippi, Missouri, Montana, Nebraska, New Jersey, New York, North Dakota, Tennessee, Virginia, and West Virginia. In Kentucky, although the northern long-eared bat does not have a State listing status, it is considered protected from take under Kentucky State law; however, since greater than 95 percent of hibernacula in Kentucky are privately owned, cave closures are not often possible to enforce (Hemberger 2011, unpublished data).

Wind energy development regulation varies by State within the northern long-eared bat's and eastern small-footed bat's ranges. For example, in Virginia, although there are not currently any wind energy developments in the State, new legislation requires mitigation for bats with the objective of reducing fatalities. As part of the regulation, operators are required to "measure the efficacy" of mitigation (Reynolds 2011 unpublished data). In Vermont, all wind projects are required to conduct bat mortality surveys, and at least 2 of the 3 currently permitted projects in the State include application of operational adjustments (curtailment) to reduce bat fatalities (Smith 2011, unpublished data).

Summary of Inadequacy of Existing Regulatory Mechanisms

No existing regulatory mechanisms have been designed to protect the species against WNS, the primary threat to the northern long-eared bat. Therefore, despite regulatory mechanisms that are currently in place for the northern long-eared bat, the species is still at risk, primarily due to WNS, as discussed under *Factor C.*

Factor E. Other Natural or Manmade Factors Affecting Its Continued Existence

Wind Energy Development

In general, bats are killed in significant numbers by utility-scale (greater than or equal to 0.66 megawatt (MW)) wind turbines along forested ridge tops in the eastern United States (Johnson 2005, p. 46; Arnett *et al.* 2008, p. 63). The majority of bats killed include migratory foliage-roosting species: the hoary bat (*Lasiurus cinereus*) and eastern red bat (*Lasiurus borealis*); migratory tree and cavity-roosting silver-haired bats (*Lasionycteris noctivagans*); and tri-colored bats (Arnett *et al.* 2008, p. 64).

Three effects may explain proximate causes of bat fatalities at wind turbines: (1) Bats collide with turbine towers, (2) bats collide with moving blades, or (3) bats suffer internal injuries (barotrauma) after being exposed to rapid pressure changes near the trailing edges and tips of moving blades (Cryan and Barclay 2009, p. 1331). It appears that barotrauma may be responsible for some deaths observed at wind-energy development sites. For example, nearly half of the 1,033 bat carcasses discovered over a 2-year study by Klug and Baerwald (2010, p. 15) had no fatal external injuries, and over 90 percent of those necropsied had internal injuries consistent with barotrauma (Baerwald *et al.* 2008, pp. 695–696). However, another study found that bone fractures from direct collision with turbine blades contributed to 74 percent of bat deaths, and therefore suggest that skeletal damage from direct collision with turbine blades is a major cause of fatalities for bats killed by wind turbines (Grodsky *et al.* 2011, p. 920). The authors suggest that these injuries can lead to an underestimation of bat mortality at wind energy facilities due to delayed lethal effects (Grodsky *et al.* 2011, p. 924). Lastly, the authors also note that the surface and core pressure drops behind the spinning turbine blades are high enough (equivalent to sound levels that are 10,000 times higher in energy density than the threshold of pain in humans (Cmiel *et al.* 2004)) to cause significant ear damage to bats flying near wind turbines (Grodsky *et al.* 2011, p. 924). Bats crippled by ear damage would have a difficult time navigating and foraging, since both of these functions depend on the bats' ability to echolocate (Grodsky *et al.* 2011, p. 924).

Wind projects have been constructed in areas within a large portion of the ranges of eastern small-footed bats and northern long-eared bats, suggesting these species may be exposed to the risk of turbine-related mortality. However, as of 2011, only two eastern small-footed bat and 13 northern long-eared bat fatalities were recorded from North American wind-energy facilities, representing less than 0.1 percent and 0.2 percent of the total bat mortality, respectively (American Wind Energy Association 2011, p. 18). Because eastern small-footed bats fly slowly and close to the ground (Davis *et al.* 1965, p. 683), they may be less susceptible to mortality caused by the operation of wind turbines.

The threat level posed by wind development to northern long-eared and eastern small-footed bats throughout their ranges varies. For example, in Illinois, wind energy development is viewed as a large threat to northern long-eared bats, especially during migration. Although the species is not considered a long-distance migrant, even limited migration distances between summer and winter habitats pose a risk to the northern long-eared bat in Illinois, due to the increasingly large line of wind farms across most of the central portion of the State (Kath 2012, pers. comm.). In 2012, 7 to 10 wind farms were in operation, and at least as many are planned. Further, northern long-eared bats have been found in pre-construction surveys for many of the wind farms (both planned and operational) (Kath 2012, pers. comm.). In Minnesota, wind energy development is moving at a rapid pace, and is one of the reasons State wildlife agency officials are concerned about the species' status in the State (Baker 2011, pers. comm.). In many States, such as Maryland, New Hampshire, South Carolina, and Vermont, wind energy projects have just recently been completed or are in the process of being installed; therefore, the level of mortality to northern long-eared bats and eastern small-footed bats has yet to be seen (Brunkhurst 2012, pers. comm.; Bunch 2011,unpublished data; Feller 2011, unpublished data; Smith 2011, unpublished data). Vermont currently has three permitted wind energy facilities in the State (the first of which is currently under construction), from which State officials see limited potential that northern long-eared bat fatalities will occur (Smith 2011, unpublished data), likely due to the current low population of the species in the State. We conclude that there may be adverse effects posed by wind energy development to northern long-eared bats and eastern small-footed bats; however, there is no evidence suggesting effects from wind energy development in itself have led to population declines in either species.

Climate Change

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

Scientific measurements spanning several decades demonstrate that changes in climate are occurring, and that the rate of change has been faster since the 1950s. Examples include warming of the global climate system, and substantial increases in precipitation in some regions of the world and decreases in other regions. (For these and other examples, see IPCC 2007a, p. 30; Solomon *et al.* 2007, pp. 35–54, 82–85). Results of scientific analyses presented by the IPCC show that most of the observed increase in global average temperature since the mid–20th century cannot be explained by natural variability in climate, and is "very likely" (defined by the IPCC as 90 percent or higher probability) due to the observed increase in greenhouse gas (GHG) concentrations in the atmosphere as a result of human activities, particularly carbon dioxide emissions from use of fossil fuels (IPCC 2007a, pp. 5–6 and figures SPM.3 and SPM.4; Solomon *et al.* 2007, pp. 21–35). Further confirmation of the role of GHGs comes from analyses by Huber and Knutti (2011, p. 4), who concluded it is extremely likely that approximately 75 percent of global warming since 1950 has been caused by human activities.

Scientists use a variety of climate models, which include consideration of natural processes and variability, as well as various scenarios of potential levels and timing of GHG emissions, to evaluate the causes of changes already observed and to project future changes in temperature and other climate conditions (*e.g.*, Meehl *et al.* 2007, entire; Ganguly *et al.* 2009, pp. 11555, 15558; Prinn *et al.* 2011, pp. 527, 529). All combinations of models and emissions scenarios yield very similar projections of increases in the most common measure of climate change, average global surface temperature (commonly known as global warming), until about 2030. Although projections of the magnitude and rate of warming differ after about 2030, the overall trajectory of all the projections is one of increased global warming through the end of this century, even for the projections based on scenarios that assume that GHG emissions will stabilize or decline. Thus, there is strong scientific support for projections that warming will continue through the 21st century, and that the magnitude and rate of change will be influenced substantially by the extent of GHG emissions (IPCC 2007a, pp. 44–45; Meehl *et al.* 2007, pp. 760–764 and 797–811; Ganguly *et al.* 2009, pp. 15555–15558; Prinn *et al.* 2011, pp. 527, 529). (See IPCC 2007b, p. 8, for a summary of other global projections of climate-related changes, such as frequency of heat waves and changes in precipitation. Also see IPCC 2011 (entire) for a summary of observations and projections of extreme climate events.)

Various changes in climate may have direct or indirect effects on species. These effects may be positive, neutral, or negative, and they may change over time, depending on the species and other relevant considerations, such as interactions of climate with other variables (*e.g.*, habitat fragmentation) (IPCC 2007, pp. 8–14, 18–19). Identifying likely effects often involves aspects of climate change vulnerability analysis. Vulnerability refers to the degree to which a species (or system) is susceptible to, and unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the type, magnitude, and rate of climate change and variation to which a species is exposed, its sensitivity, and its adaptive capacity (IPCC 2007a, p. 89; see also Glick *et al.* 2011, pp. 19–22). There is no single method for conducting such analyses that applies to all situations (Glick *et al.* 2011, p. 3). We use our expert judgment and appropriate analytical approaches to weigh relevant information, including uncertainty, in our consideration of various aspects of climate change.

As is the case with all stressors that we assess, even if we conclude that a species is currently affected or is likely to be affected in a negative way by one or more climate-related effects, it does not necessarily follow that the species meets the definition of an "endangered species" or a "threatened species" under the Act. If a species is listed as endangered or threatened, knowledge regarding the vulnerability of the species to, and known or anticipated impacts from, climate-associated changes in environmental conditions can be used to help devise appropriate strategies for its recovery.

The unique natural history traits of bats and their susceptibility to local temperature, humidity, and precipitation patterns make them an early warning system for effects of climate change in regional ecosystems (Adams and Hayes 2008, p. 1120). Climate change is expected to alter seasonal ambient temperatures and precipitation patterns across regions (Adams and Hayes 2008, p. 1115). The ability of successful reproductive effort in female insectivorous bats is related directly to roost temperatures and water availability (Adams and Hayes 2008, p. 1116). Adams and Hayes (2008, p. 1120) predict an overall decline in bat populations in the western United States from reduced regional water storage caused by climate warming. In comparison, the northeast United States is projected to see a steady increase in annual winter precipitation, although a much greater proportion is expected to fall as rain rather than as snow. Overall, little change in summer rainfall is expected, although projections are highly variable (Frumhoff *et al.* 2007, p. 8). Based on this model, water availability should not be a limiting factor to bats in the northeast United States.

Climate change may result in warmer winters, which could lead to a reduced period of hibernation, increased winter activity, and reduced reliance on the relatively stable temperatures of underground hibernation sites (Jones *et al.* 2009, p. 99). Hibernation sites chosen by eastern small-footed bats (*e.g.*, under rocks) may be even more susceptible to temperature fluctuations, which may lead to energy depletion that reduces winter survival (Rodenhouse *et al.* 2009, p. 251). An earlier spring would presumably result in a shorter hibernation period and the earlier appearance of foraging bats (Jones *et al.* 2009, p. 99). An earlier emergence from hibernation may have no detrimental effect on population size if sufficient food is available (Jones *et al.* 2009, p. 99); however, predicting future insect population dynamics and distributions is complex (Bale *et al.* 2002, p. 6). Alterations in precipitation, stream flow, and soil moisture could influence insect populations in such a way as to potentially alter food availability for bats (Rodenhouse *et al.* 2009, p. 250).

Warmer winter temperatures may also disrupt bat reproductive physiology. Both eastern small-footed bats and northern long-eared bats breed in the fall, and spermatozoa are stored in the uterus of hibernating females until spring ovulation. If bats experience warm conditions they may arouse from hibernation prematurely, ovulate, and become pregnant (Jones *et al.* 2009, p. 99). Given this dependence on external temperatures, climate change is likely to affect the timing of reproductive cycles (Jones *et al.* 2009, p. 99), but whether these effects would be to the detriment of the species is largely unknown. A shorter hibernation period and warmer winter temperatures may lead to less exposure and slower spread of WNS or persistence of the fungus, which would likely benefit both species. However, the rapid rate at which WNS is affecting the species is on a much quicker time scale than are the changes associated with climate change. Thus, longer-term effects of climate change are unlikely to have an impact on the short-term effects of WNS. Although we do have information that suggests that climate change may impact both the northern long-eared bat and eastern small-footed bat and bats in general, we do not have any evidence suggesting that climate change in itself has led to population declines in either species.

Contaminants

Effects to bats from contaminant exposure have likely occurred and gone, for the most part, unnoticed among bat populations (Clark and Shore 2001, p. 204). Contaminants of concern to insectivorous bats like the eastern small-footed and northern long-eared bats include organochlorine pesticides, organophosphate, carbamate and neonicotinoid insecticides, polychlorinated biphenyls and polybrominated diphenyl ethers (PBDEs), pyrethroid insecticides, and inorganic contaminants such as mercury (Clark and Shore 2001, pp. 159–214).

Organochlorine pesticides (*e.g.*, DDT, chlordane) persist in the environment due to lipophilic (fat-loving) properties, and therefore readily accumulate within the fat tissue of bats. Because insectivorous bats have high metabolic rates, associated with flight and small size, their food intake increases the amount of organochlorines available for concentration in the fat (Clark and Shore 2001, p. 166). Because bats are long-lived, the potential for bioaccumulation is great, and effects on reproduction and populations have been documented (Clark and Shore 2001, pp. 181–190). In maternity colonies, young bats appear to be at the greatest risk of mortality. This is because organochlorines become concentrated in the fat of the mother's milk and these chemicals continually and rapidly accumulate in the young as they nurse (Clark 1988, pp. 410–411).

In addition to indirect effects of contaminants on bats via prey consumption, documented cases of population-level effects involve direct application of pesticides to bats or their roosts. For example, when a mixture of DDT and chlordane was applied to little brown bats and their roost site, mortality from exposure was observed (Kunz *et al.* 1977, p. 478). Most organochlorine pesticides have been banned in the United States and have largely been replaced by organophosphate insecticides, which are generally short-lived in the environment and do not accumulate in food chains; however, risk of exposure is still possible from direct exposure from spraying or ingesting insects that have recently been sprayed but have not died, or both (Clark 1988, p. 411). Organophospahate and carbamate insecticides are acutely toxic to mammals. Also, some organophosphates may be stored in fat tissue and contribute to "organophosphate-induced delayed neuropathy" in humans (USEPA 2013, p. 44).

Bats are less sensitive to organophosphate insecticides than birds in regards to acute toxicity, but many bats lose their motor coordination from direct application and are unlikely to survive in the wild in an incapacitated state lasting over 24 hours (Plumb and Budde 2011, unpublished data). Bats may be exposed to organophosphate and carbamate insecticides in regions where methyl parathion is applied in cotton fields and where malathion is used for mosquito control (Plumb and Budde 2011, unpublished data). The organophosphate, chlorpyrifos, has high fat solubility and is commonly used on crops such as corn, soybeans (van Beelen 2000, p. 34 of Appendix 2; *http://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=2009&map=CHLORPYRIFOS&hilo=L*).

The neonicotinoids have been found to cause oxidative stress, neurological damage and possible liver damage in rats and immune suppression in mice (*http://www.sciencedirect.com/science/article/pii/S0048357512001617* Badgujar *et al.* 2013, p. 408; Duzguner 2012, p. 58; Kimura-Kuroda *et al.* 2011, p. 381), Due to information indicating that there is a link between neonicotinoids used in agriculture and a decline in bee numbers, the European Union proposed a two year ban on the use of the neonocotinoids, thiamethoxam, imidacloprid and clothianidin on crops attractive to honeybees, beginning in December of 2013 (*http://www.lawbc.com/regulatory-developments/entry/proposal-for-restriction-of-neonicotinoid-products-in-the-eu/*).

The more recently developed "third generation" of pyrethroids have acute oral toxicities rivaling the toxicity of organophosphate, carbamate and organochlorine pesticides. These pyrethroids include esfenvalerate, deltamethrin, bifenthrin, tefluthrin, flucythrinate, cyhalothrin and fenpropathrin (Mueller-Beilschmidt 1990, p. 32). Pyrethroids are increasingly used in the United States, and some of these compounds have very high fat solubility (*e.g.*, bifenthrin, cypermethrin) (van Beelen 2000, p. 34 of Appendix 2).

Like the organochlorine pesticides, PCBs and PBDEs are highly lipophilic and therefore readily accumulate in insectivorous bats. Outside of laboratory experiments, there is no conclusive evidence that bats have been killed by PCBs, although effects on reproduction have been observed (Clark and Shore 2001, pp. 192–194).

In New Hampshire, to limit the amount of plant material growing on the rock slope of the Surry Mountain Reservoir, the U.S. Army Corps of Engineers spray the rock slope with herbicide; this site is an eastern small-footed bat summer roosting site (Veilleux and Reynolds 2006, p. 331). It is unknown whether the direct application of herbicide on the roost area reduces the roost quality or causes mortality of adult bats, young bats, or both.

Eastern small-footed bats and northern long-eared bats forage on emergent insects and can be characterized as occasionally foraging over water (Yates and Evers 2006, p. 5), and therefore are at risk of exposure to bioaccumulation of inorganic contaminants (*e.g.*, cadmium, lead, mercury) from contaminated water bodies. Bats tend to accumulate inorganic contaminants due to their diet and slow means of elimination of these compounds (Plumb and Budde 2011, unpublished data). In Virginia, for example, the North Fork Holston River is a water body that was highly contaminated by a waterborne point source of mercury through contamination by a chlor-alkali plant. Based on findings from a pilot study for bats in 2005 (Yates and Evers 2006), there is sufficient information to conclude that bats from near-downstream areas of the North Fork Holston River have potentially harmful body burdens of mercury, although the effect on bats is unknown. Fur samples taken from eastern small-footed bats have also yielded detectable amounts of mercury and zinc (Hickey *et al.* 2001, p. 703). Hickey *et al.* (2001, p. 705) suggest that the concentrations of mercury reported may be sufficient to cause sublethal biological effects to bats. Divoll *et al.* (*in prep*) found that eastern small-footed bats and northern long-eared bats showed consistently higher mercury levels than little brown bats or eastern red bats sampled in Maine, which may be correlated with gleaning behavior and the consumption of spiders by these two bat species. Eastern small-footed bats exhibited the highest mercury levels of all species. Bats recaptured during the study 1 or 2 years after their original capture maintained similar levels of mercury in fur year-to-year. Biologists suggest that individual bats accumulate body burdens of mercury that cannot be reduced once elevated to a certain threshold.

Exposure to holding ponds containing flow-back and produced water associated with hydraulic fracturing operations may also expose bats to toxins, radioactive material, and other contaminants (Hein 2012, p. 8). Cadmium, mercury, and lead are contaminants reported in hydraulic fracturing operations. Whether bats drink directly from holding ponds or contaminants are introduced from these operations into aquatic ecosystems, bats will presumably accumulate these substances and potentially suffer adverse effects (Hein 2012, p. 9). In summary, the best available data indicate that contaminant exposure can pose an adverse effect to individual northern long-eared and eastern small-footed bats, although it is not an immediate and significant risk in itself at a population level.

Prescribed Burning

Eastern forest-dwelling bat species, such as the eastern small-footed and northern long-eared bats, likely evolved with fire management of mixed-oak ecosystems (Perry 2012, p. 182). A recent review of prescribed fire and its effects on bats (U.S. Forest Service 2012, p. 182) generally found that fire had beneficial effects on bat habitat. Fire may create snags for roosting and creates more open forests conducive to foraging on flying insects (Perry 2012, pp. 177–179), although gleaners such as northern long-eared bats may readily use cluttered understories for foraging (Owen *et al.* 2003, p. 355). Cavity and bark roosting bats, such as the eastern small-footed and northern long-eared, use previously burned areas for both foraging and roosting (Johnson *et al.* 2009, p. 239; Johnson *et al.* 2010, p. 118). In Kentucky, the abundance of prey items for northern long-eared bats increased after burning (Lacki *et al.* 2009, p. 1170), and more roosts were found in post-burn areas (Lacki *et al.* 2009, p. 1169). Burning may create more suitable snags for roosting through exfoliation of bark (Johnson *et al.* 2009, p. 240), mimicking trees in the appropriate decay stage for roosting bats. In contrast, a prescribed burn in Kentucky caused a roost tree used by a radio-tagged female northern long-eared bat to prematurely fall after its base was weakened by smoldering combustion (Dickinson *et al.* 2009, p. 56). Low-intensity burns may not kill taller trees directly but may create snags of smaller trees and larger trees may be injured, resulting in vulnerability (of the tree) to pathogens that cause hollowing of the trunk, which provides roosting habitat (Perry 2012, p. 177). Prescribed burning also opens the tree canopy, providing more canopy light penetration (Boyles and Aubrey 2006, p. 112; Johnson *et al.* 2009, p. 240), which may facilitate faster development of juvenile bats (Sedgeley 2001, p. 434). Although Johnson *et al.* (2009, p. 240) found the amount of roost switching did not differ between burned and unburned areas, the rate of switching in burned areas of every 1.35 days was greater than that found in other studies of every 2–3 days (Foster and Kurta 1999, p. 665; Owen *et al.* 2002, p. 2; Carter and Feldhamer 2005, p. 261; Timpone *et al.* 2010, p. 119).

Direct effects of fire on bats likely differ among species and seasons (Perry 2012, p. 172). Northern long-eared bats have been seen flushing from tree roosts shortly after ignition of prescribed fire during the growing season (Dickinson *et al.* 2009, p. 60). Fires of reduced intensity that proceed slowly allow sufficient time for roosting bats to arouse from sleep or torpor and escape the fire (Dickinson *et al.* 2010, p. 2200), although extra arousals from fire smoke could cause increased energy loss (Dickinson *et al.* 2009, p. 52). During prescribed burns, bats are potentially exposed to heat and gases; the roosting behavior of these two species, however, may reduce their vulnerability to toxic gases. When trees are dormant, the bats are roosting in caves or mines (hibernacula can be protected from toxic gases through appropriate burn plans), and during the growing season, northern long-eared bats roost in tree cavities or under bark above the understory, above the area with the highest concentration of gases in a low-intensity prescribed burn (Dickinson *et al.* 2010, pp. 2196, 2200). Carbon monoxide levels did not reach critical thresholds that could harm bats in low-intensity burns at the typical roosting height for the eastern small-footed and northern long-eared bats (Dickinson *et al.* 2010, p. 2196); thus heat effects from prescribed fire are of greater concern than gas effects on bats. Direct heat could cause injury to the thin tissue of bat ears and is more likely to occur than exposure to toxic gas levels during prescribed burns (Dickinson *et al.* 2010, p. 2196). In addition, fires of reduced intensity with shorter flame height could lessen the effect of heat to bats roosting higher in trees (Dickinson *et al.* 2010, p. 2196).

Winter, early spring, and late fall generally contain less intense fire conditions than during other seasons and coincide with time periods when bats are less affected by prescribed fire due to low activity in forested areas. Furthermore, no young are present during these times, which reduces the likelihood of heat injury and exposure of vulnerable young to fire (Dickinson *et al.* 2010, p. 2200). Prescribed fire objectives, such as fires with high intensity and rapid ignition in order to meet vegetation goals, must be balanced with the exposure of bats to the effects of fire (Dickinson *et al.* 2010, p. 2201). Currently, the Service and U.S. Forest Service strongly recommend not burning in the central hardwoods from mid- to late April through summer to avoid periods when bats are active in forests (Dickinson *et al.* 2010, p. 2200).

Bats that occur in forests are likely equipped with evolutionary characteristics that allow them to exist in environments with prescribed fire. Periodic burning can benefit habitat through snag creation and forest canopy gap creation, but frequency and timing need to be considered to avoid direct and indirect adverse effects to bats when using prescribed burns as a management tool. We conclude that there may be adverse effects posed by prescribed burning to individual northern long-eared bats and eastern small-footed bats; however, there is no evidence suggesting effects from prescribed burning itself have led to population declines in either species.

Conservation Efforts To Reduce Other Natural or Manmade Factors Affecting Its Continued Existence

In the Midwest, rapid wind development is a concern with regards to the effect to bats (Baker 2011, pers. comm.; Kath 2012, pers. comm.). Due to the known impact from wind energy development, in particular to listed (and species currently being evaluated to determine if listing is warranted) bird and bat species in the Midwest, the Service, State natural resource agencies, and wind energy industry representatives are developing the Midwest Wind Energy Multi-Species Habitat Conservation Plan (MSHCP). The planning area includes the Midwest Region of the Service, which includes all or portions of the following States: Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio, and Wisconsin. The MSHCP would allow permit holders to proceed with wind energy development, which may result in "incidental" taking of a listed species under section 10 of the Act, through issuance of an incidental take permit (77 FR 52754; August 30, 2012). Currently, both the northern long-eared bat and eastern small-footed bat are being considered for inclusion as covered species under the MSHCP. The MSHCP will address protection of covered species through avoidance, minimization of take, and mitigation to offset effect of "take" (*e.g.*, habitat preservation, habitat restoration, habitat enhancement) to help ameliorate the effect of wind development (77 FR 52754; August 30, 2012). In some cases, the U.S. Forest Service has agreed to limit or restrict burning in the central hardwoods from mid- to late April through summer to avoid periods when bats are active in forests (Dickinson *et al.* 2010, p. 2200).

Summary of Factor E

We have identified a number of factors (*e.g.*, wind energy development, climate change, contaminants, prescribed burning) that may have direct or indirect effects on eastern small-footed bats and northern long-eared bats. Although such activities occur, there is no evidence that these activities alone have significant effects on either species, because their effects are often localized and not widespread throughout the species' ranges. However, these factors may have a cumulative effect on the northern long-eared bat when added to white-nose syndrome, because the disease had led to dramatic population declines in that species (discussed under *Factor C*).

Cumulative Effects From Factors A Through E

None of the factors discussed above under Factors A, B, C, or E, alone or in combination, is affecting the eastern small-footed bat at a population level. Conversely, WNS (Factor C) alone has led to dramatic and rapid population-level effects on the northern long-eared bat. White-nose syndrome is the most significant threat to the northern long-eared bat, and the species would likely not be imperiled were it not for this disease. However, although the effects on the northern long-eared bat from Factors A, B, and E individually or in combination do not have significant effects on the species, when combined with the significant population reductions due to white-nose syndrome (Factor C), the resulting cumulative effect may further adversely impact the species.

Finding

Eastern Small-Footed Bat

As required by the Act, we considered the five factors in assessing whether the eastern small-footed bat is endangered or threatened throughout all of its range. We examined the best scientific and commercial information available regarding the past, present, and future threats faced by the eastern small-footed bat. We reviewed the petition, information available in our files, and other available published and unpublished information, and we consulted with recognized bat experts and other Federal and State agencies. Threats previously identified for the eastern small-footed bat include modification or destruction of winter and summer habitat, disturbance of hibernating bats from commercial and/or recreational activities in caves and mines, disease, wind energy development, climate change, and contaminants. The primary threat previously identified was WNS. While other species of hibernating bats have experienced mass mortality due to WNS, there is no indication of a population-level decline in eastern small-footed bat based on winter survey data. A review of pre-WNS and post-WNS hibernacula count data over multiple years finds that post-WNS counts were within the normal observed range at the majority of sites analyzed. Several life-history traits may reduce the susceptibility of this bat to WNS, which include their comparatively late arrival and early departure from hibernacula, departure from hibernacula during mild winter periods, solitary roosting habits, and selection of drier microhabitats (*e.g.*, cave and mine entrances). We will continue to closely monitor the spread of WNS and its effects on eastern small-footed bats. As for the other above-mentioned threats, although there is risk of exposure and individual mortality in isolated incidences, no declines in eastern small-footed bat populations have been documented.

Our review of the best available scientific and commercial information indicates that the eastern small-footed bat is not in danger of extinction (endangered) nor likely to become endangered within the foreseeable future (threatened), throughout all of its range.

Distinct Vertebrate Population Segment

After assessing whether the species is endangered or threatened throughout its range, we next consider whether a distinct vertebrate population segment (DPS) of the eastern small-footed bat meets the definition of an endangered or threatened species.

Under the Service's Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the Endangered Species Act (61 FR 4722;

February 7, 1996 (DPS Policy)), three elements are considered in the decision concerning the establishment and classification of a possible DPS. These are applied similarly for additions to or removal from the Federal List of Endangered and Threatened Wildlife. These elements include:

(1) The discreteness of a population in relation to the remainder of the species to which it belongs;

(2) The significance of the population segment to the species to which it belongs; and

(3) The population segment's conservation status in relation to the Act's standards for listing, delisting, or reclassification (i.e., is the population segment endangered or threatened).

Discreteness

Under the DPS policy, a population segment of a vertebrate taxon may be considered discrete if it satisfies either one of the following conditions:

(1) It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors. Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation; or

(2) It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the Act.

There are no characteristics of the eastern small-footed bat's taxonomy, distribution or abundance, habitat, or biology (see the *Species Information* section, above) that suggest the species may be segmented into discrete populations. Throughout its range, the eastern small-footed bat has similar morphology and, as far as we know, genetics; uses similar roosting and foraging habitat; and exhibits similar roosting, foraging, and reproductive behavior. Therefore, the best available information indicates there is no evidence of markedly separated eastern small-footed bat populations.

There are no characteristics of the eastern small-footed bat's management that suggest the species may be segmented into discrete populations. The eastern small-footed bat occurs in the Canadian provinces of Ontario and Quebec, as well as in the United States. However, the species is not listed under Canada's Species At Risk Act. In addition, we have no information to suggest that the species, its habitat, or the potential threats evaluated above in the five factor analysis are managed differently in the Canadian versus U.S. portions of the eastern small-footed bat's range. Therefore, the best available information indicates that there is no evidence that the eastern small-footed bat is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the Act.

We determine, based on a review of the best available information, that no population of the eastern small-footed bat meets the discreteness conditions of the 1996 DPS policy. Therefore, no eastern small-footed bat population qualifies as a DPS under our policy, and no population is a listable entity under the Act.

The DPS policy is clear that significance is analyzed only when a population segment has been identified as discrete. Since we found that no population segment meets the discreteness element and, therefore, does not qualify as a DPS under the Service's DPS policy, we will not conduct an evaluation of significance.

Significant Portion of the Range

Under the Act and our implementing regulations, a species may warrant listing if it is endangered or threatened throughout all or a significant portion of its range. The Act defines "endangered species" as any species which is "in danger of extinction throughout all or a significant portion of its range," and "threatened species" as any species which is "likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." The definition of "species" is also relevant to this discussion. The Act defines "species" as follows: "The term 'species' includes any subspecies of fish or wildlife or plants, and any distinct population segment [DPS] of any species of vertebrate fish or wildlife which interbreeds when mature." The phrase "significant portion of its range" (SPR) is not defined by the statute, and we have never addressed in our regulations: (1) The consequences of a determination that a species is either endangered or likely to become so throughout a significant portion of its range, but not throughout all of its range; or (2) what qualifies a portion of a range as "significant."

Two recent district court decisions have addressed whether the SPR language allows the Service to list or protect less than all members of a defined "species": *Defenders of Wildlife* v. *Salazar,* 729 F. Supp. 2d 1207 (D. Mont. 2010), concerning the Service's delisting of the Northern Rocky Mountain gray wolf (74 FR 15123; April 2, 2009); and *WildEarth Guardians* v. *Salazar,* 2010 U.S. Dist. LEXIS 105253 (D. Ariz. September 30, 2010), concerning the Service's 2008 finding on a petition to list the Gunnison's prairie dog (73 FR 6660; February 5, 2008). The Service had asserted in both of these determinations that it had authority, in effect, to protect only some members of a "species," as defined by the Act (i.e., species, subspecies, or DPS), under the Act. Both courts ruled that the determinations were arbitrary and capricious on the grounds that this approach violated the plain and unambiguous language of the Act. The courts concluded that reading the SPR language to allow protecting only a portion of a species' range is inconsistent with the Act's definition of "species." The courts concluded that once a determination is made that a species (i.e., species, subspecies, or DPS) meets the definition of "endangered species" or "threatened species," it must be placed on the list in its entirety and the Act's protections applied consistently to all members of that species (subject to modification of protections through special rules under sections 4(d) and 10(j) of the Act).

Consistent with that interpretation, and for the purposes of this finding, we interpret the phrase "significant portion of its range" in the Act's definitions of "endangered species" and "threatened species" to provide an independent basis for listing; thus there are two situations (or factual bases) under which a species would qualify for listing: A species may be endangered or threatened throughout all of its range; or a species may be endangered or threatened in only a significant portion of its range. If a species is in danger of extinction throughout a significant portion of its range, the species is an "endangered species." The same analysis applies to "threatened species." Based on this interpretation and supported by existing case law, the consequence of finding that a species is endangered or threatened in only a significant portion of its range is that the entire species shall be listed as endangered or threatened, respectively, and the Act's protections shall be applied across the species' entire range.

We conclude, for the purposes of this finding, that interpreting the significant portion of its range phrase as providing an independent basis for listing is the best interpretation of the Act because it is consistent with the purposes and the plain meaning of the key definitions of the Act; it does not conflict with established past agency practice (i.e., prior to the 2007 Solicitor's Opinion), as no consistent, long-term agency practice has been established; and it is consistent with the judicial opinions that have most closely examined this issue. Having concluded that the phrase "significant portion of its range" provides an independent basis for listing and protecting the entire species, we next turn to the meaning of "significant" to determine the threshold for when such an independent basis for listing exists.

Although there are potentially many ways to determine whether a portion of a species' range is "significant," we conclude, for the purposes of this finding, that the significance of the portion of the range should be determined based on its biological contribution to the conservation of the species. For this reason, we describe the threshold for "significant" in terms of an increase in the risk of extinction for the species. We conclude that a biologically based definition of "significant" best conforms to the purposes of the Act, is consistent with judicial interpretations, and best ensures species' conservation. Thus, for the purposes of this finding, and as explained further below, a portion of the range of a species is "significant" if its contribution to the viability of the species is so important that without that portion, the species would be in danger of extinction.

We evaluate biological significance based on the principles of conservation biology using the concepts of redundancy, resiliency, and representation. Resiliency describes the characteristics of a species and its habitat that allow it to recover from periodic disturbance. Redundancy (having multiple populations distributed across the landscape) may be needed to provide a margin of safety for the species to withstand catastrophic events. Representation (the range of variation found in a species) ensures that the species' adaptive capabilities are conserved. Redundancy, resiliency, and representation are not independent of each other, and some characteristic of a species or area may contribute to all three. For example, distribution across a wide variety of habitat types is an indicator of representation, but it may also indicate a broad geographic distribution contributing to redundancy (decreasing the chance that any one event affects the entire species), and the likelihood that some habitat types are less susceptible to certain threats, contributing to resiliency (the ability of the species to recover from disturbance). None of these concepts is intended to be mutually exclusive, and a portion of a species' range may be determined to be "significant" due to its contributions under any one or more of these concepts.

For the purposes of this finding, we determine if a portion's biological contribution is so important that the portion qualifies as "significant" by asking whether without that portion, the representation, redundancy, or resiliency of the species would be so impaired that the species would have an increased vulnerability to threats to the point that the overall species would be in danger of extinction (i.e., would be "endangered"). Conversely, we would not consider the portion of the range at issue to be "significant" if there is sufficient resiliency, redundancy, and representation elsewhere in the species' range that the species would not be in danger of extinction throughout its range if the population in that portion of the range in question became extirpated (extinct locally).

We recognize that this definition of "significant" (a portion of the range of a species is "significant" if its contribution to the viability of the species is so important that without that portion, the species would be in danger of extinction) establishes a threshold that is relatively high. On the one hand, given that the consequences of finding a species to be endangered or threatened in a significant portion of its range would be listing the species throughout its entire range, it is important to use a threshold for "significant" that is robust. It would not be meaningful or appropriate to establish a very low threshold whereby a portion of the range can be considered "significant" even if only a negligible increase in extinction risk would result from its loss. Because nearly any portion of a species' range can be said to contribute some increment to a species' viability, use of such a low threshold would require us to impose restrictions and expend conservation resources disproportionately to conservation benefit: Listing would be rangewide, even if only a portion of the range of minor conservation importance to the species is imperiled. On the other hand, it would be inappropriate to establish a threshold for "significant" that is too high. This would be the case if the standard were, for example, that a portion of the range can be considered "significant" only if threats in that portion result in the entire species' being currently endangered or threatened. Such a high bar would not give the significant portion of its range phrase independent meaning, as the Ninth Circuit held in *Defenders of Wildlife* v. *Norton*, 258 F.3d 1136 (9th Cir. 2001).

The definition of "significant" used in this finding carefully balances these concerns. By setting a relatively high threshold, we minimize the degree to which restrictions will be imposed or resources expended that do not contribute substantially to species conservation. But we have not set the threshold so high that the phrase "in a significant portion of its range" loses independent meaning. Specifically, we have not set the threshold as high as it was under the interpretation presented by the Service in the *Defenders* litigation. Under that interpretation, the portion of the range would have to be so important that current imperilment there would mean that the species would be currently imperiled everywhere. Under the definition of "significant" used in this finding, the portion of the range need not rise to such an exceptionally high level of biological significance. (We recognize that if the species is imperiled in a portion that rises to that level of biological significance, then we should conclude that the species is in fact imperiled throughout all of its range, and that we would not need to rely on the significant portion of its range language for such a listing.) Rather, under this interpretation we ask whether the species would be endangered everywhere without that portion, i.e., if that portion were completely extirpated. In other words, the portion of the range need not be so important that even the species being in danger of extinction in that portion would be sufficient to cause the species in the remainder of the range to be endangered; rather, the complete extirpation (in a hypothetical future) of the species in that portion would be required to cause the species in the remainder of the range to be endangered.

The range of a species can theoretically be divided into portions in an infinite number of ways. However, there is no purpose to analyzing portions of the range that have no reasonable potential to be significant or to analyzing portions of the range in which there is no reasonable potential for the species to be endangered or threatened. To identify only those portions that warrant further consideration, we determine whether there is substantial information indicating that: (1) The portions may be "significant," and (2) the species may be in danger of extinction there or likely to become so within the foreseeable future. Depending on the biology of the species, its range, and the threats it faces, it might be more efficient for us to address the significance question first or the status question first. Thus, if we determine that a portion of the range is not "significant," we do not need to determine whether the species is endangered or threatened there; if we determine that the species is not endangered or threatened in a portion of its range, we do not need to determine if that portion is "significant." In practice, a key part of the determination that a species is in danger of extinction in a significant portion of its range is whether the threats are geographically concentrated in some way. If the threats to the species are essentially uniform throughout its range, no portion is likely to warrant further consideration. Moreover, if any concentration of threats to the species occurs only in portions of the species' range that clearly would not meet the biologically based definition of "significant," such portions will not warrant further consideration.

We evaluated the current range of the eastern small-footed bat to determine if there is any apparent geographic concentration of potential threats for the species. We examined potential habitat threats from modification of cave and mine openings, mine reclamation, vandalism, wind energy development, and timber harvesting (Factor A); disturbance from cave recreation and research-related activities (Factor B); WNS and predation (Factor C); the inadequacy of existing regulatory mechanisms (Factor D); and collisions from wind energy development projects, climate change, contaminants, and prescribed burning (Factor E). We found no concentration of threats that suggests that the eastern small-footed bat may be in danger of extinction in a portion of its range. We found no portions of its range where potential threats are significantly concentrated or substantially greater than in other portions of its range. Therefore, we find that factors affecting the eastern small-footed bat are essentially uniform throughout its range, indicating no portion of the range warrants further consideration of possible endangered or threatened status under the Act. There is no available information indicating that there has been a range contraction for the species, and therefore we find that lost historical range does not constitute a significant portion of the range for the eastern small-footed bat. Our review of the best available scientific and commercial information indicates that the eastern small-footed bat is not in danger of extinction (endangered) nor likely to become endangered within the foreseeable future (threatened), throughout all of its range or in a significant portion of its range. Therefore, we find that listing the eastern small-footed bat as an endangered or threatened species under the Act is not warranted at this time.

We request that you submit any new information concerning the status of, or threats to, the eastern small-footed bat to our Pennsylvania Field Office, 315 South Allen Street, Suite 322, State College, PA 16801, whenever it becomes available. New information will help us monitor the eastern small-footed bat and encourage its conservation. If an emergency situation develops for the eastern small-footed bat, we will act to provide immediate protection.

Northern Long-Eared Bat

As required by the Act, we considered the five factors in assessing whether the northern long-eared bat is an endangered or threatened species, as cited in the petition, throughout all of its range. We examined the best scientific and commercial information available regarding the past, present, and future threats faced by the northern long-eared bat. We reviewed the petition, information available in our files, and other available published and unpublished information, and we consulted with recognized bat and disease experts and other Federal and State agencies.

This status review identifies that the primary threat to the northern long-eared bat is attributable to WNS (Factor C), a disease caused by the fungus *Geomyces destructans* that is known to kill bats. The disease has led to dramatic and rapid population declines in northern long-eared bats of up to 99 percent from pre-WNS levels in some areas. White-nose syndrome has spread rapidly throughout the East and is currently spreading through the Midwest. We have no information to indicate that there are areas within the species' range that will not be impacted by the disease or that similar rates of decline (to what has been observed in the East, where the disease has been present for at most 8 years) will not occur throughout the species' range. Other sources of mortality to the species include wind-energy development, habitat modification, destruction and disturbance (*e.g.*, vandalism to hibernacula, roost tree removal), effects of climate change, and contaminants. Although no significant decline due to these factors has been observed, they may have cumulative effects to the species in addition to WNS.

On the basis of the best scientific and commercial information available, we find that the petitioned action to list the northern long-eared bat as an endangered or threatened species is warranted. A determination on the status of the species as an endangered or threatened species is presented below in the proposed listing determination.

Proposed Determination for Northern Long-Eared Bat

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 information available regarding the past, present, and future threats to the northern long-eared bat. There are several factors that affect the northern long-eared bat; however, we have found that no other threat is as severe and immediate to the species persistence as WNS (Factor C). Predominantly due to the emergence of WNS, the northern long-eared bat has experienced a severe and rapid decline in the Northeast, estimated at approximately 99 percent (from hibernacula data) since the disease was first discovered there in 2007. Summer survey data in the Northeast have confirmed rates of decline observed in northern long-eared bat hibernacula data post-WNS, with rates of decline ranging from 93 to 98 percent. This disease is considered the prevailing threat to the species, as there is currently no known cure. As mentioned under Factor C, although at the current time the disease has not spread throughout the species' entire range (WNS is currently found in 22 of 39 States where the northern long-eared bat occurs), it continues to spread, and we have no reason not to expect that where it spreads, it will have the same impact to the affected species (Coleman 2013, pers. comm.). Although there is some uncertainty as far as when the disease will spread throughout the northern long-eared bat's range, all models that have attempted to project the spread of WNS (presented in Factor C) were in agreement that WNS will indeed spread across the United States. In addition, human transmission could introduce the spread of the fungus to new locations that are far removed from the current known locations (Coleman 2013, pers. comm.). This threat is ongoing, is expected to increase in the future, and is significant because it continues to extirpate northern long-eared bat populations as it spreads and is expected to continue to spread throughout the species' range. Other threats to the northern long-eared bat include wind-energy development, winter and summer habitat modification, destruction and disturbance (*e.g.*, vandalism to hibernacula, roost tree removal), climate change, and contaminants. Although these threats (prior to WNS) have not in and of themselves had significant impacts at the species level, they may increase the overall impacts to the species when considered cumulatively with WNS.

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 northern long-eared bat is presently in danger of extinction throughout its entire range based on the severity and immediacy of threats currently affecting the species. The overall range has been significantly impacted because a large portion of populations in the eastern part of the range have been extirpated due to WNS. White-nose syndrome is currently or is expected in the near future to impact the remaining populations. In addition other factors are acting in combination with WNS to reduce the overall viability of the species. The risk of extinction is high because the species is considered less common to rare in the areas not yet, but anticipated to soon be, affected by WNS, and significant rates of decline have been observed over the last 6 years in the core of the species' range, which is currently affected by WNS; these rates of decline are especially high in the eastern part of the species' range, where rates of decline have been as high as 99 percent in hibernating populations of the species. Therefore, on the basis of the best available scientific and commercial information, we propose listing the northern long-eared bat as endangered in accordance with sections 3(6) and 4(a)(1) of the Act. We find that a threatened species status is not appropriate for the northern long-eared bat because the threat of WNS has significant effects where it has occurred and is expected to spread rangewide in a short timeframe.

Under the Act and our implementing regulations, a species may warrant listing if it is endangered or threatened throughout all or a significant portion of its range. The threats to the survival of the species occur throughout the species' range and are not restricted to any particular significant portion of that range. Accordingly, our assessment and proposed determination applies to the species 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 the final recovery plan will be available on our Web site (*http://www.fws.gov/endangered*), or from our Green Bay, Wisconsin, Field 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, Tribal, nongovernmental organizations, businesses, and private landowners. Examples of recovery actions include habitat protection, habitat restoration (*e.g.*, restoration of native vegetation) and management, research, captive propagation and reintroduction, and outreach and education. The recovery of many listed species cannot be accomplished solely on Federal lands because their range may occur primarily or solely on non-Federal lands. To achieve 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, under section 6 of the Act, the State(s) of Alabama, Arkansas, Connecticut, Delaware, Florida, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, Montana, Nebraska, New Hampshire, New Jersey, New York, North Carolina, North Dakota, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, South Dakota, Tennessee, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming, and the District of Columbia, would be eligible for Federal funds to implement management actions that promote the protection or recovery of the northern long-eared bat. Information on our grant programs that are available to aid species recovery can be found at: *http://www.fws.gov/grants*.

Although the northern long-eared bat 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 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 any other landscape-altering activities on Federal lands administered by the U.S. Fish and Wildlife Service, U.S. Forest Service, NPS, and other Federal agencies; issuance of section 404 Clean Water Act (33 U.S.C. 1251 *et seq.*) permits by the U.S. Army Corps of Engineers; 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 and threatened 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.

It is our policy, as published in the **Federal Register** on July 1, 1994 (59 FR 34272), 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 effect of a proposed 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) Incidental take of the species without authorization pursuant to section 7 or section 10(a)(1)(B) of the Act.

(3) Disturbance or destruction of known hibernacula due to commercial or recreational activities during known periods of hibernation.

(4) Unauthorized destruction or modification of summer habitat (including unauthorized grading, leveling, burning, herbicide spraying, or other destruction or modification of habitat) in ways that kills or injures individuals by significantly impairing the species' essential breeding, foraging, sheltering, or other essential life functions.

(5) Unauthorized removal or destruction of trees and other natural and manmade structures being utilized as roosts by the northern long-eared bat that results in take of the species.

(6) Unauthorized release of biological control agents that attack any life stage of this taxon.

(7) Unauthorized removal or exclusion from buildings or artificial structures being used as roost sites by the species, resulting in take of the species.

(8) Unauthorized building and operation of wind energy facilities within areas used by the species, which results in take of the species.

(9) Unauthorized discharge of chemicals, fill, or other materials into sinkholes which may lead to contamination of known northern long-eared bat hibernacula.

Questions regarding whether specific activities would constitute a violation of section 9 of the Act should be directed to the Green Bay, Wisconsin Ecological Services Field Office (see **FOR FURTHER INFORMATION CONTACT**).

Critical Habitat for Northern Long-Eared Bat

Background

Critical habitat is defined in section 3 of the Act as:

(1) The specific areas within the geographical area occupied by the species, at the time it is listed in accordance with the Act, on which are found those physical or biological features

(a) Essential to the conservation of the species, and

(b) Which may require special management considerations or protection; and

(2) Specific areas outside the geographical area occupied by the species at the time it is listed, upon a determination that such areas are essential for the conservation of the species.

Conservation, as defined under section 3 of the Act, means to use and the use of all methods and procedures that are necessary to bring an endangered or threatened species to the point at which the measures provided pursuant to the Act are no longer necessary. Such methods and procedures include, but are not limited to, all activities associated with scientific resources management such as research, census, law enforcement, habitat acquisition and maintenance, propagation, live trapping, and transplantation, and, in the extraordinary case where population pressures within a given ecosystem cannot be otherwise relieved, may include regulated taking.

Critical habitat receives protection under section 7 of the Act through the requirement that Federal agencies ensure, in consultation with the Service, that any action they authorize, fund, or carry out is not likely to result in the destruction or adverse modification of critical habitat. The designation of critical habitat does not affect land ownership or establish a refuge, wilderness, reserve, preserve, or other conservation area. Such designation does not allow the government or public to access private lands. Such designation does not require implementation of restoration, recovery, or enhancement measures by non-Federal landowners. Where a landowner requests Federal agency funding or authorization for an action that may affect a listed species or critical habitat, the consultation requirements of section 7(a)(2) of the Act would apply, but even in the event of a destruction or adverse modification finding, the obligation of the Federal action agency and the landowner is not to restore or recover the species, but to implement reasonable and prudent alternatives to avoid destruction or adverse modification of critical habitat.

Under the first prong of the Act's definition of critical habitat, areas within the geographical area occupied by the species at the time it was listed are included in a critical habitat designation if they contain physical or biological features (1) which are essential to the conservation of the species and (2) which may require special management considerations or protection. For these areas, critical habitat designations identify, to the extent known using the best scientific and commercial data available, those physical or biological features that are essential to the conservation of the species (such as space, food, cover, and protected habitat). In identifying those physical and biological features within an area, we focus on the principal biological or physical constituent elements (primary constituent elements such as roost sites, nesting grounds, seasonal wetlands, water quality, tide, soil type) that are essential to the conservation of the species. Primary constituent elements are those specific elements of the physical or biological features that provide for a species' life-history processes and are essential to the conservation of the species.

Under the second prong of the Act's definition of critical habitat, we can designate critical habitat in areas outside the geographical area occupied by the species at the time it is listed, upon a determination that such areas are essential for the conservation of the species. For example, an area currently occupied by the species but that was not occupied at the time of listing may be essential to the conservation of the species and may be included in the critical habitat designation. We designate critical habitat in areas outside the geographical area occupied by a species only when a designation limited to its range would be inadequate to ensure the conservation of the species.

Section 4 of the Act requires that we designate critical habitat on the basis of the best scientific data available. Further, our Policy on Information Standards Under the Endangered Species Act (published in the **Federal Register** on July 1, 1994 (59 FR 34271)), the Information Quality Act (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001 (Pub. L. 106–554; H.R. 5658)), and our associated Information Quality Guidelines, provide criteria, establish procedures, and provide guidance to ensure that our decisions are based on the best scientific data available. They require our biologists, to the extent consistent with the Act and with the use of the best scientific data available, to use primary and original sources of information as the basis for recommendations to designate critical habitat.

When we are determining which areas should be designated as critical habitat, our primary source of information is generally the information developed during the listing process for the species. Additional information sources may include the recovery plan for the species, articles in peer-reviewed journals, conservation plans developed by States and counties, scientific status surveys and studies, biological assessments, other unpublished materials, or experts' opinions or personal knowledge.

Habitat is dynamic, and species may move from one area to another over time. We recognize that critical habitat designated at a particular point in time may not include all of the habitat areas that we may later determine are necessary for the recovery of the species. For these reasons, a critical habitat designation does not signal that habitat outside the designated area is unimportant or may not be needed for recovery of the species. Areas that are important to the conservation of listed species, both inside and outside the critical habitat designation, continue to be subject to: (1) Conservation actions implemented under section 7(a)(1) of the Act, (2) regulatory protections afforded by the requirement in section 7(a)(2) of the Act for Federal agencies to ensure their actions are not likely to jeopardize the continued existence of any endangered or threatened species, and (3) section 9 of the Act's prohibitions on taking any individual of the species, including taking caused by actions that affect habitat. Federally funded or permitted projects affecting listed species outside their designated critical habitat areas may still result in jeopardy findings in some cases. These protections and conservation tools will continue to contribute to recovery of this species. Similarly, critical habitat designations made on the basis of the best available information at the time of designation will not control the direction and substance of future recovery plans, habitat conservation plans (HCPs), or other species conservation planning efforts if new information available at the time of these planning efforts calls for a different outcome.

Prudency Determination

Section 4(a)(3) of the Act, as amended, and implementing regulations (50 CFR 424.12), require that, to the maximum extent prudent and determinable, the Secretary designate critical habitat at the time the species is determined to be endangered or threatened. Our regulations (50 CFR 424.12(a)(1)) state that the designation of critical habitat is not prudent when one or both of the following situations exist: (1) The species is threatened by taking or other human activity, and identification of critical habitat can be expected to increase the degree of threat to the species, or (2) such designation of critical habitat would not be beneficial to the species.

There is currently no imminent threat of take attributed to collection or vandalism under Factor B for the northern long-eared bat, and identification and mapping of critical habitat is not expected to initiate any such threat. In the absence of finding that the designation of critical habitat would increase threats to a species, if there are any benefits to a critical habitat designation, then a prudent finding is warranted. The potential benefits of designation include: (1) Triggering consultation under section 7 of the Act, in new areas for actions in which there may be a Federal nexus where it would not otherwise occur because, for example, it is or has become unoccupied or the occupancy is in question; (2) focusing conservation activities on the most essential features and areas; (3) providing educational benefits to State or county governments or private entities; and (4) preventing people from causing inadvertent harm to the species. Therefore, because we have determined that the designation of critical habitat will not likely increase the degree of threat to the species and may provide some measure of benefit, we find that designation of critical habitat is prudent for the northern long-eared bat.

Critical Habitat Determinability

Having determined that designation is prudent, under section 4(a)(3) of the Act we must find whether critical habitat for the species is determinable. Our regulations at 50 CFR 424.12(a)(2) state that critical habitat is not determinable when one or both of the following situations exist: (i) Information sufficient to perform required analyses of the impacts of the designation is lacking, or (ii) The biological needs of the species are not sufficiently well known to permit identification of an area as critical habitat.

We reviewed the available information pertaining to the biological needs of the species and habitat characteristics where this species is located. Since information regarding the biological needs of the species is not sufficiently well known to permit identification of areas as critical habitat, we conclude that the designation of critical habitat is not determinable for the northern long-eared bat at this time.

There are many uncertainties in designating hibernacula as critical habitat for the northern long-eared bat. First, we are not able to establish which of the large number of known hibernacula the species is known to inhabit are essential to the conservation of the species. This is due to the species typically being found in small numbers (often fewer than 10 individuals per hibernaculum). Also, those hibernacula with historically greater numbers (greater than 100) are often now infected with WNS, where the northern long-eared bat has been extirpated or close to extirpated. In addition, we lack sufficient information to define the physical and biological features or primary constituent elements with enough specificity; we are not able to determine how habitats affected by WNS (where populations previously thrived and are now extirpated) may contribute to the recovery of the species or whether those areas may still contain essential physical and biological features. Finally, for several States (*e.g.*, Alabama, Iowa, Kansas, Montana, Nebraska, North Dakota, Oklahoma) within the species' range it is unknown if hibernacula occur within parts of the State, due to either the lack of survey effort or (especially the case in the western part of the range) the species being sparsely populated over a large landscape, making locating potential hibernacula challenging. Therefore, we currently lack the information necessary to propose critical habitat for the species.

There are also uncertainties with potential designation of summer habitat, specifically maternity colony habitat. Although research has given us indication of some key summer roost requirements, the northern long-eared bat appears to be somewhat opportunistic in roost selection, selecting varying roost tree species and types of roosts throughout the range. Thus, it is not clear whether certain summer habitats are essential for the recovery of the species, or whether summer habitat is not a limiting factor for the species. Although research has shown some consistency in female summer roost habitat (*e.g.*, selection of mix of live trees and snags as roosts, roosting in cavities, roosting beneath bark, and roosting in trees associated with closed canopy), the species and diameter of the tree (when tree roost is used) selected by northern long-eared bats for roosts vary widely depending on availability. Therefore, we are currently unable to determine whether specific summer habitat features are essential to the conservation of the species, and find that critical habitat is not determinable for the northern long-eared bat at this time. We will seek more information regarding the specific winter and summer habitat features and requirements for the northern long-eared bat and make a determination on critical habitat no later than 1 year following any final listing.

Peer Review

In accordance with our joint policy published in the **Federal Register** on July 1, 1994 (59 FR 34270), we will seek the expert opinions of at least three appropriate and independent specialists regarding this proposed rule. The purpose of peer review is to ensure that our listing determination for this species is based on scientifically sound data, assumptions, and analyses. We will invite these peer reviewers to comment during the public comment period.

We will consider all comments and information we receive during the comment period on this proposed rule during preparation of a final rulemaking. Accordingly, the final decision may differ from this proposal.

Public Hearings

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 proposal in the **Federal Register**. Such requests must be sent to the address shown in the **FOR FURTHER INFORMATION CONTACT** section. We will schedule public hearing 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 Green Bay, Wisconsin, Field Office at 920–866–1717, as soon as possible. To allow sufficient time to process requests, please call no later than 1week before the hearing date. Information regarding this proposed rule is available in alternative formats upon request.

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 (NEPA; 42 U.S.C. 4321 *et seq.*), 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 references cited in this rulemaking is available on the Internet at *http://www.regulations.gov* and upon request from the Green Bay, Wisconsin, Field Office (see **FOR FURTHER INFORMATION CONTACT**).

Authors

The primary authors of this proposed rule are the staff members of the Green Bay, Wisconsin, Field Office and the State College, Pennsylvania, Ecological Services Field Office.

List of Subjects in 50 CFR Part 17

Endangered and threatened species, Exports, Imports, Reporting and recordkeeping requirements, 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. Amend § 17.11(h) by adding an entry for "Bat, northern long-eared" in alphabetical order under MAMMALS to the List of Endangered and Threatened Wildlife to read as follows:

§ 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						
MAMMALS							
*	*	*	*	*	*	*	
Bat, northern long-eared.	*Myotis septentrionalis.*	U.S.A. (AL, AR, CT, DE, DC, FL, GA, IL, IN, IA, KS, KY, LA, ME, MD, MA, MI, MN, MS, MO, MT, NE, NH, NJ, NY, NC, ND, OH, OK, PA, RI, SC, SD, TN, VT, VA, WV, WI, WY); Canada (AB, BC, LB, MB, NB, NF, NS, NT, ON, PE, QC, SK, YT).	Entire	E		NA	NA
*	*	*	*	*	*	*	

Dated: September 10, 2013.

Stephen Guertin,

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

[FR Doc. 2013–23753 Filed 10–1–13; 8:45 am]

BILLING CODE 4310–55–P

BATS OF THE BLACK HILLS
A DESCRIPTION OF STATUS AND CONSERVATION NEEDS

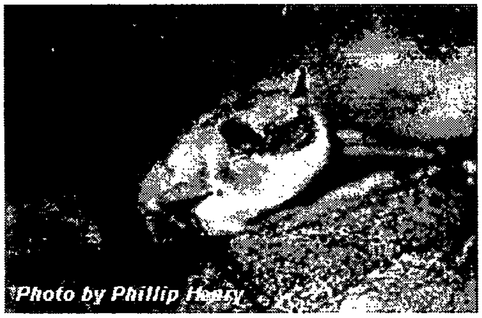


Photo by Phillip Henry

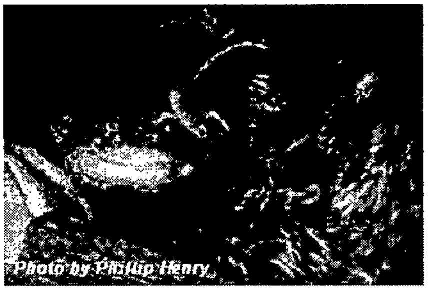


Photo by Phillip Henry


Photo by Joel Tigner

JOEL TIGNER AND EILEEN DOWD STUKEL
SOUTH DAKOTA DEPARTMENT OF GAME, FISH AND PARKS
WILDLIFE DIVISION REPORT 2003-05
MARCH 2003

CONTENTS

INTRODUCTION

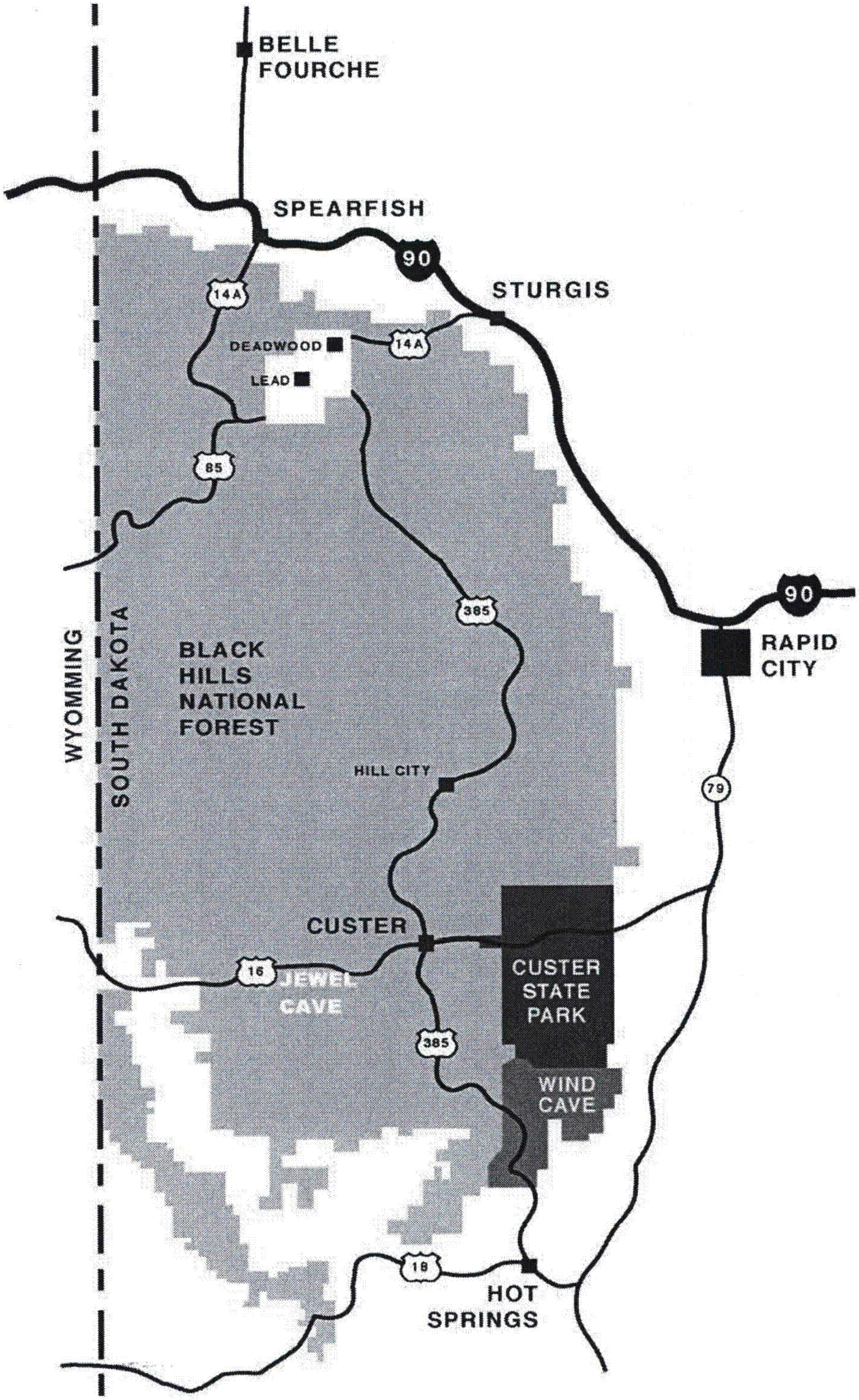
".. to most of us they are still nearly as unknown as the stars.." G.M. Allen *BATS*

Little has changed since that observation of bats was made in 1939. Despite the recent increase in the study of bats, they remain among the least understood and most maligned of animals. Falling victim to popularly held misconceptions and changes in habitat, few animals are as susceptible to human vagaries as bats (Hill and Smith 1984). In recent years, significant declines in bat populations have been documented worldwide, most in response to loss of roosting habitat (McCracken 1988). Reversing this trend will require widespread recognition of their ecological contributions as well as protection of foraging and roosting habitat (Kunz and Pierson 1994).

To gain a better understanding of bats and their requirements, this study was conducted to identify species residing in the Black Hills and their roosting habits. Each species was studied to gain insight about management actions needed to protect significant roosting and foraging sites.

This publication summarizes previous work conducted in the Black Hills, supplemented with results of fieldwork conducted primarily during the 1990s. The authors would like to acknowledge that much of the research cited during this period was funded by the South Dakota Department of Game, Fish and Parks, the Federal Aid in Wildlife Restoration Program, and the U.S. Forest Service.

The goal of this publication is to create a resource tool for others to use to continue the study of bats in the Black Hills. It is also a source of information in which species-specific management recommendations are made to protect the current populations within the region. It is hoped this publication will provide a summary of the information collected, and that by consolidating such findings, this work will suggest direction for future research, prevent costly duplication of effort, and enable informed responses to questions on management issues.


BELLE FOURCHE
SPEARFISH
90
STURGIS
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DEADWOOD
LEAD
14A
85
WYOMING
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HOT SPRINGS

BAT SPECIES OF THE BLACK HILLS

Bats represent one of the most diverse groups of animals. Within the class of Mammalia, they are second only to rodents in total number of species. Their order, Chiroptera, is comprised of over 900 species, nearly a quarter of all mammal species. Chiroptera is then divided into two sub-orders, the Megachiroptera and the Microchiroptera (van Zyll de Jong 1985). The former are found only in the Old World tropics while the latter, to which all of the Black Hills' species belong, are represented in all but extreme arctic regions (van Zyll de Jong 1985). All bats found within the Black Hills belong to the family Vespertilionidae and are exclusively insectivorous (van Zyll de Jong 1985). Until recently, their importance as the primary predator of nocturnal flying insects, many of which are responsible for substantial damage to forests and crops, has gone largely unrecognized (Tuttle 1988).

The physiological adaptations of bats to their environment are highly specific and sophisticated. Fossil evidence suggests their unique morphological adaptations to the environment have been present for at least the last 50 million years (Habersetzer et al. 1994, Jepson 1970, Novacek 1985). While displaying such typical mammalian characteristics as fur bearing, live birth and nursing of young, bats are unique among mammals with their ability of sustained flight (Fenton 1992).

One of their more amazing characteristics is their use of echolocation in foraging and navigating. Bats emit ultrasonic pulses, then receive and interpret reflection of these pulses' echoes much like sonar systems (Griffin 1958). These echoes produce a "sound picture" within the bat's brain that enables it to forage and navigate with phenomenal speed and pinpoint accuracy.

Regional bat species diversity is typically linked to habitat diversity (Stebbings and Griffith 1986). Within the Black Hills, mixtures of forest, grassland, and riparian habitat coupled with the occurrence of numerous caves and abandoned mines combine to create diversity unique to the central plains of the United States.

The South Dakota Natural Heritage Program currently monitors five species of bats found within the Black Hills. The Wyoming Game and Fish Department (1996) has set objectives for bat inventories statewide. The following bat species can be found within the Black Hills of South Dakota and Wyoming and are known to be year-round residents (Anderson 1993, Choate and Jones 1981, Martin and Hawks 1972, Tigner and Aney 1994, Turner 1974, Worthington and Bogan 1993):

Species names used throughout this document are based upon current conventions as noted at NatureServe Explorer: An online encyclopedia of life [web application]. 2002. Version 1.6. Arlington, Virginia, USA: NatureServe. Available: http://www.natureserve.org/explorer.

Myotis ciliolabrum (Western Small-footed Myotis)

Myotis evotis (Long-eared Myotis)*

Myotis lucifugus (Little Brown Myotis)

Myotis septentrionalis (Northern Myotis)*

Myotis thysanodes pahasapensis (Fringed Myotis)*

Myotis volans (Long-legged Myotis)

Corynorhinus townsendii pallescens [formerly *Plecotus t. p.*] (Townsend's Big-eared Bat)*

Eptesicus fuscus (Big Brown Bat)

Three other species, considered "tree-roosting" bats, are migratory and winter in milder climates. These are:

Lasionycteris noctivagans (Silver-haired Bat)*

Lasiurus borealis (Eastern Red Bat)

Lasiurus cinereus (Hoary Bat)

*indicates species monitored by SD Natural Heritage Program

Based upon the presence of suitable habitat, Turner (1974) suggested another species, *Euderma maculatum* (Spotted Bat), may occur in the Black Hills, though there are no records to date.

A hibernaculum survey conducted at an abandoned mine in the central Black Hills on 01/07/03 yielded a single specimen of *Pipistrellus subflavus* (Eastern Pipistrelle). This is the first record for this species in the Black Hills. Additional regional records for this species include three identified hibernating in a cave in Goshen County, Wyoming (Grenier personal communication) and from Greeley, Colorado (Fitzgerald et al.1989).

Vocal signatures for this species were also recorded using the ANABAT detector system at McKenna Spring in the southern Black Hills (Mike O'Farrell [O'Farrell Biological Consulting, Las Vegas, Nevada] personal communication).

DATA COLLECTION

Surveys

A variety of survey methods has been employed to study bat populations in the Black Hills (Cryan and Bogan 1996, and others). One limiting factor directly affecting the ability to draw conclusions from survey work conducted to date is lack of historical data to serve as a baseline with which comparisons can be made. Changes in population dynamics and patterns of distribution within the Black Hills are difficult to assess based upon current information. As such, one of the more important contributions made by recent studies is the establishment of baseline population data. If collected regularly and objectively, future biologists can use the information to monitor population trends.

Historically, reported population sizes and declines were largely based upon hibernacula surveys (Humphrey 1975, Ransome 1990, Tuttle 1977). In the Black Hills, there are only three known hibernacula that contain more than 300 bats. Most support fewer than 25 bats. Given the well-documented colonial behavior of many of the region's species during hibernation, such numbers demonstrate a significant lack of information regarding wintering behaviors. An alternative premise suggests wintering bats of the region hibernate in cracks and fissures, a common feature of Black Hills geology. If true, hibernacula surveys as a basis for gauging population trends are untenable.

One long time resident, on whose property lies a popular "show cave," described bats emerging on a summer evening in 1932 as a "column of smoke." He recalls this daily emergence lasting several minutes. Today there are no known sites in the Black Hills whose numbers would compare with this observation.

Recent Black Hills studies have exploited newer technology, such as radio telemetry, to identify roost sites for poorly understood species (Cryan and Bogan 1996, Mattson 1994).

Banding

Tigner and Aney (1994) collected information via banding and year-round roost monitoring. Banding was the primary method used to collect seasonal range and roost fidelity data beginning in 1992. Surveys conducted since 1992 have reported no observations of bats banded during earlier studies.

Bats were banded only during the active time of year. No hibernating bats were banded or disturbed to read bands. Band numbers were often hidden by roosting posture or roosting location. As a result, information from winter observations was often limited to species and sex. Males were banded on the left forearm, while females were banded on the right. This distinction was rigidly observed to enable sex determination during hibernation surveys when bats could not be disturbed. See TABLE 1 for banding information.

Bats were captured with mist nets and harp traps outside night roosts and at foraging sites. Some were caught via static hand-held nets inside the roost. Captures within a roost were only used at roosting sites where exclusion from the roost was imminent, such as building remodeling or demolition, mine closure, and intentional roost exclusion.

Survey Bias

Environmental characteristics, such as surface water, may affect distribution patterns. Riparian areas, with their higher insect prey densities, consistently yield higher capture rates than uplands (Cross 1988). Capture rates were highest in the southern Black Hills where limited surface water likely served to concentrate insect prey.

Roost availability also affects species distribution (Kunz 1982, Tuttle and Stevenson 1982). Bats with specific roost requirements are more susceptible to changes in habitat than more opportunistic species. Human induced change, such as firewood collection, timber harvest, natural or deliberate mine closure, and disturbance or vandalism within natural caves all can influence roost availability resulting in changes in distribution. In addition to roost availability, proximity to other requirements, such as foraging areas, can affect distribution (Kunz 1982).

Population trend data for migratory bats must be interpreted with caution. Migratory species may be affected by factors unrelated to summer habitats (Thomas and LaVal 1988), such as pesticide exposure during migration or on wintering range (Clark 1981). Species that are characteristically more sedentary, but about which limited information has been collected, are also difficult to assess (Thomas and LaVal 1988).

Sex segregation during maternity and nursery season also affects survey results. Netting surveys conducted at foraging sites in the southern Black Hills yielded a male:female ratio of 2:1 in *Myotis* species (Cryan and Bogan 1996, Mattson 1994, Tigner unpublished data). Similar findings are characteristic of capture rates in the northern Black Hills (Tigner unpublished data). Such segregation is likely a result of different summer roosting requirements. Cryan and Bogan (1996) have also suggested this segregation may serve to demonstrate the importance of areas in which reproductive females occur.

In general, reproductive females were more frequently captured at lower elevations (Cryan and Bogan 1996, Mattson 1994, Tigner unpublished data). Selection of lower elevations by reproductive females may be a response to thermoregulatory requirements during the maternity/nursery season.

Number of nets and net placement affect the capture rate at a given location (Kunz and Kurta 1988). Netting surveys may yield disproportionately high numbers of species less adept at obstacle evasion. While none of the Hills species could be characterized as bumbling, *C. townsendii* is the species least susceptible to traditional capture methods. This species can be commonly observed flying through small openings in mist nets requiring folded wings. Similarly, this species frequently evades capture by harp traps. After several apparent reconnaissance approaches, individuals will dive with folded wings through the top of the trap allowing momentum to carry them through the second bank of strings.

Natural population fluctuations also affect survey information. Poor foraging years or cold winters often result in high mortality or altered migration patterns (Ransome 1990). Without a historical perspective to gauge fluctuations, interpretation of "point-in-time" surveys becomes somewhat limited.

Seasonal variations in weather patterns may also affect population distribution. Variations between summers may yield different survey information. Such variations are important components in determining population trends.

Flooding may cause bat populations to decline. Many caves in the east-central region of the Black Hills exhibit historical evidence of complete flooding. While such flooding may only occur rarely, low reproductive rates in bats make population recovery slow. Such periodic cleansing may also remove evidence of historical use by bats.

BASIC BAT BIOLOGY AND ITS IMPLICATIONS

Understanding the biological adaptations that characterize bats is essential to design effective conservation objectives (Kunz 1982). It is beyond the scope of this report to detail all of these, but characteristics that may be affected by land management activities are discussed.

Hibernation

Increasing disturbance of known hibernacula throughout the Black Hills poses one of the most serious threats to year-round bat populations. Winter is one of the most critical times of year for bats (Ransome 1990). While some species demonstrate a degree of flexibility in summer roost site selection, diminished or non-existent winter food supplies require year-round resident species[1] to seek hibernacula that meet specific conditions. Some hypogeal species travel great distances to winter roosting sites (Fenton and Barclay 1980, Ransome 1990), but there is no information to suggest this behavior is characteristic of Black Hills species. Of the Black Hills' eight hypogeal species, seven[2] are confirmed year-round residents (Anderson 1993, Mattson and Bogan 1993, Tigner and Aney 1994).

During hibernation, bats lower their metabolic rate reducing the expenditure of stored energy (Ransome 1990). Each species has an optimal temperature range at which there is a minimum expenditure of these hibernal reserves (McNab 1974, Ransome 1990). Deviation from this optimal range requires the bat to regulate its metabolism. If the microclimate of the hibernaculum becomes too cold, stored reserves must be used to prevent the bat from freezing. In contrast, warm conditions prevent bats from lowering metabolic rates, and hibernal stores are depleted too rapidly. While winter survival can be completely dependent upon stored reserves, brief warm spells may also permit bats to supplement reserves by foraging and drinking (Ransome 1990).

In addition to temperature and relative humidity, physical environment is another important feature of suitable hibernacula (Ransome 1990). The hibernacula must contain an area that affords bats protection from predators. Bats are unable to evade predators during hibernation.

Thermoregulating behaviors exhibited by bats vary according to species (McNab 1974, Ransome 1990). Two of the more common behaviors adopted as means of maintaining stable temperatures are clustering and roosting within cracks and crevices (McNab 1974, Ransome 1990). If a hibernaculum microclimate becomes unsuitable, the bat will arouse from hibernation and seek conditions that are more favorable. Such movement may involve simple shifting within the hibernaculum or may require complete site abandonment and relocation (Ransome 1990).

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[1] *L. noctivagans, L. borealis* and *L. cinereus* are considered migratory.

[2] No winter records have been recorded for *Myotis evotis* in the Black Hills.

The habit of forming hibernation clusters can put large segments of bat populations at risk. Disturbing a small number of bats in a cluster may result in a cascade effect. The movement of a few bats in contact with other hibernators could disturb a large percentage of the collective roost (Thomas 1995). Identifying such sites and protecting them from disturbance is an important component of any conservation strategy for bats in the Black Hills.

Arousal from hibernation is extremely demanding on stored energy reserves. Each arousal that elevates a bat's body temperature to permit flight can cost 10 to 30 days of hibernal reserves (Tuttle 1991). Frequent disturbances within hibernacula can result in exhaustion of energy reserves resulting in starvation (Ransome 1990). Unusually cold winters or poor foraging seasons can result in lower hibernal reserves, thereby increasing susceptibility to disturbance (Tuttle 1991). In addition to environmental causes of arousal, human disturbance can be deleterious. Examples include spray-painting cave and mine interiors, constructing campfires, and discharging fireworks. Less obvious is the increase in ambient temperature caused by body heat dissipation and lighting sources. Noises generated by movement and talking can also disturb hibernating bats (Thomas 1995). Complete arousal from hibernation can be prolonged with the bat awakening after the source of disturbance has departed, leaving those responsible unaware of their impact.

Historically, hibernal requirements were probably met by the abundant natural caves found throughout the limestone periphery of the Black Hills. Based upon numbers, the largest known hibernacula in the Black Hills are located in natural caves. Through a variety of circumstances, many natural caves are no longer suitable. Commercial cave development, natural erosion, and human disturbance all contributed to a reduction in the number of available hibernacula.

Mining created artificial roosting and hibernacula alternatives for seven of the hypogeal bat species in the Black Hills. However, given the minimal amount of research conducted to date on mine utilization, it is difficult to determine the role mines play in sustaining bat populations. This relationship remains one of most important areas yet to be investigated. One abandoned mine identified in Custer County during the winter of 2002/03 contained the third largest collective of *Corynorhinus townsendii* yet identified in the Black Hills.

In the Black Hills, bats generally begin arriving at hibernacula in late September or early October. Depending upon weather conditions and the species, bats hibernate from October until April. Based upon observations of banded bats, hibernacula also serve as night roosts throughout the summer season. With two exceptions, all positive identifications in hibernacula were of bats banded at the same roost during the summer months.

Jewel Cave National Monument, located 18.5 km west of Custer, SD, is the largest known wintering site for bats in the Black Hills (Choate and Anderson 1997). Recent winter surveys yielded total counts of approximately 1200 bats comprised of seven species (Choate and Anderson 1997). These include: *Corynorhinus townsendii*, *Myotis ciliolabrum*, *Myotis lucifugus*, *Myotis septentrionalis*, *Myotis thysanodes pahasapensis*, *Myotis volans*, and *Eptesicus fuscus*. Jewel Cave serves as hibernacula to one of the largest known collectives of

Corynorhinus townsendii in the western United States (Worthington and Bogan 1993). As such, its ecological importance cannot be over-emphasized.

In addition to providing winter respite to resident species, Jewel Cave also contains large numbers of bats that are known to travel great distances to hibernacula (Fenton and Barclay 1980). For this reason, this cave may represent an important wintering location for bats from outside the immediate Black Hills region (Worthington and Bogan 1993).

While it is unknown when bats began using Jewel Cave, it has been an important hibernaculum since at least the 1950's (Worthington and Bogan 1993). Summer use of the cave is generally limited to night roosting, though small numbers were documented using the site as a day roost (Choate and Anderson 1997). All species that hibernate there have been documented using the site as a night roost (Choate and Anderson 1997, Mattson and Bogan 1993).

The documented success of this location as a hibernaculum for such a wide variety of species is likely attributable to two factors. First is the diversity of microclimate conditions found within the cave. As has been noted, differences in hibernaculum selection among species are well documented. The present number of species attests to the range of conditions.

The second important factor is the limited level of disturbance characteristic of the site. Bat access to the cave is via the original entrance, which is gated to restrict human access (Mattson and Bogan 1993). No winter tours are conducted through hibernal areas, and access is restricted from October through April to minimize disturbance to the bat population (Kate Cannon personal communication).

Reproduction

While significant variations occur among species, there are some general characteristics common to bat reproduction that are important considerations for conservation strategies.
In general, mating occurs in the fall of the year (Racey 1982). Females store the male's sperm until spring, whereupon fertilization and implantation occur (Racey 1982). Given poor environmental conditions, females can delay fertilization, implantation, and even gestational growth of the embryo by entering torpor until conditions are suitable (Racey 1982). Increased levels of precipitation and the resultant decrease in foraging activity delay reproduction and may prevent breeding entirely in some individuals (Grindal et al. 1992, Racey 1982).

Females begin to form maternity roosts upon emergence from hibernation in the spring. Such roosts are collectives of females that may have traveled to the site from a wide area. Requirements for such sites vary by species. Two important factors are proximity to foraging areas and roost temperature (Racey 1982).

Bats generally give birth to a single altricial pup and only once a year. As during hibernation, bats are particularly susceptible to disturbance at this time. Disruption of maternity roosts can result in reabsorption of the embryo or spontaneous abortion. Disturbance at nursery roosts can result in the abandonment of non-volant pups.

In some species, nursery roosts may be completely different sites from maternity roosts. Again, proximity to foraging areas and roost temperatures are common requirements (Tuttle and Stevenson 1982). Warm roosting temperatures hasten parturition and development of the juveniles (Racey 1982). A roost's proximity to foraging areas is particularly important before the pups are volant (Tuttle and Stevenson 1982). Females have very high energy demands during this time of year. Long flights to foraging sites consume high levels of energy. Additionally, females must return to the nursery roost periodically throughout the night to nurse offspring. Once pups are volant, mastering foraging technique and accumulating body weight for successful hibernation are more efficient in areas with high insect densities close to the roosting site (Tuttle and Stevenson 1982).

Nursery roost members begin to disperse in the late summer and early fall when bats either migrate or return to hibernacula. Low reproductive rates, susceptibility to disturbance, and specific roost requirements are three important elements that underlie the need for conservation strategies and habitat management.

Night Roosts

Night roosts serve a variety of functions. One of the more important functions is to provide a resting site following a period of foraging (Kunz 1982). Generally, night roosts are found close to foraging areas and provide bats a secure resting spot for digesting and socializing (Kunz 1982).

While some individuals may be opportunistic in night roost selection, larger collective sites (e.g. caves, mines, buildings) found in the Black Hills are not atypical. Many smaller caves in the Hills are used exclusively as night roosts by several bat species. All caves and mines identified as hibernacula are also used by those same species as night roosts throughout the summer. Segregation of species at night roosts has not been observed in the Black Hills. One cave in the northern Hills [T5N R5E Sec 28] yielded all eight species known to roost underground during a single evening's netting (Tigner and Aney 1994).

Night roosts frequently contain scattered droppings throughout the interior. In addition, some species transport larger prey back to a favorite feeding perch within a night roost beneath which small piles of droppings and discarded insect parts may be found. One sheltered porch of an abandoned cabin, used as a feeding perch by *C. townsendii*, contained a piling of moth wings and other assorted insect bits that was 3 cm in depth.

Seasonally, night roosts within the interior of the Black Hills demonstrate very different patterns of use. Following hibernation, until mid-summer, bats netted at night roosts were almost exclusively adult males. This capture pattern continues until late summer when adult females and juveniles are routinely caught having returned from nursery roosts. Evidence for this movement was displayed by the recovery of a banded *M. septentrionalis* in a building nursery roost at the periphery of the Black Hills near Sturgis, SD. The closest banding site was a night roost 13 km away.

Roost Fidelity

As noted, bats require specific roosting habitat that typically are used from year to year by the same bats and successive generations. Human residents of buildings with bat maternity roosts often notice and comment upon such seasonal use when seeking assistance with roost management. Accumulations of droppings frequently attest to the repeated use of summer roosts.

Strong roost fidelity may be due to a relative scarcity of suitable sites (Kunz 1982). This may be particularly true where bats continue to roost at sites with high disturbance levels. In addition, the permanency of the structure housing the roost may affect the degree of fidelity (Kunz 1982).

An understanding of roost fidelity and its potential impact on population dynamics is an important component for habitat managers. In a recent review of the literature on this subject, Lewis (1995) presents three benefits of roost fidelity. First, sites that provide high quality roosting conditions are more likely to show persistent use. Repeated use of quality sites eliminates energy depleting searches for alternate roosting sites.

Second, sites whose conditions are improved by occupancy may demonstrate higher levels of fidelity. The maintenance of roost microclimate resulting from collective inhabitance, as found within nursery roosts, may promote roost fidelity.

The third benefit is that of maintaining social relationships with other members of the species. For females that form maternity and nursery collectives, roost fidelity can serve to facilitate the collective's formation (Lewis 1995).

In addition to nursery roosts, site fidelity to night roosts and foraging areas has been observed in species residing in the Black Hills. While only limited information on reproductive behavior has been collected for most of the region's species, some differences have been noted. *M. lucifugus* and *E. fuscus* both demonstrated strong fidelity to maternity and nursery sites.

In contrast, maternity and nursery roosts of *M. t. pahasapensis* frequently change roost sites though some evidence of reuse may indicate a fidelity to a network of roosts (Cryan and Bogan 1996).

Mattson's (1994) study of *L. noctivagans* also demonstrated frequent roost-changing activity in maternity roosts. Such activity suggests potential benefits exceed the liabilities associated with frequent roost relocation (Cryan and Bogan 1996). Benefits may include avoidance of disturbance or parasites, predator evasion, roost microclimate selection, and minimization of flight distance to foraging areas (Lewis 1995).

Predation

Little information has been reported from the Black Hills on bat predation. Mattson (1995) observed owl predation on a probable juvenile *Lasionycteris noctivagans* resting on the bole of a roost tree. The species was thought to be an eastern screech-owl (*Otus asio*) or a northern saw-whet owl (*Aegolius acadicus*).

Backlund (personal communication) identified a skull of *L. cinereus* from the pellet of what was thought to be a long-eared owl (*Asio otus*) collected 100 km east of the Black Hills. Owls are one of the more common predators cited in the literature though no predators are known to be bat specialists (Fenton 1992).

While no evidence has been collected for owl predation at larger roosting sites in the Black Hills, Tuttle and Stevenson (1982) note that owl predation may be disrupted by human presence. Direct observation of predation at caves was made only while observers were concealed within a blind using night vision equipment.

In March 1992 two *C. townsendii* were identified hibernating in the lowest chamber of a natural cave [T3N R6E Sec 29]. While droppings and nests of the bushy-tailed woodrat (*Neotoma cinerea*) were present throughout the cave, no nests were located in the chamber in which the bats were hibernating. In November 1993 two *C. townsendii* were found in the same location, and a bushy-tailed woodrat nest had been constructed in the chamber. While two bats were observed hibernating in November of 1993, only a portion of a single forearm with a small attachment of wing membrane was found during a survey conducted in February 1994. It was located near the previously mentioned nest amid pieces of collected litter, providing circumstantial evidence of possible predation by this rodent species.

Raccoons (*Procyon lotor*) were frequently found in abandoned mines during winter surveys in the Black Hills. While no direct observations have been made in the Black Hills, this species is known to prey on bats (Barbour and Davis 1969). The same authors reported frequent predation of *L. borealis* by blue jays (*Cyanocitta cristata*). Other records of predation in the Black Hills include skunk, marten, voles, snakes, and raptors (Herreid 1961, Martin 1961, Sperry 1933, Nagorsen and Brigham 1993).

One of the more common predators is the domestic cat. Given the close association between many bat species and buildings, it is not a surprising relationship. In the United Kingdom, domestic cats are considered the single greatest predator of bats (Richardson 1985).

SPECIES ACCOUNTS AND IDENTIFICATION

"Bats are such unusual creatures that some effort is required to think of them as actual animals living in a world of common sense and concrete reality." D.R. Griffin *Listening In The Dark*

The following pages provide individual descriptions of the bat species found within the Black Hills region. These include general descriptions of physical characteristics with an emphasis on points that aid in distinguishing species. For a more definitive key, see van Zyll de Jong (1985).

A brief natural history for each species is also provided. This section includes information on seasonal roosting requirements, reproduction, and range. Wherever possible, such information is based upon observations made within the Black Hills. References are made to northern and southern Black Hills. Such references indicate an area north or south of a line bisecting the region that runs through Rapid City, SD. In the interest of protecting roosting sites, specific locations to all sites referenced in this report are filed with the South Dakota Department of Game, Fish and Parks, Black Hills National Forest, and Wyoming Game and Fish Department.

In-hand identification of most Black Hills bat species is fairly straightforward. The *Myotis* species are at times difficult to distinguish owing to individual variation found within identifying characteristics. The following descriptions attempt to highlight features most common and useful in identification of species in the Black Hills. Sex determination is easily accomplished with a captive animal, as males display a conspicuous penis. Roosting posture generally prevents sex identification during hibernation when individuals cannot be disturbed.

Juvenile field identification is achieved by illuminating through the metacarpal-phalangeal joints within the wing membrane. Incomplete bone ossification at the joints in juveniles appears as translucent bubbles within the distal ends of bones. In adults, this characteristic is absent. These bubbles become less apparent with age and by summer's end are difficult or impossible to identify in juveniles born in the spring. Juvenile joints frequently give a rounded, more swollen appearance when compared with adult joints. However, given individual variation, age identification based solely on the latter of these two characteristics is likely to be less reliable.

Pelage color is not a reliable characteristic for species identification because of the substantial differences occurring within species. The exception to this rule is the Eastern Red Bat (*L. borealis*), which generally displays a pelage significantly different from other species found within the Hills.

See TABLES 3 and 4 for forearm measurements and weights.

Myotis ciliolabrum[1] (Western Small-footed Myotis)

M. ciliolabrum is the smallest bat in the Black Hills with an average forearm length of 31.27 mm and average weight of 5.72 gm. The calcar is keeled and as noted by its common name, the foot is small with average length being 6.5 mm (van Zyll de Jong 1985). The skull has a flattened appearance, and the ears are relatively long with a narrow tragus approximately half of the ear length. Though variations in color exist, it is frequently seen with near cream-colored pelage, lighter ventrally accentuated by a black mask, ears, and membranes.

M. ciliolabrum is a year-round resident of the Black Hills. Regarding behavior, it is a very gentle bat when handled properly. While this species is common, local populations are usually small in number though exceptions do occur. The largest number captured during a single evening's netting occurred at the historic entrance of Jewel Cave National Monument. On 8/5/93, Mattson and Bogan (1993) reported capturing 93 individuals, consisting of 80 males and 13 females.

At this same location, Turner (1974) reported an evening's capture of 48 individuals, 43 males and 5 females, on 7/24/68. These captures occurred within a span of three hours (Barbour and Davis 1969).

Another large group, 27 individuals consisting of 17 males and 10 females, was netted entering a cave [T3N R6E Sec 32] on 9/2/92 (Tigner unpublished data). Between 1992 and 1995, excluding the preceding references, average capture rate for this species at night roosts throughout the Black Hills was 3.5 individuals (Tigner unpublished data).

The largest known hibernation site for this species was an abandoned mine near Mystic, SD [T2N R4E Sec 33]. The site was an adit with a single southwest-facing portal. It was approximately 110 meters in length with several short drifts and rooms. This mine had been monitored since 1992 yielding consistent bat numbers during winter surveys. Totals for *M. ciliolabrum*: 1992:21, 1993:15, 1994:21, 1995:18, 1998:38. Five other species used this mine as a hibernaculum. As is common with many of the Black Hills' mines, the portal was located in unstable material and collapsed sometime during 1999. Such events serve to highlight the importance of identifying and protecting the remaining sites providing suitable bat habitat.

During foraging, the flight pattern is slow and erratic with orienting echolocation calls characterized by more rapidly emitted pulses than other Black Hills' species. Based on studies in other areas, *M. ciliolabrum* feeds primarily upon small insects, such as Diptera, Coleoptera, Cicadellids, and Trichoptera (van Zyll de Jong 1985).

This species characteristically hibernates individually, and movement is minimal. Our data indicate little change in hibernacula populations between November and February. *M.*

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[1] Earlier literature has referred to this species as *Myotis leibii* or (earlier) *Myotis subulatus* (Say bat)

ciliolabrum is commonly found hibernating in mines and caves. No clusters have been observed in the Black Hills. Martin and Hawks (1972) report finding a single crevice containing four individuals in a natural cave in the southern Hills. It is frequently found to inhabit narrow crevices but also roosts on the surface of vertical walls and from ceilings. Both behaviors have been observed simultaneously by different individuals within the same hibernaculum. Frequently forearms are splayed outward away from the body during hibernation. Such posturing behavior is likely to be thermoregulatory, designed to disperse body heat and lower body temperature (Bakken and Kunz 1988). It prefers cooler, drier hibernacula and is frequently found at the same winter sites as *C. townsendii.*

Although no exact locations of maternity or nursery roosts have been identified in the Black Hills of South Dakota or Wyoming, the region's numerous rocky outcrops and crevices seem to offer abundant summer roosting sites. Tuttle and Heaney (1974) describe nursery roosts in the Badlands of South Dakota, 115 km east of the Black Hills, as cracks and crevices in the clay and volcanic ash mixture characteristic of the area. All roosts contained up to four lactating females. Females typically give birth to a single pup (Barbour and Davis 1969), though twins were reported at a roost in the Badlands of South Dakota (Tuttle and Heaney 1974).

Adult females and juveniles were commonly netted at night roosts throughout the Black Hills during the active time of year. In contrast, from spring through midsummer, captures at night roosts of *Eptesicus* and other *Myotis* species were nearly always adult males. This suggests that females select maternity and nursery roosts in the Hills proper and do not move to other areas. Elevational gradient has been suggested as an important determinant in formation of maternity/nursery roosts (Cryan and Bogan 2000).

In contrast, *M. ciliolabrum* may be able to achieve roost thermal requirements by selecting sites that mitigate temperatures based upon other factors. One such roost, near an abandoned mine on Custer State Park, exists in rock outcroppings at an elevation of approximately 6250 feet. Towering well above the surrounding forest canopy, these rock faces have a clear southerly exposure. Summer surveys during 2001 and 2002 have included mist netting a small pool immediately adjacent to these rock faces. Both years have yielded *en masse* captures of lactating females of this species. Approximate number of bats arriving collectively at this pool was ten. Early capture times, before and at sunset, support the assumption that there is a nearby nursery roost.

Another explanation for wider distribution of reproductive females and smaller local populations may be the result of prey availability. Diet analysis might provide information concerning varying population density.

Although successfully captured at night roosts with both mist nets and harp traps, its small size often allows escape from the latter. It frequently becomes blocked and entangled by the first bank of strings, but escapes by retreating. Despite its small size, this bat is a strong flier and can take off from a level surface.

For bats in general, small access points into roosts increase predation threat and may be avoided (Tuttle and Taylor 1994). *M. ciliolabrum* demonstrates the widest acceptance of restrictive roost entry size at hibernacula. One example is based upon observations made at a natural cave on private property near Rapid City, SD. The only access into the cave was via a ceramic drain tile inserted into a solid wall built by the cave's owner to prevent unauthorized entry. The tile, approximately 15 cm in diameter and 35 cm in length, was installed to provide access to the solid door's locking mechanism. It is uncertain whether this bat can fly through this opening without landing given the narrowness and irregularity of the approach to the closure wall, which is approximately 20 m from the access point. Three *M. ciliolabrum* were the only bats identified using this site as a hibernaculum.

Based on existing information, hibernacula selection is somewhat of a paradox. High body fat to mass ratios allow large species to use relatively cold, dry hibernacula (Ransome 1990). Fat reserves serve as a buffer against harsh or fluctuating hibernacula conditions. Given its diminutive size, this species clearly does not fall into this category. However, it may be able to avoid harsh conditions by selecting crevices.

At present levels of understanding, the principle threat to this species may be the availability of suitable hibernacula. This supposition is probably true if, as has been suggested, abundant sites for maternity and nursery roosts are available throughout the natural terrain of the Black Hills. More information is needed on maternity and nursery roosts and hibernacula requirements for this species.

Reproductive females were recorded from both the northern Hills (one pregnant female on 7/5/94 at a natural cave night roost [T3N R6E Sec 32]) and southern Hills of South Dakota (one pregnant female while foraging on 7/7/95 at Lower Woodcock Spring).

Lactating females were captured on 7/14/93 (foraging, Keystone, SD sewage lagoons) and 7/7/94 (foraging, Hazelrodt Picnic Ground, SD). Post-lactating females were captured on 8/17/94 and 9/9/94.

Earliest capture of a volant juvenile was on 7/24/68 entering Jewel Cave (Turner 1974). Other captures of juveniles:
7/31/92 - eight entering natural cave [T3N R6E Sec 32]
8/05/92 - three entering natural cave [T3N R6E Sec 29]
8/27/92 - two entering natural cave [T4N R5E Sec 12]
9/11/93 - one netted foraging at Roby Spring (Mattson 1994)
9/13/94 - one entering natural cave [T5N R5E Sec 28]
9/20/94 - three entering natural cave [T3N R6E Sec 28].

Three banded individuals, 2 adult male and 1 adult female, have been recaptured. All three occurred at two natural caves in the northern Hills. Both sites serve as summer night roosts for both sexes where they were originally banded.

A banded female was found hibernating high on an interior wall in the first cave [T4N R5E Sec 16]. Height prevented reading the band. This same cave, surveyed on 3/4/93, served as a hibernaculum to a banded male found head-down in a vertical crevice in the ceiling.

A banded adult male was recaptured at the second cave [T3N R6E Sec 32] on 6/26/95. It was banded at the same site on 9/2/92. Age could not be determined at banding.

Myotis evotis (Long-eared Myotis)

M. evotis is of medium size, with an average forearm length of 38.17 mm and an average weight of 7.5 gm. *M. evotis* generally has a shorter forearm, range 36-41 mm, than *M. t. pahasapensis*, range 40-43 mm. Total ear length is a good distinguishing feature. The ear length, 17-22 mm, is substantially longer than that of *M. septentrionalis*, 15-18 mm, and proportionally longer than *M. t. pahasapensis*, 19-20 mm (van Zyll de Jong 1985). When pressed forward, ears extend a minimum of 5 mm beyond the nose tip and overall ear length exceeds 50 percent of the forearm length. *M. evotis* has variable brown pelage with contrasting blackish ears and wing membranes. Individuals caught in the Black Hills have darker brown pelage.

Very little information regarding winter hibernation exists, but *M. evotis* may use caves and mines (Manning and Jones 1989). No winter records were recorded for this species in the Black Hills. This species is found in a wide variety of habitat types though most are associated with forested areas (Manning and Jones 1989, Nagorsen and Brigham 1993). *M. evotis* forages on a variety of insects with beetles and moths comprising most of the diet (Black 1974).

This report represents an extension in range for this species as only one confirmed specimen has been reported from Harding County in the northwestern corner of the state (Andersen and Jones 1971, Jones and Choate 1978). Earlier specimens from the Hills purported to be *M. evotis* were determined to be *M. t. pahasapensis* (Jones and Choate 1978). Identification of *M. evotis* has been made in the field based on the following description by van Zyll de Jong (1985). *M. evotis* exhibits a slightly shorter forearm than *M. t. pahasapensis* with a longer overall ear length. There is no conspicuous fringe around the free edge of the uropatagium on *M. evotis*, though slight inconspicuous fringes do occur. The fringe on *M. evotis* is sparser than on *M. t. pahasapensis*. Variation in the degree of conspicuousness can make the distinction between these two species difficult. Overall ear length was used as the determining factor. Bats whose ears, when pressed forward, extended 5 mm beyond the nose tip and were greater than 50 percent of the forearm length were classified as *M. evotis*.

M. evotis was captured at night roosts in both mist nets and harp trap. Six adult males were netted at night roosts in the northern Black Hills of South Dakota (T3N R6E Sec 32; T5N R5E Sec 28; T4N R5E Sec 16), one adult female and one adult male were netted foraging over a small woodland pond in Wyoming (T55N R63W Sec 26), one male was netted over a stock tank near Jewel Cave National Monument, and one nursery roost was found near Sturgis, SD. One non-reproductive adult female was netted foraging adjacent to the Cheyenne River near Cascade, SD.

The nursery roost, comprised of approximately 20 to 25 individuals, was located (7/26/93) in the attic of an older, two-story brick building, constructed circa 1900, in Sturgis, SD. Based upon observed variation in body size, this figure includes juveniles. The bats were roosting at the edge of a large metal exhaust vent under adjacent flashing and roofing. The cluster roosting location was characterized by access to both the outside and interior of the attic, though numerous other access points were available throughout the attic.

One non-reproductive adult female was roosting approximately 1 m from the area containing the rest of the roost. Because of the longer ear length and distance from other bats, this female was captured by hand for examination. Though no measurements were taken, ear length was greater than 50 percent of the forearm length with ear tips extending well beyond the nose tip. Fringe on the free edge of the uropatagium was very sparse and finer than typically seen on *M. t. pahasapensis.* When returned to its roosting spot, the bat quickly rejoined the others. The roost was used during the summer of 1993, but not during 1994 or 1995. Extensive restoration, which included construction work in the attic coupled with simultaneous work in adjacent buildings, may have caused the bats to abandon the site.

Priday and Luce (1995) reported three captures northwest of Sundance, WY. The first was netted at a night roost in a mine on 6/20/94. The remaining pair was netted over a nearby stream approximately 400 m away.

Based on capture data this bat appears to be less abundant than *M. t. pahasapensis.* Range for this species includes all of the Black Hills.

Myotis lucifugus (Little Brown Myotis)

M. lucifugus, medium in size, has an average forearm length of 37.49 mm and an average weight of 8.33 gm. The calcar is not keeled, which helps distinguish it from *M. volans*. Ear length is less than *M. septentrionalis* and does not extend beyond the nose tip when pressed forward. Tragus is blunt and approximately half the length of the ear. Pelage color varies considerably ranging from light or medium brown to very dark brown, and it displays a characteristic glossy appearance that helps distinguish it from morphologically similar *M. volans*. Wing membranes and ears are dark brown.

M. lucifugus is common throughout the United States and abundant in the Black Hills. It is the current record holder for age longevity at over 34 years (Davis and Hitchcock 1995). This is one of the more opportunistic species both in foraging habits and roost selection (Fenton and Barclay 1980). It can be found in a variety of habitat types and is known to roost in buildings, caves, mines, and trees (Fenton and Barclay 1980). *M. lucifugus* commonly feeds flying low over water surfaces with a shallow wing beat. Aquatic insects comprise a large portion of this species' diet (Fenton and Barclay 1980).

As is common with many Vespertilionid species, males roost individually or in small groups during the summer months, segregated from females. As evidence of its opportunistic roosting characteristics, maternity roosts are now found more commonly in buildings than in natural roosting sites (van Zyll de Jong 1985). Trees also function as nursery roosts (Fenton and Barclay 1980). In the Black Hills, all known maternity and nursery roosts are in buildings.

Females give birth to a single pup with juveniles becoming volant at three weeks of age (Fenton and Barclay 1980). The earliest volant juvenile was captured (7/4/70) near Custer, SD (Turner 1974).

Four maternity and nursery roosts were identified in both the southern and northern Black Hills and all show signs of a high degree of roost fidelity. All known maternity and nursery roosts for this species are located within 0.5 km of water. In South Dakota, the largest known maternity roost is located within an attic at a camp near Custer State Park. This roost was first recorded during the summer of 1970 when it contained 100 to 150 adults (Turner 1974). It was estimated to contain 200 bats during the summer of 1993 (Mattson 1994).

Another large maternity roost of approximately 100 individuals was identified in the gable of a two-story wood framed house near the McNenny Fish Hatchery in South Dakota (6/15/93). Because of extensive renovations, including complete removal of the roof, approximately 100 bats were hand-captured from behind a shutter (6/24/93). Thirty-five individuals, all pregnant females, were banded.

A separate roost containing 20 pregnant females was found on the same evening approximately 1 km from the first roost in the attic of a building at McNenny Fish Hatchery. The fourth maternity roost was located in the attic of a two-story brick building near Sturgis, SD on 7/26/93.

Approximate roost size was 50 adults. Surveyed the following year (7/6/94), this roost contained approximately 100, which included some juveniles of the year.

With the exception of *C. townsendii*, this is the only species for which a probable maternity roost was identified underground. On 5/20/93, a natural cave [T5N R5E Sec 28] near Sturgis, SD contained a cluster of 18 individuals including three banded females. Bands were not read to minimize disturbance to the roost. The cluster was located in a dome above a main passage. Females of this species were banded previously at this location.

M. lucifugus was routinely captured with mist nets and harp traps at night roosts throughout the Hills though usually in smaller numbers. During the spring and early summer, captures at night roosts were comprised almost exclusively of adult males. Absence of females at this time was likely due to their congregation in maternity and nursery roosts in other areas (Turner 1974). Juveniles and adult females became more common at night roosts during the latter part of the summer and early fall.

Two caves in the northern Hills yielded the highest capture rates [T3N R6E Sec 32 and T5N R5E Sec 28]. Both sites were monitored during the winters of 1992-1995 with neither being used as a hibernaculum. On 9/2/92, the former of the two caves yielded 25 individuals, 20 males including one confirmed juvenile and 5 females; while the latter, on 9/9/92, yielded 29 individuals, 15 males and 14 females.

The largest number of *M. lucifugus* netted at a foraging site was 42, netted within one hour at the sewage lagoons of Jewel Cave National Monument. These bats probably were a portion of the bats known to roost at Jewel Cave (Cryan and Bogan 1996).

This bat hibernated only in sites, including natural caves and mines, with relatively high humidity including natural caves and mines. During hibernation, individuals were often found with droplets of condensation covering the entire body. Conservation strategies for this species should include protection of hibernacula that contain relative humidity greater than 90 percent.

The largest hibernaculum in the Black Hills for this species is Jewel Cave. The Dungeon Room, in which most of the *M. lucifugus* and *M. volans* hibernate, has a relative humidity of greater than 90 percent. These species apparently arrive at similar times with numbers remaining constant throughout the winter season.

Another natural cave [T3S R2E Sec 3], 15 km north of Jewel Cave, also serves as a hibernation site to large number of *Myotis*. Approximately 300 individuals were recorded during winter surveys with largest numbers comprised of *M. lucifugus* and *M. volans*. These two species are morphologically similar and, owing to the height of hibernating clusters, not easily distinguishable during hibernation. Based upon observations at this site, *M. lucifugus* and *M. volans* have a tolerance to limited roost entry size. While the area in which the bats hibernate is spacious, ceiling height is approximately 3-4 m, access requires flight through a narrow passage.

Given *M. lucifugus*' affinity for roosting within man-made structures, its maternity and nursery roosts may be at greater risk than species relying upon more natural roosting sites. As such, this species is one that should benefit from an increased public awareness of bat-related benefits.

Myotis septentrionalis[1] (Northern Myotis)

M. septentrionalis is a medium-sized bat with an average forearm length of 36.07 mm and average weight of 7.13 gm. It is distinguished primarily by its ear length and tragus. Average overall ear length is 16.4 mm. This measurement is greater than *M. lucifugus*, 13.8 mm, and *M. volans*, 13.3 mm, but less than *M. thysanodes*, 19.5 mm, and *M. evotis*, 19.8 mm (van Zyll de Jong 1985). The tragus is long, narrow and pointed at the tip. *M. septentrionalis* frequently has a mask that is balder than similar *Myotis* species. Membranes, ears, and mask are generally medium to dark brown.

This bat routinely displays an aggressive attitude when netted. It constantly vocalizes from time of capture until release and frequently attempts to bite handlers. Illumination by flashlight in roosts often caused this bat to vocalize. Little information exists on the dietary preferences of this species, though some evidence indicates it is a generalist (Nagorsen and Brigham 1993, van Zyll de Jong 1985).

M. septentrionalis is abundant throughout the Black Hills region, however, winter occurrence was only recently confirmed (Worthington and Bogan 1993, Tigner and Aney 1993, Tigner and Aney 1994). During the course of one survey conducted between 1992 and 1995, only seven hibernating individuals were located (Tigner and Aney 1994, Tigner unpublished data). There are no hibernation records in Wyoming (Bob Luce personal communication). Sites selected for hibernation include both natural caves and abandoned mines.

Several common features are exhibited by all hibernacula where this species was identified. The first of these is relative humidity greater than 90 percent. All sites contained standing water with condensation accumulations throughout the interior. All bats roosted in crevices protected from normal airflow. Of the seven hibernacula identified for this species, five are abandoned mines and two are natural caves. The abandoned mines, varying in interior length from 5 m to 200 m, contained short side drifts off the main adit, and all possessed standing interior water. Crevices in these side workings were selected for hibernation. Three of the adits were completely sealed by snowfall.

The first natural cave hibernaculum [T5N R5E Sec 29], near Sturgis, SD, has a history of human disturbance. Shortly after this site was identified, in mid-December 1994, a campfire was built in the cave's interior causing abandonment by hibernating bats for the balance of the winter.

There were only a handful of records noted from the second natural cave, Jewel Cave National Monument (Worthington and Bogan 1993). Given this species' apparent affinity for crevices, they may be easily overlooked.

The largest number of *M. septentrionalis* found in a hibernaculum in the Black Hills was in an extensive abandoned mine near Hill City, SD. A conservative count from the survey conducted

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[1] Earlier literature referred to this species as *Myotis keenii* or Keen's Myotis.

during the winter of 2002/03 was forty individuals. Similar numbers were found during the site's initial survey during the winter of 2000/01.

M. septentrionalis was captured at night roosts with both mist nets and harp trap. Only one maternity or nursery roost was located within the Black Hills (Tigner and Aney 1994). It was identified on 7/6/94 near Sturgis, SD in the roof apex of a two-story brick building. The roost was comprised of approximately 75 individuals, including juveniles. A banded adult female was observed within the main cluster, but to minimize disturbance the band was not read. The closest banding site was a cave night roost approximately 13 km from this location. The observation of this banded female provides direct evidence of movement from within the Hills to a maternity roost at the periphery. This identification also indicates the possibility of locally important maternity and nursery roosts.

A small maternity and nursery roost containing less than 10 individuals was identified in Wall, SD (6/30/94) approximately 110 km east of the Black Hills. One lactating female was captured emerging from beneath shake roofing of a United States Forest Service visitor center building.

Thirty-seven juveniles were netted between 1992 and 1995 (Tigner unpublished data). During a single evening's netting (8/18/92), 13 of these juveniles, 12 males and 1 female, were captured entering a natural cave night roost [T4N R5E Sec 16]. Seasonally, the earliest volant juvenile was captured on 7/26/68 entering a natural cave night roost [T2S R2E Sec 17] (Turner 1974). Three male juveniles were netted entering a cave [T3N R6E Sec 32] on 7/31/92.

Two post-lactating females were netted during the summer of 1994. On 7/29/94, the first was netted while foraging at Red Bat Pond near Whitewood, SD. The second was captured on 8/11/94 entering a natural cave night roost [T4N R5E Sec 12].

Based on banding recoveries, *M. septentrionalis* demonstrates fidelity to night roosts. Eleven banded individuals were recaptured. Seven recaptures occurred at night roosts where the bat was originally banded. Seven of these, all males, were banded as juveniles. The table below gives the banding/recapture dates of these seven *M. septentrionalis*.

Site Type	**Original Band Date (Age at Banding)**	**Recapture Date (Age at Recap)**	**Time Length**
Night Roost (Cave)	8/12/93 (J)	9/2/93 (J)	21 days
Night Roost (Cave)	8/18/92 (J)	9/16/92 (J)	29 days
Night Roost (Cave)	9/30/93 (J)	7/5/94 (A)	9 months
Night Roost (Cave)	9/2/92 (J)	8/12/93 (A)	11 months
Night Roost (Cave)	8/27/92 (J)	6/17/94 (A)	1 year 10 months
Night Roost (Cave)	9/9/92 (J)	8/24/99 (A)	6 years 9 months
Night Roost (Cave)	7/31/92 (J)	6/7/02 (A)	9 years 10 months

Myotis septentrionalis **- NIGHT ROOST SAME SITE BANDING RECAPTURES**

The remaining four recaptures may be evidence for fidelity to a specific foraging area or a home range. One male juvenile has been netted on three occasions at the same stock tank. Captured and banded on 5/31/94, Mattson (personal communication) recaptured this bat at the original site 24 days later. This same individual was again recaptured on 7/11/99 at the same location (Tanya Dewey personal communication). The table below gives the details of these four recaptures.

Site Type	**Original Band Date (Age/Sex at Banding)**	**Recapture Date[s] (Age at Recap)**	**Time Length**
Stock Pond	8/11/92 (J / Male)	6/23/94 (A)	1 year 10 months
Stock Pond	7/29/94 (A / Female)	8/8/01 (A)	7 years
Stock Tank	5/31/94 (J / Male)	6/24/94 (A) 7/11/99 (A)	5 years 1 month

Myotis septentrionalis **- FORAGING SAME SITE BANDING RECAPTURES**

More information is required on this species' maternity roost habits. If characteristically found in buildings, this species would benefit from increased public awareness of maternity roost importance. Hibernacula requirements need further study. Given its affinity for crevices, quantifying microclimate conditions as well as physical characteristics are two areas that warrant further examination.

Myotis thysanodes pahasapensis (Fringed Myotis)

M. t. pahasapensis is a bat of medium size. The average forearm length is 40.82 mm and average weight is 7.8 gm. The common name is derived from the fringe of conspicuous coarse hairs around the free edge of the uropatagium. As with many identifying characteristics, the fringe is variable. Fringe variation does not appear to be age or sex related. Ears are longer than other *Myotis* species with the notable exception of *M. evotis*. *M. evotis* displays a proportionately longer ear than *M. t. pahasapensis* with overall ear length in *M. t. pahasapensis* being less than 50 percent of the length of the forearm length (van Zyll de Jong 1985). Wing membranes and ears are very dark to black in color.

M. t. pahasapensis' known range is restricted to the Black Hills area (Jones and Choate 1978, van Zyll de Jong 1985). Beetles represent the largest prey group but moths are also taken (Black 1974). Though a year-round resident, hibernating individuals were difficult to locate. Because of a preference to hibernate in cracks and crevices, identification is difficult. During the course of surveys conducted during the winters of 1992-1995 a total of 10 individuals were found hibernating. Five of these were in abandoned mines, roosting individually in crevices. The remaining five, three of which were in crevices, were found in natural caves roosting individually.

Identification during hibernation was made via the longer ear length, black membranes and overall size. Two individuals were roosting on the surface of the rock. One was hanging, head down, approximately 1.5 m from the floor on a vertical surface. The other was hanging by both feet from the ceiling, in a room with maximum height of about 1 m, and swaying with the air currents in the cave [T6S R5E Sec. 21].

The largest number of this species found hibernating in a single location is four. This site, surveyed in January 2003, is an extensive abandoned mine in the central Black Hills containing standing water throughout its interior. Three were located in crevices in the rock and one was found on the rock's surface.

Given the comparatively low number of hibernating individuals discovered during the course of the survey, it is likely this species prefers cracks and fissures for hibernation. Such behavior would make location and identification difficult during winter. Martin and Hawks (1972) reported finding this species in Jewel Cave, frequently hibernating while facing up a vertical wall and clinging to the surface by both feet and thumbs with forearms slightly splayed away from the body. As noted under *M. ciliolabrum*, such posturing may be thermoregulatory in nature employed to lower body temperature by increasing exposed surface area (Bakken and Kunz 1988).

Using radio telemetry Cryan and Bogan (1996) conducted the most comprehensive study of roosting habitat in the southern Black Hills during the summer of 1995. Males were found to roost individually in rock crevices. Reproductive females formed communal maternity and nursery roosts in rock crevices that averaged 18.9 individuals (Cryan and Bogan 1996).

Maternity roosts were observed to move daily with evidence suggesting the possibility of roost fidelity (Cryan and Bogan 1996).

Maternity roosts have been identified in the northern and southern Black Hills of SD (Cryan and Bogan 1996, Tigner and Aney 1994). One maternity roost was in an attic of a two-story brick building near Sturgis, SD (6/8/94). The roost was divided into two locations approximately 1 m apart. The first was around an exhaust fan midway between the apex of the roof and the floor of the attic. The exhaust fan was located at the juncture of two wings of the building. The second location was closer to the floor, still at the roofline, around a black, cast iron standpipe. Both groups appeared to be of similar size, about 25 individuals, with apparent juveniles in both. Both sites appeared to contain crevices large enough for bats to exit. No bats were found during a survey of this site on 7/6/94. A single adult female was observed in this attic on 4/22/94, although no evidence of previous parturition or pregnancy was noted.

The earliest recorded volancy for juveniles is 8/10/68 (Turner 1974). This date compares well with later survey work which netted three juveniles entering a natural cave night roost [T4N R5E Sec. 16] on 8/18/92 and one juvenile netted foraging near Lower Woodcock Spring on 8/24/94.

M. t. pahasapensis was captured at night roosts with both mist nets and harp traps. The largest number caught during a single evening (5/22/94) was 12 adults, 11 male and 1 female, netted at Water Draw Spring (Mattson 1994). In addition, eight adult males and two non-parous females were netted entering a natural cave night roost [T3N R6E Sec. 32] on 9/2/92. One other significant series of captures occurred at a night roost [T4N R5E Sec. 12] on 9/10/93. Within a span of 15 minutes, eight individuals were netted entering the natural cave. Seven were post-lactating females and one an adult male displaying canines with greater than 50 percent wear. Effective closure of all openings at the cave's mouth was not achieved and other individuals were seen entering the cave at this same time.

A total of 75 individuals, 54 males and 21 females, were banded between 1992 and 1995. Five recaptures of banded individuals all occurred at the original banding site, all serving as night roosts. The first recapture was at a cave [T4N R5E Sec 16]. This juvenile male was originally banded on 8/18/92 and weighed 7 gm. He was recaptured on 9/16/92 and weighed 10 gm. Though recaptured within 30 days, the substantial difference in weights may be attributable to recent foraging activity rather than body weight increase.

The second recapture was at a natural cave night roost [T2S R2E Sec 16]. An adult male was banded on 8/25/94 and recaptured on 6/20/95. The three other recaptures all occurred at the same location, a natural cave [T3N R6E Sec 32] serving as a night roost. The first occurred on 5/20/94 and the last two on 9/20/94. All three adult males were originally banded on 9/2/92.

Males, when netted, were frequently found to have dirt or clay-like substance within their fur and crevices of the wing membrane. Such deposits were clearly present in early evening captures suggesting day roosting in soft soil crevices. One banded male, recaptured at a night roost, was found to have dirt between the band and the forearm. This dirt easily fell away when the band was manipulated for removal and likely would have been groomed away by the bat. No evidence of injury to the bat was noted.

The following table provides a listing of band recaptures for this species.

Site Type	**Original Band Date (Age/Sex at Banding)**	**Recapture Date[s] (Age at Recap)**	**Time Length**
Night Roost (Cave)	8/18/92 (J / Male)	9/16/92 (J)	29 days
Night Roost (Cave)	9/2/92 (J / Male)	9/20/94 (A)	2 years
Night Roost (Cave)	9/2/92 (J / Male)	5/20/94 (A)	1 year 9 months
Night Roost (Cave)	9/2/92 (J / Male)	9/20/94 (A)	2 years
Night Roost (Cave)	8/25/94 (A / Male)	6/20/95 (A)	10 months

Myotis thysanodes pahasapensis **– Band recaptures**

Additional information is needed on the maternity and nursery roost requirements and habits for this species. While roosts were found in a building and natural rock crevices, additional roost locations would improve the knowledge base. The frequent relocation of maternity and nursery roosts is unreported in this species and warrants additional study (Cryan and Bogan 1996). Locally abundant captures may be indicative of unique or significant habitat characteristics (Cryan and Bogan 1996). Such characteristics should be identified.

Hibernacula requirements also need to be quantified. It is far more commonly identified during the summer months than in winter hibernation. While this is likely in response to an apparent affinity for winter roosting in crevices, the extent to which these sites are available to this species remains unknown.

Myotis volans (Long-legged Myotis)

M. volans is a medium-sized bat with an average forearm length of 37.93 mm and average weight of 7.84 gm. It is very similar to *Myotis lucifugus* in morphology, and the two are often difficult or impossible to distinguish during hibernation. It often displays a more dense line of furring extending from elbow to knee on the ventral surface of the wing membrane. The calcar has a distinct keel that is weak or lacking in *M. lucifugus*. Pelage is varying shades of dull brown lacking the characteristic glossy sheen often found on *M. lucifugus*. The ears and wing membranes are generally darker brown. Owing to its shorter rostrum and steep forehead (van Zyll de Jong 1985), this species can occasionally be distinguished from *M. lucifugus* by the smaller appearing head relative to body size (Tigner personal observation).

M. volans is one of the more common species found within the Black Hills and is a year-round resident. It demonstrates close associations with coniferous forests, foraging throughout the forest canopy (Barbour and Davis 1969) and feeds primarily on small moths (Whitaker, Jr. J.O. et al. 1981*a*). Roosting sites identified in the Black Hills include caves, mines, trees, and rock crevices (Cryan and Bogan 1996, Tigner and Aney 1994).

This bat hibernates in abandoned mines, but larger numbers were observed in natural caves with the largest known population wintering in Jewel Cave. As with *M. lucifugus*, this species seems to have an affinity for hibernacula with higher humidity. It is often found with droplets of condensation covering the entire body while hanging from a wall or ceiling. The largest concentration in Jewel Cave was in the Dungeon Room where relative humidity is typically greater than 90 percent.

As noted under *M. lucifugus*, another natural cave [T3S R2E Sec 3], 15 km north of Jewel Cave, also serves as a hibernation site to a large number of *Myotis*. Approximately 300 individuals were recorded during winter surveys at this site, the largest proportion being *M. volans* and *M. lucifugus*.

While two studies suggested *M. volans* to be the most common bat in the Black Hills (Turner 1974, Mattson and Bogan 1993), there is little information on its reproductive habits. In the southern Black Hills, adult males have been found to roost solitarily during the summer in rock crevices in the southern Black Hills (Cryan and Bogan 1996). They are segregated from females, which congregate in maternity roosts. Only two maternity and nursery roosts have been identified for this species in the Black Hills.

The first roost was located in the attic of an H-shaped two-story brick building near Sturgis, SD. The roost identified on 7/6/94 was comprised of approximately 75 individuals, including non-volant pups. The pups were in one cluster of approximately 30 individuals, which included adults, near the apex of the roof against a brick wall. The remaining bats roosted in smaller clusters or individually throughout the same wing of the building. Two volant juvenile males, one volant juvenile female, and three lactating females were captured by hand for cursory examination. All juveniles flew without apparent difficulty when released.

It is thought most maternity roosts are probably located in tree cavities (van Zyll de Jong 1985). Using radio telemetry, one lactating female was located, in the southern Black Hills, roosting beneath the bark of a large snag, dbh 66 cm, with at least six other bats (Cryan and Bogan 1996). These bats were observed emerging, but species of the others was not confirmed.

M. volans was successfully captured at night roosts in both mist nets and harp traps. Excluding the volant juveniles mentioned above, the earliest capture of a volant juvenile was on 7/31/92 when 2 male juveniles were netted entering a natural cave [T3N R6E Sec 32] serving as a night roost. Of the 13 juvenile captures between 1992 and 1995, 11 were at night roosts in natural caves with the remaining 2 netted while foraging at Lower Woodcock Spring on 8/24/94. Turner (1974) reported a juvenile capture on 9/7/67 at Roby Springs, SD.

One juvenile netted at a cave [T4N R5E Sec 16] was banded on 8/18/92 and was recaptured at the same site on 9/16/92. Only two other recaptures were recorded during this study. Both occurred at the same cave [T3N R6E Sec 32] on 9/30/93. The first recapture was an adult male originally banded at this site on 9/2/92, and the second, an adult male, was banded at the site earlier in the season on 9/2/93.

Additional information is needed on maternity and nursery roost requirements for this species. Absence of reproductive females from many summer foraging areas throughout the Black Hills suggests the possible occurrence of important roosting sites or areas with certain characteristics (Cryan and Bogan 1996). Such concentrations may also make this species more susceptible to habitat changes. Identification of specific habitat requirements is necessary for development of effective conservation strategies.

Corynorhinus townsendii pallescens[1] (Townsend's Big-eared Bat)

C. townsendii is a medium-sized bat with an average forearm length of 44.31 mm and an average weight of 11.59 gm. Pelage color varies but is generally buff with paler dorsal fur. When awake, this bat is easily identified by its very long rabbit-like ears and characteristic bulbous nose lumps. However, when roosting, especially in torpor and during hibernation, the ears may be much less visible. It folds its long ears back leaving only the pointed tragus erect. The erect tragus gives the appearance of *Myotis*-like ear tips resulting in frequent misidentification by the untrained observer. Record for longevity for this species is greater than 21 years (Perkins 1994).

Observed evening emergence time varied but generally occurs at lower light levels, approximately 30 minutes after sunset. It was commonly observed exhibiting "light sampling behavior"[2] prior to evening emergence from caves and mines. Such behavior is likely designed to insure light levels are low enough to minimize risks of predation by diurnal birds of prey (Erkert 1982).

Diet studies found *C. townsendii* to be a lepidopteran specialist whose diet is comprised almost exclusively of moths (Pierson et al.1999). It is known to forage within a wide variety of habitat types (Pierson et al.1999) and has been observed foraging within the canopy at wooded edges within the Black Hills.

This species was observed repeatedly returning to a feeding perch to feed on captured prey. Two such examples were documented in the Black Hills. The first was within a sheltered entry porch of an abandoned cabin and was used simultaneously by at least two individuals. The quantity of insect wings at this site indicated repeated use. The moth wing pile was 3 cm at its greatest depth. The second location was a cave serving as a night roost. Both sites were characterized predominantly by accumulation of moth wings and fecal droppings immediately beneath the perch.

C. townsendii is the most frequently encountered underground species within the Black Hills. To some extent, this is attributable to the species' propensity to roost on the surface of the rock and often in the twilight areas between daylight and darkness. Such roosting behavior allows for easy discovery compared with species that roost in crevices or darker areas. Such roosting selection also renders this species particularly susceptible to casual human disturbance. *C. townsendii* is the only Black Hills' bat dependent year-round upon underground roosting sites. Nearly all records of this species in the Black Hills are from natural caves or abandoned mines.

C. townsendii demonstrates a high degree of roost fidelity (Humphrey and Kunz 1976). This was observed in the Black Hills as this species accounted for more than half of the recaptures of

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[1] formerly *Plecotus townsendii pallescens;* see Frost and Timm 1992 and Tumlison and Douglas 1992 for a discussion of systematics.

[2] term coined by Twente (1955) for intermittently repeated return flights of short duration from roosting location within the cave/mine to the outside access point.

individuals banded between 1992 and 1995. Most were recaptured at the original banding site. *C. townsendii* is also considered sedentary and not known to migrate great distances between seasonal roosts (Kunz and Martin 1982). Some males were found to inhabit the same roost on a year-round basis.

In the Black Hills, males tend to roost individually during the summer months although small groups of three have been identified. One of the more common roosting sites for individual males is within short adits, 2 to 3 m in length, which are numerous throughout the region.

During seasonal surveys (four per year), two banded adult males were monitored at two separate abandoned mine adits. The first of these bats was monitored for three consecutive years and was absent during two fall surveys. The second was a year-round resident for two years and absent from one fall survey. Location within the mine is generally in the same vicinity during each survey. Individual males were observed within the same hibernaculum often at the same location during consecutive winters.

C. townsendii generally occurs in smaller numbers at night roosts. The largest number captured during an evening's trapping was five individuals. The site of these captures is also a large hibernaculum during the winter months. Netting surveys are likely to under-represent this species owing to its ability to avoid obstacles. *C. townsendii* also echolocates very softly and can be difficult to detect in the field on ultrasonic echolocation detectors.

While *C. townsendii* was successfully captured at night roosts with both mist nets and harp traps, the former were generally a more effective capture tool. When approaching a night roost staged with a harp trap, this species was observed turning back and departing resulting in no captures for the evening. Most individuals captured in the harp trap were males either entering at a high rate of speed or attempting to exit following day roosting at the site. Mist netting in open areas such as over water sources and meadows adjacent to forested areas rarely results in captures of this species. Most captures are achieved at restricted roost access points.

Females form maternity and nursery roosts and give birth to a single pup (Pearson et al. 1952). Roost fidelity to nursery roosts is high (Pearson et al. 1952). Only three confirmed maternity and nursery roosts were identified in the Black Hills. During the summer of 1993, two were located in natural caves in the northern Black Hills with the largest comprised of approximately 110 individuals, including juveniles. The other roost contained approximately 30 to 35 individuals during 1993 and 1994.

In circumstances unusual for this species, a nursery roost comprised of approximately 30 individuals was reported in an abandoned building south of Hot Springs, SD near Cascade Creek in July 1996 (Cryan 1997). Subsequent surveys yielded only isolated individuals using this location as a day roost with some evidence of night roosting (Luce 1998). High levels of disturbance associated with road construction in the immediate vicinity may have contributed to its abandonment as a maternity and nursery roost (Cryan 1997). Additionally, during the summer of 1998, a feral cat was observed entering the area of the building previously used by the bats (Tigner unpublished data).

Mattson (1994) cites reference to a shelter cave within Wind Cave National Park containing 50 females in 1991 and 75 females in 1992. This roost was not used during 1993 possibly due to cooler weather conditions (Mattson 1994). Additionally, a small natural cave on private land east of Custer State Park reportedly contained approximately 40 individuals one day in early August of 1994 (Travis Vickers personal communication). This site was also a small shelter cave that gave no evidence of long-term use.

One maternity roost was identified in an abandoned mine near Rochford, SD during the summer of 2002. Approximate size of this roost was 20 adults. This site also serves as a hibernaculum for the species. This is the first abandoned mine documented to serve as a maternity roost in the Black Hills.

Known maternity and nursery roosts in the region are characterized by spacious interiors with inaccessible domes or wide expanse ceilings. Both characteristics have been noted in nursery roost selection in other areas (Pierson et al.1999). These sites have fewer signs of disturbance than typically seen at caves in the region due probably to more remote location and difficulty of access. Access points for both roosts are large openings in vertical rock faces that overlook steep perennial drainages.

Extensive cave and mine surveys in the southern Black Hills failed to locate maternity and nursery roosts (Mattson and Bogan 1993). Unknown roosting locations, coupled with the absence of females from netting surveys, have been termed the greatest mystery of bats in the southern Hills (Cryan and Bogan 1996). Given this species' sedentary lifestyle and the existence of one of the largest hibernating populations in the western United States at Jewel Cave National Monument, maternity roost location remains one of the most important questions to be answered.

Parturition dates for *C. townsendii* vary substantially in the Black Hills. Weather conditions can have a detrimental effect on breeding success (Grindal et al. 1992). The first nursery roost was identified on 6/6/92 and was comprised of 75 individuals, including juveniles. At this time, juveniles were capable of flight, which would place parturition in early to mid-May (van Zyll de Jong 1985, Pearson et al. 1952). The preceding winter was comparatively mild for the Black Hills with prolonged warm spells. These milder climatic conditions likely contributed to the early parturition date.

In contrast, the spring and summer of 1993 were unusually cool and wet. Surveys of the same site yielded very different results. To minimize the disturbance associated with roost entry, the cave's access was monitored using a night vision scope (6/20/93). Approximately 10 individuals were counted emerging. A physical survey of the site was conducted (8/20/93) when approximately 110 individuals, including juveniles, were identified. There was a wide range of juvenile ages, the majority of which were non-volant. At least five with minimal furring were seen clinging to their mothers placing them at less than one week in age and moving the parturition date into August (Pearson et al. 1952). Unfavorable climatic conditions probably contributed to the delayed parturition. Survival of the first winter for young is estimated to be approximately 50 percent (Pearson et al. 1952). Such a late parturition date within the Black Hills almost certainly resulted in much higher, if not complete, mortality for the pups born in 1993.

The second maternity roost, found on 6/3/93, contained approximately 35 adults. The site was still in use on 7/7/93 and nearly the same size. There were no apparent juveniles, and all present were in torpor. The roost was abandoned by early August of 1993. Netting at the site on 9/3/93 yielded the capture of 11 adults, 1 male and 10 females. Seven of the females were parous though not lactating and three were non-reproductive. No juveniles were netted.

As further evidence of a poor reproductive season, no juvenile *C. townsendii* were captured in the Black Hills during summer 1993.

There were substantial increases in disturbance during the maternity season of 1994 at both sites. The larger of these two roosts was monitored (without entry) on 6/18/94 when approximately 15 individuals were observed exiting. At least two separate groups of people visited this site during July 1994. Vehicles and climbing equipment (required for access) were observed at the site on both occasions. An internal survey of the site conducted on 8/20/94 found no bats or evidence of use.

The second site was surveyed on 7/22/94. As no *C. townsendii* were observed within the cave, it was netted that same evening. In addition to other species, 27 *C. townsendii* were netted, the majority within a 30-minute period approximately one-half hour after sunset. Early captures were banded. This group consisted of seven lactating females, one probable pregnant female, six non-reproductive adult females, and one adult male. The balance were females, which were released upon extraction from the net to minimize further stress. The nets were closed to avoid stressing pregnant females. The early arrival of these bats *en masse* suggests another unknown roosting site in the area. To date this location has not been found. A daytime survey was conducted one week later when approximately 30 individuals were seen in a cluster. Upon observing the cluster, we immediately withdrew to minimize disturbance.

A survey of the site (8/20/94) found evidence of human activities. A large campfire was constructed within the cave's interior resulting in heavy deposition of soot on the upper portion. Three dead non-volant juvenile *C. townsendii* were on the floor of the cave. No adults were found. Such episodes highlight the need to protect important sites by restricting access during sensitive times of the year, as well as increasing public education.

In addition to this site's importance as a maternity roost, it also serves as the largest hibernaculum for this species in the northern Black Hills. During the first winter survey (1/27/94), there were 35 individuals. A survey conducted on 2/22/95 followed a week of unseasonably warm temperatures, and only seven individuals were identified. A survey conducted on 2/22/96 found 37 individuals (Oscar Martinez personal communication). In addition to its importance for *C. townsendii,* this site serves as the largest hibernaculum for *E. fuscus.* High levels of disturbance have continued at this site. As a result, this site has been lost as significant bat habitat.

Hibernacula are generally cooler and drier than for most *Myotis* species and include both mines and caves. *C. townsendii* seems to prefer hibernacula with a temperature below 10°C (Pierson et al.1999). Lone individuals were identified hibernating throughout an entire winter in mine adits as shallow as 3 m in length. Such sites had recorded temperatures below 0°C.

One of the largest hibernating populations in the western United States is at Jewel Cave National Monument (Choate and Anderson 1997). Recent surveys found approximately 800 to 900 individuals (Choate and Anderson 1997). Hibernation surveys at this site identified this bat roosting individually and within clusters comprised of up to 90 to 100 individuals (Worthington 1992). Arrival generally begins in early October, with peak numbers being observed in late December (Mattson and Giannuzzi 1994). Depending upon weather conditions, departure generally occurs during April (Mattson and Giannuzzi 1994).

Similar patterns of arrival and dispersal are known at other hibernacula within the Black Hills. To prevent disturbance, access to hibernation areas within Jewel Cave is restricted during this period (Kate Cannon personal communication). Though varying methods of census were employed during winter, numbers at this site are not thought to be decreasing (Choate and Anderson 1997). More systematic surveys were used in recent counts. If continued, such surveys should serve to improve census accuracy and permit trend assessments (C. Giannuzzi and T.A. Mattson personal communication).

The only evidence of use by *C. townsendii* during the summer months in Jewel Cave is night roosting by adult males (Choate and Anderson 1997, Mattson and Bogan 1993). A male banded on 7/25/93 by Mattson was observed hibernating within Jewel Cave during two surveys conducted during the winter of 1993-94 (Tigner personal observation, C. Giannuzzi personal communication).

No other bats banded between 1992 and 1995 were observed wintering in Jewel Cave. It should be noted, however, that clustering commonly observed in *C. townsendii* and some *Myotis* species could easily mask the presence of a band. Given the lack of information on dispersal of this population, the possibility of migration to the site by bats from northern portions of the Black Hills cannot be eliminated. Should future research uncover a migration, the viewpoint of *C. townsendii* as a sedentary species might require reevaluation.

The second largest hibernating population was located in a natural cave 14 km north of Jewel Cave National Monument. This cave contained 300 hibernating individuals during the winter of 1994-95. Though located in close proximity to Jewel Cave, no evidence of common roosting was observed. Those *C. townsendii* banded at the site during summer also were observed hibernating here.

In recognition of this site's importance, it has recently been placed within a no treatment buffer zone by the United States Forest Service to minimize potential disturbance to the roost. A gate was installed to protect this site from disturbance during the winter. No historical data on population size for this site were located.

Supporting evidence of this species' sedentary lifestyle, most recaptures of banded individuals were recovered at the original banding site. All hibernacula with banded bats whose band could not be read served as night roosts where banding occurred previously.

Five exceptions to same location band recovery were recorded. The first of these was an adult male originally banded on 8/27/92 while entering a natural cave night roost [T4N R5E Sec 12]. It was observed hibernating in another cave [T4N R5E Sec 16] on 2/16/94. The distance between these two sites is approximately 5.5 km. During the same survey on 2/16/94, a banded adult female was found hibernating. She was banded on 8/20/93 at a natural cave serving as a nursery roost. The distance between these two locations is 4 km.

While surveying a privately owned cave on 1/12/94, a banded female was identified. Though the band number could not be read, this cave is approximately 1 km from the cave serving as a nursery roost for this species. Given the close proximity of these two sites, it is likely this female was banded at the nursery roost on 8/20/93.

Finally, on 9/13/94 an adult female was captured and banded at a natural cave [T5N R5E Sec 28] being used as a night roost. She was observed hibernating in a mine adit [T3N R5E Sec 18] on 2/1/95. The distance between these two sites is 17.5 km (11 miles) and is the longest distance for a band recovery for any bat species within the Black Hills.

A survey of an abandoned mine known to serve as a *Corynorhinus* hibernaculum on 1/8/03 yielded identification of a banded male *C. townsendii.* Closest banding site to this location for a male of this species was at a mine approximately 3.5 miles away (5.5 km). Band number could not be read owing to the height of the bat within the mine.

Site Type	**Original Band Date (Age at Banding)**	**Recapture Date(s) (Age at Recap)**	**Time Length**
Night Roost/Hibernacula (Cave)	6/3/93 (A)	1/27/94 (A)	7 months
Night Roost/Hibernacula (Cave)	9/16/92 (J)	4/8/93 (A) 11/24/97 12/01/99	7 months 5 years 7 years
Night Roost/Hibernacula (Mine)	6/15/93 (A)	10/25/93 (A) 12/21/93 3/7/94 12/1/94	4 months 6 months 9 months 1 year 6 months
Night Roost/Hibernacula (Cave)	4/29/93(A)	12/29/93 (A) 2/23/94	8 months 10 months
Night Roost (Cave)	6/3/93 (A)	9/3/93 (A)	3 months
Night Roost/Hibernacula (Cave)	6/3/93 (A)	1/27/94 (A)	7 months
Night Roost (Cave)	7/31/92 (J)	5/20/94 (A)	1 year 10 months
Night Roost (Cave)	6/3/93 (A)	7/22/94 (A)	1 year 1 month
Night Roost (Cave)	5/10/94 (A)	5/26/95 (A)	1 year
Night Roost (Cave)	9/20/92 (J)	7/9/93 (A)	10 months
Night Roost/Hibernacula (Cave)	6/3/93 (A)	12/28/99 (A)	7 years
Night Roost/Hibernacula (Cave)	8/18/92 (J)	4/8/93 (A)	8 months

Corynorhinus townsendii**-** NIGHT ROOST / HIBERNACULA - SAME SITE BAND RECAPTURES

Site Type (Orig. Banding / Recap.)	**Original Band Date (Age at Banding)**	**Recapture Date[s] (Age at Recap)**	**Time Length/Distance Between Captures**
Cave / Cave	8/20/93 (A)	2/16/94 (A)	6 months / 4 km
Cave / Cave	8/27/92 (J)	2/16/94 (A)	1 year 6 mo. / 5.5 km
Cave / Mine	9/13/94 (A)	2/16/95 (A)	4 months / 17.5 km
Cave / Cave	8/20/93*	1/12/94	5 months / 1 km
Mine / Mine	6/15/93*	1/8/03	8.5 years / 5.5 km

Corynorhinus townsendii**-** DIFFERENT SITE BAND RECAPTURES

*Probable; closest banding site for the species to the recapture location (no banding was conducted at the recap site)

Given our present knowledge about the species known to inhabit the Black Hills, *C. townsendii* is likely to be at greatest risk of significant population declines. This assessment is based upon *C. townsendii's* comparatively narrow range of acceptable roost requirements and its susceptibility to disturbance (Humphrey and Kunz 1976). Increasing disturbance levels at natural roosting sites coupled with closure of abandoned mines will, in the future, limit roost availability. Identification and protection of important roosting sites are important considerations.

C. townsendii accepts bat gates at hibernacula and at night roosts. While no maternity or nursery roosts have been gated in the Black Hills, gated, abandoned mines in Colorado and Wyoming are used by this species (K. Navo and B. Luce personal communications). Protection of important sites is presently considered the best management practice for this species.

A comprehensive conservation strategy has been completed for this species including an exhaustive review of the published literature (Pierson et al.1999).

Eptesicus fuscus (Big Brown Bat)

E. fuscus is a medium to large species with an average forearm length of 45.72 mm and an average weight of 17.54 gm. Pelage color varies substantially within the Black Hills but most frequently is a medium to dark brown. The fur is long with ears and membranes ranging from dark brown to black in color. Head and snout are broader than in *Myotis* species from which it can also be distinguished by its greater size.

It is the most common bat found roosting in buildings and is one of the more successful species within the Black Hills. Roost records include buildings, trees, railway tunnels, mines, caves, and at least one metal electrical fuse box. *E. fuscus* is found in a variety of hibernacula with varying microclimates. The variety of conditions is likely a result of its larger size and increased capacity for stored fat reserves (Kurta and Baker 1990).

E. fuscus was found hibernating in caves and mines and is the only species in the Black Hills known to hibernate in buildings. In colder locations, it is frequently found in crevices though it was observed roosting on rock surfaces. High relative humidity within hibernacula is not a requirement. Hibernation areas tend to be at spacious sites that do not require acrobatic flight through small passageways. Most of those hibernating at Jewel Cave were located in areas close to the historic entrance. Bat gates constructed to prevent unauthorized access into roosts did not deter this species. Monitoring at the two sites presently gated has shown they continue to use both sites as night roosts and hibernacula.

The largest known hibernaculum is a cave [T6N R4 Sec 6] in the northern Black Hills that contained approximately 100 individuals (1993-94 and 1994-95). Accurate counts are difficult due to the height and clustering in crevices, which is common at this site. It is also the largest hibernaculum in the northern Hills for *Corynorhinus*.

E. fuscus moves out of hibernacula earlier than *Myotis* species in the Black Hills. It is common for many to leave by the middle of March. Whether such hibernating groups are females, males, or a mixture of both is not known. It is likely the large body size allows for a wider range of acceptable roosting conditions during the early spring. As such, roosts considered marginal for hibernation may become acceptable with the approach of warmer weather.

Martin and Hawks (1972) suggested the possibility of female migration out of the Black Hills during winter. Subsequent hibernacula surveys identified four banded females at three separate caves, all located in the northern Black Hills. Band identification was not possible with three of these bats, because of height or clustering. All three of these sites were summer banding sites for females. The fourth banded female identified during hibernation was found on 2/23/94 and had been banded as an adult at the same location on 9/30/93. Turner (1974) also cites a record of a hibernating female being collected from a mine adit on 3/4/46 near Hill City, SD.

In addition to those banding recaptures previously mentioned, four other recaptures were recorded. All were recaptured at the original banding site. The first recapture was an adult male banded originally at a natural cave night roost [T3N R6E Sec 32] on 7/31/92. It was recaptured while night roosting at the same location on 8/12/93. The second recapture also occurred at this location. An adult male was banded on 9/2/92 and recaptured within the hour exactly one year later on 9/2/93.

The remaining two recaptures both occurred at Red Bat Pond in the northern Black Hills. The first was a pregnant female originally banded on 5/27/94 weighing 16.5 gm. She was recaptured at the same site on 8/12/94 post-lactating, weighing 25.0 gm. Increase in weight likely represents recent foraging activity. Griffin (1958) cites a record of this species ingesting 4.0 gm of insect prey in 90 minutes of foraging. The second recapture at this location was an adult male originally banded at this site on 8/12/94 weighing 23.0 gm and recaptured on 6/23/95, weighing 16.5 gm.

Considered an opportunistic forager, this species is found routinely throughout a variety of habitat types and demonstrates flexibility in roost selection and foraging behavior (Brigham 1991, Kurta and Baker 1990). Powerful jaws allow it to feed on large hard-shelled beetles though other smaller prey is also taken (Kurta and Baker 1990). Black (1974), analyzing dietary habits of this species, considered it a beetle specialist.

Of 158 adults examined between 1992 and 1995, 33 individuals (21 percent), 15 males and 18 females, displayed noticeable wear to the canines ranging from a slight noticeable rounding of canine tips to near complete wear (Tigner unpublished data). While nutritional deficiencies may contribute to this condition, it is likely to be related to selection of hard-shelled insect prey.

Though a strong, fast flying bat, it lacks the acrobatic and evasive abilities of smaller *Myotis* species. Grounded individuals are often unable to resume flight without climbing to an elevated launching position. Emerging from maternity roosts, this species frequently dives 1 to 2 meters before achieving flight. This species will often bounce off mist nets erected over surface water, lose momentum, and land in the water. It is a capable swimmer and can be difficult to catch in deeper water.

Maternity and nursery roosts in buildings are frequently discovered by human occupants by vocalizations and movement during periods of warm weather. In addition, this bat's larger size, hence, easier visibility at emergence, helped to identify roosts. In the Black Hills, all known maternity roosts are located in buildings. However, maternity roosts are also known to occur in snags (Brigham 1991). Though not located, evidence suggests a maternity roost occurred near Red Bat Pond in the northern Black Hills during the summers of 1994 and 1995. Very early arrival of large numbers of pregnant and lactating females coupled with the absence of structures in the immediate vicinity suggest a tree roosting colony.

Large maternity roosts of several hundred are not uncommon in the Black Hills. The largest maternity roost, comprised of 300 adults, was around a chimney in the attic of a two-story brick building near Sturgis, SD. Similar buildings located near this site also contained large numbers. Fluctuations in numbers at specific locations suggest these roosts share individual members.

Shared locations were confirmed when a small roost of 30 females was captured, banded, and relocated to an adjacent building prior to installation of a sprinkler system. Banded females were then observed in three other roosts during subsequent surveys. A conservative estimate in the vicinity of these buildings was 1000 individuals.

Variation in parturition dates in the Black Hills was observed within and between seasons. One nursery roost, located at the South Dakota School of Mines and Technology at Rapid City in a building scheduled for demolition, contained 28 individuals (7/14/94). The group was comprised of 18 adult females, 14 lactating, 2 pregnant, 1 non-reproductive, 1 unrecorded reproductive condition, and 10 juveniles, 4 male and 6 female, all volant. Volancy begins between 18 and 35 days of age (Kurta and Baker 1990). The group was held in a single holding cage in a cool environment for two days, which induced torpor, banded and released at the original location following the building's demolition.

As this bat frequently selects buildings for roosting sites, it is likely to benefit from an appreciation of bats and their habitat requirements. Of particular importance is the development of a public education program including the proper methods of excluding roosts from buildings. Complaints of noise, large dropping deposits, and odor associated with urine accumulation are the most frequent reasons individuals seek help to exclude this bat.

Lasionycteris noctivagans (Silver-haired Bat)

The silver-haired bat is a medium sized bat with an average forearm length of 41.30 mm and an average weight of 12.31 gm. Pelage is dark, usually black, with silver-tipped hairs scattered throughout. Fur continues onto the uropatagium. Ears are rounded with a blunt, rounded tragus and ears and wing membranes are black in color. *L. noctivagans* is substantially smaller than *L. cinereus,* which also presents a frosted appearance.

L. noctivagans is considered a seasonal migrant arriving in spring and migrating south to warmer climates in the fall. In the Black Hills, the earliest record for a capture is on 5/5/94 when three adult females were netted at Roby Spring (Mattson 1994). Farther north, three individuals were observed foraging over Apex Pond on 5/11/94. Latest record in the season for this species was 9/16/94 for a pair foraging over beaver dam ponds on East Creek in Wyoming.

All captures of *L. noctivagans* occurred over water sources. Most were caught during July, though it was commonly netted as late as September in the southern Black Hills. This is substantially later than the region's other common seasonal migrant, *L. cinereus*, which is generally a rare capture by the end of August.

Two records of possible migrants were recorded at Ellsworth Air Force Base, 20 km east of the Black Hills. The first capture on 9/5/92 was a nulliparous adult female day roosting in the fork of a tree only 1 m from the ground. The second record (8/31/93) was a parous female captured while day roosting in a crevice on tree bark 1.5 m from the ground.

Evidence of the migratory habits was confirmed with the recapture of an adult male, in Denver, CO in October 1997, which was banded 7/29/94 at a woodland pond near Whitewood, SD (P. Murphy personal communication). The distance between the banding site and the recovery location is approximately 523 km. This distance is one of the longest documented for this species.

Turner (1974) cites a record of a *L. noctivagans* being collected from a cave on 11/19/67 near Rapid City, SD, apparently in hibernation, and surmises that some individuals may winter in the Black Hills.

Support for this was found during hibernacula surveys in a natural cave immediately adjacent to French Creek in Custer State Park. A survey conducted on 1/25/02 identified a single individual of this species in a vertical crevice. A follow-up survey was conducted on 3/6/02 and found a single specimen hibernating in a different nearby location, the first roost location being abandoned. The following winter, 2002-03, the site was again surveyed. Two individuals roosting separately were identified on 12/18/02. Based upon diminishing fall capture rates coupled with increasing spring captures, it is likely that the majority of this population relies upon migration. These recent records appear to document a segment of the population remaining in the Hills throughout the winter months.

L. noctivagans is a slow flier commonly seen drinking over woodland ponds early in the evening, often prior to sunset, during periods of warm weather. In the Black Hills, roost emergence for foraging usually occurs approximately 30 minutes after sunset. Considered an opportunistic feeder, the diet is comprised of a variety of insects across its North American range (Kunz 1982, Whitaker et al. 1981*b*).

While all species of bats in the region will roost in trees, *L. noctivagans* is one of the three species of the Black Hills that roost almost exclusively in trees. Virtually all information collected on roosting preferences comes from work conducted in the southern Black Hills by Mattson (1994) using radio telemetry.

Males were found to roost solitarily beneath loose bark or within cracks or crevices on the boles of trees. They were also observed changing roost trees frequently, usually daily, and roosting at varying heights on the bole. Twenty percent roosted less than 2 m from the ground.

Females give birth to one or two pups with twins being more common (Kunz 1982). Ten maternity and nursery roosts were identified, all in Ponderosa pine snag cavities. Average height of maternity roosts was approximately 10 meters. The number of bats roosting at these sites ranged from 6 to 55 individuals.

Three of the maternity and nursery roosts identified in 1994 were monitored intermittently during the summer of 1995 with no evidence of reuse (Tigner unpublished data). The low number of observations coupled with the frequent relocation of this species does not warrant any conclusions regarding seasonal roost fidelity. Mattson (1994) found maternity roost inhabitance averaged eight days.

Though never located, a maternity and nursery roost was presumed to exist in the vicinity of the Keystone, SD sewage lagoons. On 7/14/93, shortly after sunset, approximately 11 individuals were foraging and drinking over a large pond adjacent to the lagoons. A lactating female from this group was netted.

Earliest capture of a volant juvenile was recorded on 7/11/94 near Hazelrodt Picnic Ground near Custer, SD. Thought to be newly volant, juveniles were observed flying around a snag serving as a nursery roost on 7/8/94 (Mattson 1994, Mattson 1995). Volancy in this species begins at 21 to 28 days (van Zyll de Jong 1985). This species has demonstrated considerable variability in parturition as a pregnant female was also netted earlier on 7/7/94 at Hazelrodt Picnic Ground. Post-lactating females were netted 8/12/94 and 9/6/94.

One lactating female was netted while foraging over water at Ranch A in Wyoming on 7/25/95. Priday and Luce (1995) report capturing two individuals on 6/20/94 over a stream northwest of Sundance, WY.

Mattson (1995) observed owl predation on a probable juvenile taken while resting on the bole of the roost tree. The predator was thought to be an eastern screech owl (*Otus asio*) or a northern saw-whet owl (*Aegolius acadicus*).

A difference in distribution was noted for *L. noctivagans* following comparisons between summers of 1993 and 1994 (Tigner and Aney 1994). Weather conditions between the two years were distinctly different with lower temperatures and greater rainfall characterizing the summer of 1993. Survey work conducted in 1993 in the northern Hills resulted in capture of only two individuals with six additional records based upon sightings and echolocation monitoring via ultrasonic detector. Mattson (1994), surveying in the southern Hills during the summer of 1993, frequently found this species to be the most common bat netted with 108 total captures including adult females and juveniles.

Based upon routine captures and frequent sightings of this bat in the northern Black Hills during the summers of 1994 and 1995, it is possible that unfavorable weather conditions limited the northward range of this bat during the summer of 1993.

Until recently, little was known about the natural history of this species. Difficulties in locating roosting sites have largely been overcome by the application of improved radio telemetry equipment and techniques. While definitive conclusions regarding this species' habitat requirements would be premature, several common factors have been identified.

Maternity roosts for *L. noctivagans* are located in snags, frequently old woodpecker cavities (Mattson 1994, Vonhof 1996). The snags tend to be large, with dbh 38-62 cm (Betts 1996, Mattson 1994, Vonhof 1996), which likely provides a relatively stable roost microclimate. They also typically have an unobstructed southern exposure that probably elevates roost temperatures (Betts 1996, Mattson 1994, Vonhof 1996). Communally roosting Vespertilionids typically select such warmer maternity and nursery roosts, as increased temperatures serve to shorten gestation length and promote rapid development of juveniles (Racey 1982). The selection of roost sites by bats is likely the most important factor determining juvenile survival (Tuttle and Stevenson 1982). Another common finding has been the frequent relocation of the roost generally within a localized area (Betts 1996, Mattson 1994, Vonhof 1996).

L. noctivagans is likely to be susceptible to changes in forested habitats. Reductions in snag numbers result in fewer roosting sites for this species. As such, forest management practices (e.g. timber management and firewood collection) need to maintain the availability of larger snags over time and in numbers necessary for sustaining this species.

Summer monitoring is important to assess *L. noctivagans* trends. As a seasonal migrant, summer surveys represent the only means of monitoring this bat. While additional information needs to be collected on specific habitat requirements, the role of snag management will undoubtedly serve as the basis for success of this species in the region.

Lasiurus borealis (Eastern Red Bat)

L. borealis is a medium sized bat with an average forearm length of 39.7 mm and an average weight of 12.5 gm (van Zyll de Jong 1985). It is one of the more colorful bats with reddish-orange pelage and long, pointed wings. Ears are short and rounded. The furring on this bat is long and dense and extends down onto the uropatagium (van Zyll de Jong 1985). It is easily distinguished from all other species found within the region by its coloration.

The Eastern Red Bat is the least common bat known in the Black Hills. They are fast fliers and forage in straight lines or large circular patterns feeding primarily on large moths and beetles (Shump and Shump 1982). In general, this is a solitary roosting species, though small family groups of 4 to 5 bats are not uncommon during the summer months (Shump and Shump 1982). In contrast to its solitary lifestyle, there is evidence of group migration (Shump and Shump 1982). It is thought to be a seasonal migrant arriving during the spring or summer and departing before cold weather arrives.

Considered a tree bat, this species roosts in the foliage of deciduous and coniferous trees, but generally does not rely upon cavities for protection (Barbour and Davis 1969). Multiple births of two to four pups are common and characteristic (Kunz 1982). It has been suggested the increased litter size may be in response to increased risks of predation owing to its characteristic exposed roosting posture within the branches of trees (van Zyll de Jong 1985). Common predators include blue jays (*Cyanocitta cristata*) and various raptors (Barbour and Davis 1969).

L. borealis was captured in five Black Hills locations. There are no records from the Hills in WY (Bob Luce personal communication). Based upon an earlier capture of a volant juvenile (7/29/68) and later, a lactating female, on 8/20/68, Turner (1974) surmised there were at least two families of *L. borealis* in the vicinity of Moon Campground that year. An adult male was captured at this same site on 8/8/93 (Mattson 1994). The westernmost record was an individual netted at Wildcat Peak on 8/1/93 (Mattson 1994). The third location was in the northern Black Hills near Whitewood, SD over Red Bat Pond where a non-reproductive adult female was netted on 8/12/94. She displayed slight wear of the canines.

Three individuals were captured during the summer of 1998. The first location, Alkali Creek near Sturgis, SD yielded the capture of a juvenile female (8/19/98). Two other individuals were netted at an ephemeral woodland pond (T3N R6E Sec. 32) on 8/23/98. The first of these was an adult male captured simultaneously with another that escaped the net (Tigner 1998).

Based upon the limited numbers of observations and its migratory characteristics, it is difficult to determine population characteristics within the region.

Lasiurus cinereus (Hoary Bat)

L. cinereus is the region's largest bat with an average forearm length of 52.69 mm and average weight of 27.6 gm (van Zyll de Jong 1985). It is also one of the more colorful bats with fur mixture of blacks and browns with frosted white on the tips. Ears are short and rounded with black trimming around the edges. Greater overall size distinguishes it from all other Black Hills species.

In flight, it is distinguishable by its large size, rapid speed, and forceful echolocating call which, when not foraging, is generally characterized by a slow emission of pulses at low frequency (18000 kHz). This species commonly hisses and emits a spitting sound when netted or disturbed while displaying a menacing, open-mouthed defensive posture. Such posturing is no bluff, and gloves should be worn and extra care taken if handling this species.

L. cinereus is a strong bat capable of flight from a level surface including from the surface of water. It is a fast flier that commonly feeds at treetop level above the forest canopy. The diet is comprised mainly of large moths with other insects being taken to a lesser extent (Black 1974, van Zyll de Jong 1985).

All captures in the Black Hills were over water sites with nearly all occurring well after dark, though it was observed flying high prior to sunset. Most captures begin in early June with capture rates and observations decreasing by the end of August, suggesting an early migration. Similar observations, made in Canada, noted mid-August as the beginning of migration (van Zyll de Jong 1985).

The earliest record is an adult male captured (5/27/94) at Red Bat Pond near Whitewood, SD. The latest seasonal record of capture is 8/31 (Turner 1974), though one specimen killed by a dog was turned into the veterinary office at Ellsworth Air Force Base, 13 km east of the Black Hills, on 10/14/94.

Westernmost records are from Ranch A in Wyoming. Two adult males were netted over a creek on 7/25/95. Priday and Luce (1995) reported capture of an individual over a stream northwest of Sundance, WY in the Black Hills National Forest on 6/20/94. Two other records occurred at Stots Springs in western South Dakota. An adult female was captured on 8/13/93 and an adult male on 7/27/94.

Findley and Jones (1964) described summer sexual segregation in portions of North America. In contrast, Black Hills surveys observed both sexes in the region (Mattson 1994, Turner 1974, Tigner unpublished data). Pregnant females, lactating females, and juveniles have all been captured in the Black Hills. Earliest capture of a volant juvenile was 7/18/94 (Mattson 1994), which suggests parturition before mid-June (van Zyll de Jong 1985).

Although considered migratory, one banded adult female was recaptured at the original banding site, Red Bat Pond, approximately one year after banding. Netted originally on 7/29/94, she was parous but not lactating and exhibited no signs of pregnancy. At recapture on 6/23/95, she was obviously pregnant. Tuttle (1995) cites an example from Wisconsin of a female returning to the same tree for three consecutive years to raise young.

Not considered a colonial bat, netting surveys generally yield low numbers of captures during an evening. There were exceptions to this low capture rate in the southern Black Hills. Four such examples were noted.

On 7/7/93, 9 adult individuals, 4 male and 5 female, were netted over the sewage lagoons at Jewel Cave National Monument (Mattson 1994). The second occurrence was on 6/29/94 at Log Trough Ponds when 8 adults, 7 male and 1 female, were captured (Mattson 1994). The third survey resulting in large numbers of captures was on 7/18/94 also at Log Trough Ponds when 6 individuals were netted. This group was comprised of 3 adult males, 2 juvenile males, and 1 juvenile female (Mattson 1994). Other individuals were noted flying in the immediate vicinity but were not captured (Tigner personal observation). The final survey was conducted on 7/7/95 in the bottom of a steep-walled, narrow canyon over two small pools near Lower Woodcock Spring. During this survey, 7 individuals, 3 males and 4 females, were captured. This group was comprised entirely of adults including 3 lactating females (Paul Cryan personal communication).

MANAGEMENT RECOMMENDATIONS

Low reproductive rates, susceptibility to variations in seasonal weather conditions, narrow habitat requirements, and susceptibility to disturbance combine to highlight the need for conservation strategies for bats in the Black Hills.

Roost Protection

Whether year-round residents or seasonal migrants, all bat species found within the Black Hills are affected by the availability of suitable roosting sites. It is this aspect of their biology that is considered the most important limiting factor for distribution (Humphrey 1975).

Eight of the 11 species discussed rely on underground roosting sites at some point during the year. Dependence upon sites that provide specific microclimate conditions is probably the greatest limiting factor for species in the region. As such, it is likely the future management of natural caves and abandoned mines will play a decisive role in conservation of bat species in the Black Hills. Such management frequently requires restricting access to sites that are deemed significant during sensitive times of the year to minimize disturbance.

Site significance is often based upon a judgment of existing conditions within an area (Pierson et al.1999). Sites that contain large collectives of individual species or high species diversity are two examples of sites that should be considered significant. Another example of a significant roost would be a maternity or nursery roost for a species such as *C. townsendii,* which is known to be particularly susceptible to disturbance (Pierson et al.1999).

Bat Gates

Controlling access to significant underground roosting sites is most often achieved via installation of a specially designed bat gate. Such gates are designed to allow passage of bats in flight while restricting unauthorized human entry. In addition to these two criteria, gate design and installation must also take a variety of other circumstances into account.

Restricting existing air movement at access points of underground sites can change internal microclimates, causing abandonment (Tuttle 1977). Successful gate designs generally allow channeling of air rather than blocking air movement. Designs proven successful to a species should be given first consideration. Alternative designs should be monitored for efficacy. As this is a relatively new form of management, recent information should be obtained on current design recommendations before proceeding with gate construction.

Timing of gate installation should be adapted to minimize disturbance to the target population. If the site being protected serves as a hibernaculum, gate construction should be completed prior to the arrival of bats in the fall. Optimal gate location within the site should also be considered. Gating small access points can facilitate predation on entering or emerging bats. Placing gates in larger openings allows bats more maneuverability in navigating restrictive openings.

For a more complete discussion of gates and effective designs, see Tuttle and Taylor (1994) and Pierson et al. (1999).

Caves

Historically, caves probably met the hibernal roosting requirements for year-round resident bats. From the perspective of bats that rely upon underground roost sites, not all caves are equivalent. Numerous factors contribute to determining whether a site will be used and for what purpose (Ransome 1990). As has been commonly found in other areas, only a small percentage of total sites may provide adequate habitat for resident bat species (Tuttle and Stevenson 1978).

For a variety of reasons, many of these locations may no longer be viable roost sites. Commercial development of natural caves can be incompatible with roosting bats. Higher levels of disturbance associated with frequent visitation during the warmer seasons can prevent bats from using sites as day roosts. In attempting to protect the unique physical features found in commercially developed sites, physical barriers that restrict entry may be erected. Non-commercial wild natural caves often are damaged if unprotected.

Solid door closures restrict ingress and can alter the microclimate by preventing natural air exchange (Tuttle 1977).

Equivalent levels of protection may be achieved by installation of bat friendly designs that allow bats access and permit natural airflow. Sites developed for commercial purposes are frequently larger than many of the wild caves and, as a result, may contain a variety of microclimates if natural conditions were permitted. Such wide-ranging conditions may provide habitat to a variety of species (Tuttle and Stevenson 1978).

Cooperative partnerships between private cave owners and public agencies could minimize the costs associated with converting access points and developing appropriate management plans. Encouraging such ventures could provide a cost effective method by which new and secure bat hibernacula could be developed.

Disturbance in wild caves has reduced the suitability and number of available roosting sites. With increasing outdoor recreation demands, such disturbance is likely to increase in the future. Recreational caving can often cause unacceptable disturbance to bat roosts, which is often unintentional but no less consequential in impact. Significant disturbance can result from many factors. Body heat and non-electric light sources increase ambient temperature; noise generated by moving or talking, and close examination of hibernating bats may prove deleterious. Such disturbance may go unnoticed due to a delay in the response time required for a bat to arouse from hibernation.

Increasing interest in recreational caving underscores the important role formal caving organizations can play in educating that segment of the population that adopts this activity as a pastime. Such caving organizations are often comprised of conservation minded individuals with an interest in protecting cave resources. Enlisting their participation in conservation strategies should be considered a priority of habitat managers.

Some people who enter caves, through either ignorance or malevolence, do not subscribe to any code of underground conservation. Examples of disturbance documented in underground sites in the Black Hills are numerous.

Some of the more disruptive and damaging activities inside caves and abandoned mines include discharging firearms and fireworks, spray-painting, campfire construction, and intentionally killing roosting bats and other wildlife. Fire building is particularly common and likely results in the greatest level of long-term disturbance. In addition to elevating interior temperatures, which are detrimental during hibernation, and accumulating smoke, deposition of soot on ceilings can eventually result in site abandonment.

Even those sites that require greater skill to enter may need restricted access to prevent multiple disturbances during sensitive times of the year. Such forms of disturbance are not limited to a particular season. Winter disturbance was documented with increasing frequency in many locations monitored since 1992.

The Black Hills contains a myriad of roads that allow access to much of the forest. This results in increased casual disturbance of roosts. Road closures may serve to reduce disturbance levels at roost sites by reducing the ease of accessibility.

Caves that are closed should be accompanied by signs explaining the reason for closure as well as times when the site is accessible for visitation. Increasing public awareness is key to developing effective bat protection strategies. Posting informational or prohibitory signs at roost locations that do not control access via gates is not recommended. Vandalism of such signs in remote locations is common. In addition, they may serve to increase the curiosity factor, which can result in increased levels of casual disturbance. Informational signs of this nature could be incorporated into more protected areas such as campgrounds and visitor centers.

One exception to this scenario would be posting educational signs at locations that are more difficult to access and would therefore have a lower visitation rate. For example, one such roost, requiring technical climbing to access, serves as a nursery roost for *C. townsendii*. Gating this location would be extremely difficult. Placement of a permanently affixed sign inside the site's access which describes its significance, times of year considered most sensitive, and appropriate responses should bats be observed, could serve to limit disturbance and increase awareness.

To date, six locations in the Black Hills, three natural caves and three abandoned mines have been gated to protect roosting bat populations. Gate design followed recommendations that met with success in other parts of the country. Site survey work was conducted for at least one year to determine how the location was used and by which species. The three sites were used by all[1] species documented from the site after gating.

Post-gating surveys found activity levels and patterns of use to be unchanged. (See TABLE 5.) One cave, which allows public access during the summer months but was gated to control access during winter hibernation, was used by juveniles and adults as a night roost. Recaptures of banded bats entering the site or observed during hibernation indicate the site continues to serve the same purposes.

An initial increase in number of bats hibernating within one of the gated mine was observed following installation. Unfortunately, the site was vandalized shortly after this observation and construction of campfires within the location resulted in roost abandonment for the duration of that winter. At this writing, this location has remained secure for five years. Winter surveys have shown overall numbers of bats to return to previous levels with the addition of a new species, *Myotis septentrionalis.* This species was not observed at this site during pre-gating winter surveys.

Of particular concern in the Black Hills is the protection of maternity nursery roosts for *C. townsendii*, as this is the only Black Hills species thought to characteristically form such roosts underground. While no maternity or nursery roosts have been gated to date within the Black Hills, similar gate designs proved acceptable to maternity and nursery roosts of *C. townsendii* in abandoned mines in Colorado and Wyoming (K. Navo and B. Luce personal communication). Protection of such sites should be given a high priority.

It should also be noted that access to hibernation sites at Jewel Cave National Monument requires bats to fly through or over a gate placed at the historic entrance. This gate was replaced with a more bat friendly design during the fall of 2000. With the exception of *M. evotis*, all other hypogeal species of the Black Hills have been documented utilizing this site (Mattson and Bogan 1993, Worthington and Bogan 1993).

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[1] One exception to this was the capture of a single *M. evotis* at one of the locations prior to gate installation. Given the low capture rate for this species, coupled with its acrobatic flying ability, it is likely its absence from post-gating surveys (at this location) reflects a lower population density rather than a rejection of gating.

Abandoned Mines

The rate of abandoned mine closure has accelerated throughout the United States in recent years (Tuttle and Taylor 1994). Such closures have largely been in response to concerns of public safety and resulting liability (Tuttle and Taylor 1994). Public land management practices have historically not considered abandoned mines as wildlife habitat (Pierson and Brown 1992), but this viewpoint is changing as documentation of the importance of abandoned mines grows. As evidence of this recognition, the U.S. Bureau of Mines included bat gates in a recent publication describing mine closure guidelines (U.S. Dept. Interior 1994).

To date, closure of abandoned mines within the Black Hills has largely been restricted to private land or active claims on public land. Most of these closures have been conducted without any assessment of use as bat habitat. In addition to intentional closure of abandoned mines, many older sites are closing from natural degradation at entrances.

For North America north of Mexico, 29 species of hypogeal bats have been documented utilizing mines as roost sites (Kunz and Pierson 1994). In the Black Hills, seven bat species were found to roost in abandoned mines. The extent to which abandoned mines sustain bat populations in the Black Hills is unknown. Based upon observations in other regions, for those Black Hills' species that roost underground, closure of unsurveyed abandoned mines should be considered a substantial threat.

Abandoned mines are most commonly used as hibernacula and night roosts. It should also be noted that in other areas of the western United States, *C. townsendii* commonly use abandoned mines as maternity and nursery roosts (Pierson et al.1999).

In the Black Hills, most bat survey work has been limited to well known mines and safely accessible adits. Mines with vertical access entry or difficult interior passages have had minimal evaluation. Such sites are likely to have less human disturbance as well as natural predators. In other areas of the country, such characteristics have been associated with larger collectives of roosting bats (Tuttle and Taylor 1994).

The importance of abandoned mines as important bat habitat is well documented. One of the best examples occurred in 1992 when a mine in northern Michigan, slated for closure the following spring, was found to contain approximately one million hibernating bats (Tuttle and Taylor 1994). The site, Millie Hill Mine, Iron Mountain, MI, represents the second largest hibernaculum ever discovered in North America. This mine has since been protected by installation of a specially designed protective cage allowing bat access and preventing unauthorized human entry.

While the above scenario represents exceptional circumstances, the implication for the practice of unsurveyed mine closure is not diminished. Given our limited knowledge, it is entirely possible that a handful of sites within the Black Hills may provide shelter to significant populations. Loss of such sites, if they exist, would be detrimental to the bat populations of the region.

If appropriate microclimate conditions exist at such sites, it is possible they provide habitat to significant numbers of bats. For those sites that have (or are potential habitat), alternatives to complete closure have been developed and successfully employed to allow continued use by bats while restricting unauthorized human entry (Tuttle and Taylor 1994). In addition, such closures are often less costly to employ than traditional methods of closure (Tuttle and Taylor 1994). Sites requiring protection should be treated in the same manner as natural caves, including development of a comprehensive management plan. [See BAT GATES]

As with caves, not all mines meet roost habitat requirements. Temperature, humidity, physical structure, location, level of disturbance, and distance to other habitat requirements are important considerations when evaluating such locations. However, it is also important to recognize the potential variability between mines that may be in proximity to one another. As such, the significance of one location cannot be determined by survey work conducted at a separate, nearby site.

Some general mine characteristics are associated with bat roosting. Mines with multiple portals generally possess more complex airflow, which may provide for a wider range of internal temperatures and humidity (Tuttle and Taylor 1994). Such variability may provide roosting requirements to a wider variety of species.

Sites containing support timbers throughout are less likely to contain large numbers of bats. Many of the adits in the Black Hills have supporting structures located at the portals or for a short distance in less stable areas. These structural supports are frequently used by bushy-tailed wood rats (*Neotoma cinerea*) for nesting sites as well as ladders to higher levels within the mine. These mines rarely contain roosting bats. Avoiding these areas may be in response predation threats by these or other terrestrial predators. [See PREDATION]

The physical structure of the mine itself may play a role in site selection. In the Black Hills, mines with large interior chambers are likely to be used as roost sites. Such features provide bats a roosting location with minimal predation risk. Size of tailings or waste rock deposits cannot be used as a reliable determination of mine size. Such deposits, particularly at older sites, may have been altered by natural erosion or mechanical means.

Single portal mines comprised of long adits, greater than 25 m, are likely to be used by more bats than similar sites of shorter length. Such sites may contain a wider variety of roosting temperatures. Hibernating bats that roost on the rock surface were observed moving deeper into mines during periods of cold weather. Such sites may also provide more room for evading or hiding from potential predators if disturbed during summer day roosting.

Mines with portals blocked by vegetation are used less frequently than those with a clear open access, even if reduced. During road construction associated with timber sales, slash was deposited to block access to abandoned mines (Tigner personal observation). While likely done with the best of intentions, such practices may serve to restrict access to potentially important roosts and should be prohibited.

Sites that contain natural or man-made cracks and crevices are more likely to be used by a variety of species. These features are exploited by some species as a roosting thermoregulatory behavior (Ransome 1990), while others appear to roost exclusively during hibernation in such locations. For some species, crevice roosting may render interior microclimates acceptable when surface roosting would not.

In some cases, the instability of a mine or other hazards requires that the mine be closed. The following considerations can reduce the risk of trapping substantial numbers of bats. Temporary measures to prevent bat access should be employed in the time between interior survey and permanent closure. If the entire mine has been examined and found to contain no roosting bats, inexpensive plastic sheeting can be used to block portals and prevent bats from entering until permanent closure. If the entire interior of the mine cannot be surveyed completely, such closures should still be employed, but sheeting should be removed nightly to allow trapped bats to emerge. Closures should be employed during the late summer and early fall to permit adequate time for juveniles to achieve flying ability in case the site is used as a nursery roost.

Surveys of all mines should include evaluation by individuals trained to recognize bat signs. It is important to note that such surveys are often only specific to the season of survey. For example, sites used exclusively as hibernacula may give little indication of use if surveyed during July.

With the increasing pressures on natural caves, a management plan supplemented by protecting artificial roosting sites such as abandoned mines will enable larger numbers of sites to be protected with minimal impact upon recreational interests.

For detailed descriptions of evaluation methods of abandoned mines as bat habitat see Pierson et al. (1999) and Tuttle and Taylor (1994).

Land Management Surrounding Significant Bat Roosts

Altered roost microclimates can result from changes occurring in surrounding vegetation or landscape features. Such changes can be of particular significance within smaller sites with less complex airflow (Tuttle and Stevenson 1978). For example, increases in roost temperature can occur when overstory is removed permitting the roost's access point longer exposure to sunlight. Vegetation can also shelter access points. Changes in vegetation or landscape features that result in increased air movement across the access point can cause changes in roost microclimate.

One means of limiting changes to microclimate is to establish buffer zones around significant access points. While specific conditions may affect the size of such buffers, a minimum of 150 m horizontal radius to access points for roosts containing *C. townsendii* has been recommended (Pierson et al.1999).

This method was used on the Black Hills National Forest to protect the microclimate of a cave that functions as a large hibernaculum for *C. townsendii.* Once the cave's importance was explained to the timber contractor, he agreed not to harvest within the delineated buffer. The buffer is approximately two ha in area with the access point located roughly within the center. The cave lies within a narrow drainage with adjacent ridge tops serving as two buffer boundaries. Management intent is to retain the forest structure in the future. Heavy equipment is prohibited within the boundaries to further minimize disturbance.

Sites deemed significant should have formal site-specific management plans prepared to assure continuity of protection. Such plans should be comprehensive in scope including surrounding land management recommendations, forest dynamics, potential for natural disturbance such as fire, monitoring recommendations, and methodology.

Snag Management

Three Black Hills bat species, *L. noctivagans*, *L. borealis*, and *L. cinereus*, are considered tree-roosting bats. Trees are also known to be utilized as roosts by all other resident bats found in the Black Hills with the possible exception of *Corynorhinus*.

Though individuals may roost in protected areas of healthy trees, cavities within dead and dying trees are generally associated with collective roosting. Such colonies are required for the reproductive success of several of the bat species found in the Black Hills. The species for which the greatest documentation of this behavior has been collected is *L. noctivagans* (Mattson 1994).

Several factors have combined to reduce the number of snags. These include commercial timber harvest that began before the turn of the century in the Black Hills. Management of forests for commercial harvesting is designed to improve timber production, which results in the reduction of the number and quality of snags available for wildlife. Such reductions are owing to the practice of removing diseased trees and those posing physical hazards, as well as harvesting trees before maximum size is attained (resulting in an overall reduction in the mean size, age, and decadence of trees).

Firewood collection in forested close to urban areas has also resulted in a reduction of the number of available snags.

Bat Roosts in Buildings

Six bat species in the Black Hills are known to use buildings as maternity or nursery roosts. The extent to which other species have adopted such roosts in the Black Hills region remains unknown. In addition to collectives of bats, individuals will frequently roost on or within buildings.

As the human population increases, such roosting behavior is likely to continue and may increase. Collectives that commonly form in buildings are maternity or nursery roosts. Proper management techniques for building roosts may become an important component of future conservation strategies. Misconceptions about bats roosting in buildings are common. Public education will be an essential part of bat conservation.

Buildings offer a number of different attractive features as roost locations. The first of these is temperature. Warmer roost temperatures shorten gestation length and hasten physical development of juveniles. Attics typically have relatively warm temperatures and are frequently chosen for roosting.

An attic may provide a variety of temperatures. Such a range allows the bats to seek an area of optimal temperature without having to relocate. When roosts are located in buildings with pitched rooflines, the roosting location is frequently found at the apex during the cool evening temperatures. During the heat of the day, bats frequently move down the roofline away from the apex on the shaded side of the house. Roosts may also be located around chimneys and stone or masonry walls, which generally hold warmth longer than other materials.

A second feature of many maternity and nursery roosts identified within buildings in the Black Hills is the presence of a spacious and open attic. In addition to temperature variations, an open attic provides pups a secure area for learning to fly. Juveniles that fall to the ground can climb along unfinished wooden supports or rafters to rejoin the roost and the mother. Flying in large open areas within the confines of an attic minimizes predation risk.

Since ideal roosting sites are often accessible by only small access points, bats may exploit locations where they are less likely to confront large predators or roost competitors that may be more common in natural roosting locations.

The most effective means of ridding a building of unwanted bats is by removing the roost access (Barclay et al. 1980, Greenhall 1982). This generally requires blocking all openings. This strategy is more effective and less disruptive if completed prior to the arrival of the bats in the spring or following their departure in the fall.

Effectively evicting bats from a building requires a series of simple steps. Failure to follow the described procedures can result in bats becoming entrapped in the structure. Bats trapped inside roosts will seek alternative exits and may end up in the interior via openings around plumbing and electrical wiring. This is generally viewed as unsatisfactory for humans and bats.

The first step in excluding bats is to identify the main roost access point usually by observations at dusk or early evening. There may be several openings, but generally, one is used more than the others are. Following the identification of this access point, all other potential entrances should be closed or filled. Closure can be achieved using a variety of materials and must be conscientiously applied to the entire structure. Leaving unblocked access points often results in a new favorite being established.

Bats are not rodents and do not create or enlarge holes in buildings. They do not build nests, chew wiring or other similar activities characteristic of rodent infestation. In light of this, plugging openings with soft material such as insulation will suffice for closing small openings. Large openings for ventilation should be covered with screen. Smaller *Myotis* species have been observed entering and exiting through a single opening within the trough of pointing between two bricks and the overlapping wooden fascia board where roof meets wall. Such small openings can make restricting access to bats difficult in many older structures.

Once all cracks and openings have been sealed, the favorite access should be covered with a piece of plastic netting or sheeting that hangs over the opening and is attached only at the top. This creates a loosely hung flap over the opening. This flap must hang flat over the surface in which the hole occurs. The opening at the bottom allows bats inside to emerge but blocks their return.

Roost openings are frequently located in corners or areas that are difficult to cover with a flat lying piece of netting. If such is the case, a funnel shaped tube constructed of sheet plastic can be attached over the access point. The large end of the funnel is sealed around the roost access opening with the small end hanging down. Again, the bats are able to emerge but unable to reenter the tube and the roost.

These devices should be left in place for four to five evenings of good weather. During this period, all bats will have the opportunity to emerge. After this time, the temporary devices can be removed and the hole filled permanently. The exclusion process should only be undertaken in early spring before females have given birth or in late summer or early fall after the juveniles are volant. If done during the nursery season, non-volant pups can be separated from mothers. Searches for access points into the roost by the adult females are likely to be more thorough if pups are trapped inside.

If the roost is located within a large open attic in a building that is difficult or too costly to seal completely, measures can be taken to make the site a less desirable roost. When bats are not in residence, fall or winter are best, the attic can be divided into separate compartments. This can be achieved by stapling inexpensive sheet plastic to rafters forming a curtain within the attic. Curtains create a physical barrier to flight. The curtain should extend the full width of the attic and should come within 30 cm of the attic floor. Slitting the plastic curtain with vertical cuts will facilitate air movement and allow easier human passage within the attic.

In larger attics, these curtains can be hung up to 3 m apart if no access points to the outside occur within the compartments. Care should be taken to center such partitions between heads of fire sprinklers to prevent blocking the water spray in the event of a fire. Creating these smaller compartments restricts the flight area within the roost and may be enough of a nuisance to cause roost abandonment. If the bats continue to use the site, it usually limits their activity to one area and often prevents them finding their way into the building's interior by restricting movement within the attic.

Other Methods of Bat Exclusion

Many traditional methods for discouraging bats from roosting in buildings are ineffective. One of the more common remedies is scattering mothballs throughout the roosting area. Numerous roost surveys have found bats and mothballs frequently occupying the same area with no apparent effect on roost use.

Electronic devices that emit ultrasonic noise to drive away roosting bats are also ineffective (Bomford and O'Brian 1990, Hurley and Fenton 1980). One maternity roost of *Eptesicus fuscus* was observed immediately above one such device mounted to a supporting column within an attic in Hot Springs, SD (Tigner personal observation).

Poisons are the least effective method of control. They do not provide a long-term solution to the problem and can frequently be more problematic. In South Dakota, the use of poisons to kill bats is illegal. Poisoned bats often become grounded, which may bring them into contact with pets and children. Poisoned bats are often able to fly some distance from the roost thereby spreading the problem to surrounding areas (Greenhall 1982). All methods of bat roost control in Wyoming are subject to approval by the Wyoming Game and Fish Department (Bob Luce personal communication).

Public Education

Public education remains one of the keys to bat survival. As the greatest source of threats to bats originates with human activities, creating an increased public awareness of their ecological importance should provide a significant contribution to protecting these animals.

Since several species have been documented roosting in man-made structures, an increased public awareness of how best to manage undesired bat roosts is needed. Nuisance roosts are often maternity roosts which, if disturbed or disrupted, may result in the loss of a complete generation of bats. Owing to their low reproductive rate, such disruptions can have significant impacts on population size for many years. Bat topics can easily be incorporated into traditional public education media. Resources to supplement such public education undertakings are available through organizations such as Bat Conservation International in Austin, Texas [www.batcon.org].

Materials describing appropriate measures for dealing with unwanted bats are a particularly effective means of public education. Materials should be made available through offices of public agencies that are most likely to receive such requests. These include county extension offices, South Dakota Game, Fish and Parks offices, Wyoming Game and Fish Department, United States Forest Service.) [See Stukel et al. (1995) for examples.]

Public Health

One of the more frequent justifications cited for persecuting bats is based upon fear of threats to human health. The following information addresses some of the more commonly asked questions and popular misconceptions about bats and their impact on public health.

Rabies is probably the best known and most feared zoonotic disease of public health significance. While misconceptions about the relationship between bats and this viral disease are numerous, recent research has demonstrated this relationship to be far less menacing than is often portrayed. As with all mammal species, bats are susceptible to contracting this disease. While some bat species seem to be more susceptible to contracting rabies, the disease in bat populations is not thought to be increasing (Brass 1993). Occurrence of the disease in bats in North America has been estimated to be less than ½ of 1 percent (Constantine 1988).

The traditional view of bats as asymptomatic carriers of rabies, immune to its progression, is now known to be untrue (Brass 1994). Another common fallacy is that bats serve as reservoirs of the virus from which the disease is spread to other species. Research has not demonstrated bats to be an important link in transmission of the disease to terrestrial mammals (Brass 1993).

One of the surest methods of avoiding a possible exposure is simply not to handle wild mammals. Not handling a possibly infected bat will reduce your risk of contracting rabies to nearly zero. When handling dead or live animals is required, always wear thick gloves. Teaching children the importance of a hands-off approach to wildlife should be one of the primary goals of any education program. Natural inquisitiveness and innocence, coupled with anthropomorphizing common in children's media, can leave them particularly susceptible to contracting this disease. Indeed, worldwide statistics show children to be the group most frequently affected by this disease (Brass 1994).

Human exposure to rabies is far more likely to come via contact with an infected dog or cat than through contact with bats (Brass 1994). As such, vaccinating household pets against rabies is one of the most important preventive measures that can be taken against the spread of this disease (CDC 1996). Vaccinated pets that may have been exposed to an infected animal are generally prescribed a rabies booster, although a veterinarian should advise the appropriate course of action.

In the event of a possible exposure, one of the most effective means of preventing the disease is immediate and thorough washing of the infected site with soap and water. However, this is not to be considered an alternative to receiving prompt medical treatment (CDC 1991). All possible exposures should be promptly evaluated by qualified medical personnel (CDC 1991). Where possible, collection of the suspect animal for testing is recommended. Effectiveness of treatment of this disease is highest when begun soon after exposure (Berkow 1987).

The term exposure is not limited to a bite that results in rupture of the skin. Handling wildlife with cuts or scratches on hands, which become contaminated with saliva or other material of an infected animal, constitutes possible exposure. Such contamination can also occur through contact with a handler's mucous membranes (CDC 1991). These forms of potential exposure must be treated in the same manner as a bite (CDC 1991).

Incubation of the disease in a host can vary substantially from a few days to a year (Brass 1994). This finding is of particular importance to all mammalian wildlife handlers. The common belief that only healthy animals successfully hibernate through the winter is not valid, and such animals cannot be considered free of this disease (Brass 1994). In addition, handling specimens frozen for long periods may still pose a threat as freezing can preserve the virus that causes the disease (Constantine 1988).

The Center for Disease Control (CDC) recommends all individuals in high risk occupations, such as veterinarians, veterinary assistants, and wildlife handlers, receive a series of pre-exposure vaccinations to boost antibody levels in the event of an exposure (CDC 1991). The rabies pre-exposure vaccines do not provide immunity to contracting this disease. They are designed only to boost antibody titer and to improve the response to treatment that is still required following all possible exposures (CDC 1991).

For those individuals whose studies or work require handling bats and other mammalian wildlife, every effort should be taken to minimize potential exposures. These include pre-exposure vaccines followed by serology and vaccine boosters, promptly reporting and treating all potential exposures to the disease, and wearing protective gloves when handling animals. If gloves cannot be worn, animals should not be handled by people with cuts or abrasions on their hand.

For an exhaustive review of bats and rabies, see Brass (1994).

Other

Concerns about parasites are commonly expressed by many people. Nearly all parasites known to infest bats are species specific and pose little threat to pets or humans (Constantine 1988). No transmission of disease has been documented by parasites of bats to humans (Constantine 1988).

Histoplasmosis is primarily a disease of the respiratory system that manifests symptoms similar to tuberculosis (Berkow 1987). Infection comes from inhaling dust containing the fungal spores that cause the disease (Berkow 1987). The spores are commonly associated with pigeons, poultry, and bat droppings. Human exposure is generally associated with disturbing dry fecal deposits resulting in airborne dust containing the spores that is then inhaled. Individuals working in dry dusty conditions where fecal deposits may occur, particularly in confined spaces, should wear appropriate respirators. Spraying fecal deposits with water prior to disturbance will reduce dust and help minimize exposure (Benenson 1990).

FUTURE BAT RESEARCH IN THE BLACK HILLS

While much has been learned about bats in the Black Hills, many questions remain unanswered. Additional information is needed on the maternity roosting requirements of species for which there is minimal information. Further study also needs to be done on the role of abandoned mines in supporting bat populations and the identification and protection of important maternity roosts, nursery roosts, and hibernacula.

To facilitate the collection and dissemination of new information, agencies charged with the management of public lands and wildlife should develop formal relationships for the expressed purpose of maintaining monitoring and to continue habitat research. This networking would benefit those conducting research as well as the bats themselves. Alliances will provide benefits in two main areas. The first of these benefits is economic. Given the scarcity of resource agency funding, pooling resources will improve the quality and amount of work that can be conducted.

Agency participation in collaborative efforts may take a variety of forms and need not be limited to financial support. Providing vehicles, trained field personnel, researcher accommodations, and support equipment are some contributions that would be helpful.

The second benefit from collaborative ventures will be to provide a coordinated approach to future research. Cooperative ventures would prevent costly duplication of research, identify priority areas, and serve as a formal avenue of communication and dissemination of results. Consolidation of collected information into a commonly supported database would provide an effective and complete tool for future evaluation of species distribution and population trends.

Research duplication can also have a negative impact upon specific populations or roost sites. Hibernacula surveys can be a source of disturbance. Uncontrolled duplication, aside from wasting resources, can have detrimental effects on the site. Coordinated projects would reduce the possibility of excessive levels of disturbance at important roost sites. Permit applications for mist-netting, collection of specimens, and banding should include specific information as to where, how, and why the work is to be done. Such applications should be reviewed and evaluated by persons qualified to determine the applicant's ability and screened for possible duplication. This information would enable a uniform standard to be maintained in conducting future bat research.

The *Guidelines for the Protection of Bat Roosts*, as prepared and discussed by the American Society of Mammalogists (1992), could serve as the framework under which all future research would be conducted. In addition to these guidelines, the recent Idaho State Conservation Effort has additional measures specific to *C. townsendii* that should be incorporated into this framework of standards (Pierson et al.1999). Beginning in January 2001, SDGFP began requiring that applicants requesting scientific collector's permits to sample or collect bats in South Dakota provide detailed descriptions of research/monitoring plans prior to being approved for a state collector's permit [See Appendix II.]

To prevent unnecessary disturbance, specific locations of significant roosting locations, whether natural caves or abandoned mines, should be considered confidential and not for public distribution.

In Conclusion

Bat species of the Black Hills require a variety of habitat types. Changes within these habitat types will have an inevitable impact upon those species dependent upon them. Species such as *C. townsendii* or *L. noctivagans* with specific summer roosting requirements are particularly susceptible to these changes. The variation observed in hibernacula selection for year-round residents also demonstrates the diversity of required habitat. Accelerating changes resulting from increasing pressures on existing habitat likely represent the greatest threat to the survival of bat species in the region.

Given the rapid pace of change, the success of mitigation measures designed to maintain diversity could only be gauged through continued species monitoring. The importance of continued bat population monitoring in the Black Hills cannot be overemphasized. Though biological surveys are often judged to be an end in themselves, we agree with Bogan et al. (1989) in viewing them as the only reliable means to monitor population trends over time. As such, biological surveys should be viewed as an ongoing process for collecting information to assess effects of land management and help guide resource decisions.

Taken independently, much of the bat work conducted to date within the Black Hills can be considered little more than anecdotal. Lacking a historical perspective to provide points for comparison, individual point-in-time surveys only give a reflection of present conditions. Population trends are impossible to discern from such limited information. Collectively, this work, if combined with ongoing and future research, will provide a basis for understanding the complex interactions between bats and the unique habitat found within the Black Hills.

Fully understanding the complex relationship between bats and their environment must remain a goal for the future. Identifying habitat requirements is but the first step in long-term protection of these ecologically important species. Once identified, conserving these requirements becomes the challenge facing wildlife and land management agencies. Without the implementation of these identified management guidelines, future surveys will serve only to provide simple documentation of the extirpation of bats from the Black Hills.

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SPECIES BANDED	♂	♀	SPECIES TOTAL
M. ciliolabrum	55	21	76
M. evotis	05	01	06
M. lucifugus	101	74	175
M. septentrionalis	137	27	164
M. t. pahasapensis	54	21	75
M. volans	67	22	89
E. fuscus	72	90	162
C. townsendii	34	71	105
L. borealis	0	01	01
L. cinereus	18	05	23
L. noctivagans	43	18	61
SPECIES TOTALS	**586**	**351**	**937**

Table 1 Species, sex and total number of bats banded: 1992-1995.

SPECIES	♂	♀	TOTAL
M. ciliolabrum	2	1	3
M. lucifugus	2	0	2
M. septentrionalis	10	1	11
M. t. pahasapensis	5	0	5
M. volans	3	0	3
E. fuscus	3	11	14
C. townsendii	10	13	23
L. cinereus	0	1	1
L. noctivagans	1	0	1

Table 2 Banding recaptures by species: 9/92-12/02.

No records of recaptures have been noted for *M. evotis* or *L. borealis*.

Tables 3 and 4 provide common measurements collected from bats in the Black Hills (Tigner unpublished data).

SPECIES	**♂ FA [mm] (n)**	**♀ FA [mm] (n)**	**FA average for species (n)**	**FA range for species (mm)**
M. ciliolabrum	31.16 (76)	31.53 (32)	**31.27** (108)	29-34
M. evotis	37.80 (5)	40.00 (1)	**38.17** (6)	37-40
M. lucifugus	37.37 (108)	37.66 (73)	**37.49** (181)	32-41
M. septentrionalis	36.03 (141)	36.29 (28)	**36.07** (169)	32-43
M. t. pahasapensis	40.50 (54)	41.70 (20)	**40.82** (74)	37-44
M. volans	37.79 (72)	38.35 (23)	**37.93** (95)	35-41
E. fuscus	45.18 (73)	46.14 (96)	**45.72** (169)	41-51
L. borealis	**	42.00 (1)	**39.7**[1] (8)	36-42[2]
L. cinereus	52.35 (20)	53.83 (6)	**52.69** (26)	50-55
L. noctivagans	41.09 (45)	41.73 (22)	**41.3** (67)	39-44
C. townsendii	43.18 (34)	44.85 (72)	**44.31** (106)	40-47

Table 3 Forearm (FA) measurements by sex and species.

[1, 2] from van Zyll de Jong 1985

Weights are shown to provide a comparison between species only. In calculating the mean weights, no adjustments were made for variables such as pregnancy[1] or recent foraging activity[2].

SPECIES	♂ weight [gm] (n)	♀ weight [gm] (n)	mean weight for species [gm] (n)	weight range for species [gm]
M. ciliolabrum	5.72 (62)	5.71 (28)	**5.72** (90)	4.0-9.5
M. evotis	7.50 (2)	7.50 (1)	**7.50** (3)	7.5
M. lucifugus	8.23 (96)	8.64 (29)	**8.33** (125)	5.0-11.5
M. septentrionalis	7.14 (113)	7.09 (22)	**7.13** (135)	4.5-11.0
M. t. pahasapensis	7.70 (45)	8.11 (14)	**7.80** (59)	6.0-10.5
M. volans	7.73 (56)	8.18 (17)	**7.84** (73)	5.0-11.5
E. fuscus	17.18 (68)	17.88 (72)	**17.54** (140)	11.0-26.5
L. borealis[3]	**	20.00 (1)	**12.5**[4] (4)	10.0-17.44[4]
L. cinereus	23.58 (20)	20.50 (2)	**27.6**[4] (22)	25.4-30.3[4]
C. townsendii	11.04 (27)	12.21 (24)	**11.59** (51)	9.0-19.0

Table 4 Weight by sex and species.

[1] Kurta and Kunz (1987) in surveying published literature, found the mean weight of bat pups at birth to be 22.3 percent of the mother's post-partum body weight.

[2] One adult female *C. townsendii* banded emerging from a cave was recaptured 2.5 hours later reentering the cave. At emergence, she weighed 11.5 gm and at recapture (with bulging stomach) 16 gm, a gain of 4.5 gm or nearly 40% of its body weight (Tigner unpublished data).

[3] No males were netted; only one female was recorded.

[4] from van Zyll de Jong 1985.

SPECIES	NIGHT ROOST	DAY ROOST	HIBERNACULUM

	BEFORE➔AFTER	BEFORE➔AFTER	BEFORE➔AFTER
M. ciliolabrum	♂♀ ➔ ♂♀	♂✓ ➔ ♂✓	✓ ➔ ✓
*M. evotis**	♂ ➔ X		
M. lucifugus	♂♀ ➔♂♀	♂♀ ➔ ♂♀	
M. septentrionalis	♂ ➔ ♂	♂ ➔ ♂	
M. t. pahasapensis	♂♀ ➔ ♂♀	✓ ➔ ✓	✓ ➔ ✓
M. volans	♂♀ ➔ ♂♀		
C. townsendii	♂♀ ➔ ♂♀	♂♀ ➔ ♂♀	♂♀ ➔ ♂♀
E. fuscus	♂♀ ➔ ♂♀	♂✓ ➔ ♂✓	✓ ➔ ✓

* Represents a single capture at only one of the 3 gated locations; ✓=unknown sex; X=no captures

Table 5 Roost usage at gated sites.

APPENDIX I BATS OF THE BLACK HILLS - BRIEF DESCRIPTIONS

The following is a brief description of the eleven bat species known from the Black Hills. Included are behavioral and physical observations.

Distinguishing among *Myotis* species can be frustrating at times and nearly impossible if the bat is hibernating since hibernating bats should not be disturbed. Pelage color alone is not a reliable identifying characteristic as significant variations occur within species.

Myotis ciliolabrum (Western Small-footed Myotis) - Forearm is 29-34 mm. *M. ciliolabrum* is the smallest bat found in the Black Hills. Its skull gives a flattened appearance, and bat has characteristic black mask and ears that often contrast with pale brown fur. Calcar is keeled. *M. ciliolabrum* is a year-round resident that hibernates in caves and mines in crevices and rock surfaces and generally prefers cooler hibernacula. No maternity roosts have been identified in the Black Hills.

Myotis evotis (Long-eared Bat) - Forearm is 36-41 mm. Ears very long, 22-25mm in length, extending 5 mm or more beyond nose tip when pressed forward. Ear length is more than ½ length of forearm. Fur is long with color varying in shades of brown. Ears and membranes are very dark to black, and there is often an inconspicuous fringe of minute hairs along edge of uropatagium.

Myotis lucifugus (Little Brown Myotis) - Forearm is 34-41 mm. Blunt tragus is approximately ½ length of ear. Fur varies in shades of brown often with a sheen. Membranes and ears dark brown. Calcar is absent or weak. *M. lucifugus* commonly feeds very low over water surface. Maternity roosts are common in buildings during summer. It is a year-round resident that hibernates in caves and mines with high levels of humidity and is found in crevices and roosting on rock surfaces.

Myotis septentrionalis (Northern Myotis) - Forearm is 32-39 mm. Size is similar to *M. lucifugus* but has longer ears, 17-19 mm, with long, pointed tragus. Calcar is not keeled, and fur is dull, not glossy. Membranes and ears medium brown. Face is often more bald than other *Myotis* species. A very vocal bat when disturbed. It is a year-round resident that hibernates in caves and mines and is found in crevices.

Myotis thysanodes pahasapensis (Fringed Myotis) - Forearm is 39-44 mm. Ears are long, 16-20 mm, and generally very dark to black as is the mask. A distinct fringe of small stiff hairs is found along the edge of uropatagium. Subspecies found only in the Black Hills region. *M. t. pahasapensis* is a year-round resident that hibernates in caves and mines and roosts in crevices and on rock surfaces.

Myotis volans (Long-legged Myotis) - Forearm is 35-41 mm. It often displays a characteristic furring from elbow to knee on underside of wing membrane and a well-developed keel to the calcar. *M. volans* generally has darker brown membranes. A shorter rostrum gives head a smaller appearance than *M. lucifugus*. This species is a year-round resident that hibernates in caves and mines and is found on rock surfaces and in crevices. Maternity and nursery roosts have been found in a building and a snag.

Corynorhinus townsendii (Townsend's Big-eared Bat) Forearm is 41-47 mm. Ears are very long, >25 mm, with two lumps near the end of the snout. This species is often misidentified as the ears are commonly folded back and adjacent to folded wings while roosting or hibernating. Tragus does not fold back with ear and are often mistaken for *Myotis* ear tips. It is dependent upon caves and mines. *C. townsendii* roosts in the open on rock surfaces often near the openings of caves and mines. Echolocation is very soft and can be difficult to detect in the field. A very acrobatic flier, this species is known to utilize feeding perches during foraging. *C. townsendii* is a year-round resident with the largest known hibernating population in the Black Hills located at Jewel Cave.

Eptesicus fuscus (Big Brown Bat) - Forearm is 41-51 mm. *E. fuscus* is generally larger than *Myotis* species with a broad head and snout. Ears are short with rounded tragus. Calcar is keeled. It commonly roosts in buildings during the summer months. *E. fuscus* is a year-round resident that hibernates in caves and mines roosting in crevices and on rock surfaces.

Lasionycteris noctivagans (Silver-haired Bat) Forearm is 37-44 mm. Fur is dark, usually black, and silver-tipped. Ears are short and rounded. Rarely found underground, this bat usually roosts on tree trunks in crevices or beneath loose bark. Maternity roosts are located in tree cavities. This species is a slow flier and feeds over woodland ponds and streams. Although a few winter records exist, *L. noctivagans* is considered a migratory species in SD.

Lasiurus borealis (Eastern Red Bat) - Forearm is 36-42 mm. Fur is reddish, and wings are long and pointed. Ears are short and rounded. Interfemoral membrane is heavily furred. *L. borealis* routinely gives birth to more than one pup. Twins, triplets and quadruplets are not uncommon. This species roosts almost exclusively in the branches of trees and is generally a high-flying bat. There are few records of *L. borealis* in the Black Hills. It is considered migratory.

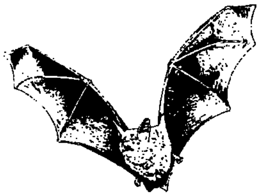
Lasiurus cinereus (Hoary Bat) - Forearm is 48-58 mm. *L. cinereus* is the largest bat in the Black Hills. Fur is multicolored and heavy. Ears are short and rounded edged in black. Many hairs tipped in white give the appearance of being frosted. It is not a colonial bat and is generally a high, fast flier. *L. cinereus* is a summer resident that migrates southward by the end of August.

APPENDIX II

BAT SAMPLING AND COLLECTION PROTOCOL GUIDELINES AND REQUIREMENTS

(Effective 1 January 2001)

SOUTH DAKOTA DEPARTMENT OF GAME, FISH AND PARKS
523 E. Capitol, Pierre, SD, 57501



The Black Hills of South Dakota and Wyoming have historically been of interest to bat researchers and specimen collectors. Many specimens have been collected for a variety of purposes, and these specimens are housed in museum and university collections throughout the country. In recent years, natural and intentional mine closures combined with the continued interest in research on Black Hills bat populations have caused concern about impacts of sampling and collection on local bat populations. This document presents specific guidelines and requirements for bat sampling and collection related to monitoring and research activities throughout South Dakota. This document is not intended to provide recommendations on study or sampling design, which can be obtained from other sources (ex: *Inventory Methods for Bats – Standards for components of British Columbia's Biodiversity No. 20.* 1998. Resources Inventory Committee, British Columbia).

The overriding intent of this document is to provide for the safety of individual bats and for the long-term stability of bat populations in South Dakota. State Scientific Collector's Permits will be issued only for research and monitoring activities that are compatible with this overriding intent.

Components of this document are arranged as guidelines or requirements. The permittee should assume that deviations from requirements will not be approved unless justified to the satisfaction of the SD Department of Game, Fish and Parks.

This document is a supplement to the *South Dakota Department of Game, Fish and Parks Free Scientific Collector's Permit Application.* Legal authorities: SDCL 41-6-32 (scientific collector's license) and SDCL 41-1-2 (state authority for wildlife protection).

REQUIREMENTS:

1. Applicants must provide evidence that they are familiar with the desired sampling techniques and with identification of bat species likely to be found in the study area. This requirement applies to all parties that will operate under the Scientific Collector's Permit. The Scientific Collector's Permit cannot be transferred or delegated to individuals not covered under the Permit.
2. In addition to completing the *South Dakota Department of Game, Fish and Parks Free Scientific Collector's Permit Application*, applicants must submit a copy of their study or monitoring plan. At a minimum, the plan should include species of concern, number of individuals to be collected or handled, study area and duration, techniques, personnel, and eventual location of specimens or tissues.
3. Mist nets must be attended at all times. Harp traps must be checked frequently to minimize effects of predation, weather, absence of mother from pup, etc.
4. No collection or handling of bats will be allowed at hibernacula. Surveys or other activities at hibernacula will be reviewed on a case-by-case basis.
5. Live bats may not be held overnight without permission of SDGFP.
6. No surveys will be allowed for western big-eared bats (*Corynorhinus townsendii*) at known maternity/nursery roosts. If a researcher suspects that western big-eared bats are using a site as a maternity/nursery roost, surveys must end immediately, and SDGFP must be notified. In general, surveys of maternity/nursery roosts of all bat species are discouraged.
7. Applicants desiring to handle bats must submit proof of pre-exposure vaccinations for rabies.
8. Applicants desiring to band bats must comply with established South Dakota protocol for band type, placement, color, and numbering system.
9. In general, no collection of specimens or body tissue will be allowed of bat species monitored by the SD Natural Heritage Program. A current list of monitored species can be viewed at: http://www.state.sd.us/gfp/Diversity/index.htm. If individuals of monitored species are captured, they will be released immediately at the point of capture. Any unintentional collection of individuals of monitored species will be reported to SDGFP within 72 hours and surrendered to SDGFP.
10. Collection of any bat species must be approved by SDGFP. Collection of more than two specimens of nonmonitored species must be justified within the context of a study or monitoring plan. This plan should include a description of the method of killing and a description of availability of tissue or specimens for examination by other researchers.
11. Bat survey or monitoring activities should not alter or damage natural or artificial sites.
12. Only personal, battery-powered or non-heat-generating light sources will be used within roosts. No open-flame torches, smoke-producing instruments, photography equipment, carbide lamps, or toxicants will be taken into roost sites.
13. Permit holders will notify SDGFP of the location of any maternity roosts discovered during study or monitoring activities.

GUIDELINES:

1. Applicants should be familiar with humane methods of removing other nocturnal animals likely to be captured in bat sampling equipment.
2. Applicants should be familiar with property boundaries in their selected study area and should seek permission from the appropriate public or private landowner.
3. Applicant should notify public or private landowner when survey activities will be conducted.

GLOSSARY OF SELECTED TERMS

adit - horizontal underground mine passage that connects to the outside

calcar (keeled calcar) - a piece of cartilage attached to the heel of the foot that extends along the free-edge of the tail membrane; "keeled calcar" refers to a small portion (flap) of the membrane that extends beyond the edge of the calcar

cave - naturally formed underground cavity

day roost - general term for a roosting location used by bats during the active season (may include maternity and nursery roosts)

drift - horizontal underground mine passage that follows a vein or layer of rock

echolocation - a highly advanced "sonar" system used by bats to orient themselves and locate food

hibernaculum - location used for overwintering; usually characterized by cooler (above freezing) stable temperatures

hibernation - energy conserving behavior used during longer cold weather periods when food is scarce; characterized by lower metabolic rate and body temperature

hypogeal - occurring beneath the surface

insectivorous - feeding on insects (all SD and WY bats are insectivorous)

maternity roost - roost location used by a group of reproductive adult females

mine - man-made underground cavity

night roost - location used between bouts of foraging for resting, ingesting larger prey, and socializing

nulliparous – never given birth

nursery roost - roost location used by adult reproductive adult females and their offspring

parous - having given birth

pelage - fur

portal - access point or opening at a horizontal mine entrance

shaft - vertical opening/passage into a mine

thermoregulation - controlling body temperature to achieve optimal use of stored energy reserves

torpor - a short-term adaptive behavior for conserving energy characterized by lowered body temperature and metabolic rate

tragus - small, erect piece of cartilage attached at the base of and located in front of the ear

Harp trap (a.k.a. Tuttle trap) - portable capture device used primarily at roost access points or within restricted flyways; comprised of a frame of two vertical banks of filament (each strung in harp-like fashion) divided by a narrow space with a collection bag hung below this gap. Bats generally fold their wings to pass through the first bank of strings and become blocked by the second bank whereupon they drop down into the holding bag. The remaining area of the cave/mine opening not covered by the trap is blocked with plastic sheeting. (See Tuttle 1974)

ultrasonic echolocation detector - in its simplest form, an electronic bat survey instrument that receives the echolocation pulses, which are generally above the range of human hearing, that are emitted by bats and lowers them into the range audible to humans.

uropatagium - membrane of skin that extends between the legs (a.k.a. tail membrane)

volant - able to fly

Red
Knot

NATIONAL ARCHIVES AND RECORDS ADMINISTRATION
1985

FEDERAL REGISTER

Vol. 78 Monday,
No. 189 September 30, 2013

Part II

Department of the Interior

Fish & Wildlife Service

50 CFR Part 17
Endangered and Threatened Wildlife and Plants; Proposed Threatened Status for the Rufa Red Knot (*Calidris canutus rufa*); Proposed Rule

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
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.

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Updated Status Review of Sicklefin and Sturgeon Chub in the United States

United States Department of the Interior
U.S. FISH AND WILDLIFE SERVICE
Region 6
Denver, Colorado

March 2001

UPDATED STATUS REVIEW OF SICKLEFIN AND STURGEON CHUB IN THE UNITED STATES

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I. SUMMARY

On August 8, 1994, the U.S. Fish and Wildlife Service (Service) received a formal petition to list the sicklefin chub (*Macrhybopsis meeki*) and the sturgeon chub (*Macrhybopsis gelida*) as endangered throughout their range pursuant to the Endangered Species Act (ESA) of 1973. The sicklefin and sturgeon chub are members of the Cyprinidae or minnow family and are endemic to the Missouri River basin and the Mississippi River below St. Louis in the central United States. Both of these species are highly adapted to living in free-flowing rivers with high levels of turbidity. The construction and operation of dams and reservoirs on the main stem Missouri River and channelization of the Middle and Lower Missouri River are the principal factors impacting sicklefin and sturgeon chub habitat by altering flow regimes, turbidity levels, and water temperature.

On January 19, 1995, the Service published a positive 90-day finding in the Federal Register that the petition and data available from other sources provided substantial information indicating that the petitioned action may be warranted. The Service requested comments on the 90-day finding; however, limited input was received. The Service also established a Status Assessment Team to gather information documenting sicklefin chub and sturgeon chub populations and determine whether listing these species as threatened or endangered under the ESA was warranted. A draft 12-month finding was completed in August 1995 and subsequently revised in 1997, 1999, and 2000 to include substantial new information. The Montana Rivers Coalition filed a 60-day notice of intent to sue the Secretary of the Department of the Interior on April 6, 2000, for the Service's failure to act on the petition in the time frames established by the ESA. The Montana Rivers Coalition's action resulted in a stipulated settlement agreement in which the Service agreed to submit the 12-month finding for the sicklefin and sturgeon chub for publication in the Federal Register on or before April 12, 2001.

The Service has received information concerning the status of sicklefin and sturgeon chub populations from State game and fish departments, the U.S. Bureau of Reclamation (Reclamation), U.S. Geological Survey, tribal representatives, universities, and other organizations and individuals.

The Service also reviewed information on the sicklefin and sturgeon chub from journal articles, agency reports and file documents, telephone interviews, and written correspondence with fisheries biologists familiar with these species.

The Service found that historic collection data documenting sicklefin and sturgeon chub populations are limited and provide an incomplete picture of their range and population levels. Both species have received little attention from fishery biologists until recent years.

The sicklefin chub was historically found in the Yellowstone River, Missouri River, and Mississippi River downstream from the confluence with the Missouri River. Construction of the six Missouri River main stem dams by the U.S. Army Corps of Engineers (Corps) between 1937 and 1964 eliminated sicklefin chub populations in the 800 miles of river converted to reservoir habitat and in approximately 200 miles of free-flowing reaches located below Garrison, Oahe, Big Bend, and Fort Randall Dams. The Service estimates that the sicklefin chub currently occupies approximately 54 percent of its historic range in the Missouri River basin. Field studies indicate that self-sustaining populations of sicklefin chubs occur in three reaches of the Missouri River--above the headwaters of Fort Peck Reservoir in Montana, in the Yellowstone - Missouri River confluence area of Montana and North Dakota, and in the Missouri River from St. Joseph, Missouri, to the confluence with the Mississippi River. Data collected by the Missouri Department of Conservation since 1997 indicate that a viable population of sicklefin chub are present in the Middle Mississippi River and in the Wolf Island area (river mile 930.7 to 935.0) of the Lower Mississippi River.

Sturgeon chub have been collected at or near the same locations where sicklefin chub populations have been documented in the Yellowstone, Missouri, and Mississippi Rivers. Sturgeon chub also have been historically collected in 30 of the larger tributaries to the Yellowstone and Missouri Rivers. Construction and operation of the six Missouri River main stem dams by the Corps has effectively isolated sturgeon chub populations. The sturgeon chub, like the sicklefin chub, has been extirpated from approximately 800 miles of the Missouri River that has been converted to reservoir habitat and from the 200 miles of free-flowing reaches below Garrison Dam in North Dakota and Oahe, Big Bend, and Fort Randall Dams in South Dakota. Operation of the Missouri River main stem dams continues to impact the chubs. The Service estimates that sturgeon chub currently occupy about 1,155 miles or about 55 percent of its former range in the Missouri River. Data available from fishery investigations conducted since 1994 indicate that viable populations of sturgeon chub are present in three reaches of the Missouri River--above the headwaters of Fort Peck Reservoir in Montana, upstream from Lake Sakakawea in North Dakota and Montana, and in the Missouri River in Missouri. A self-sustaining population of sturgeon chub also is present in the Lower Yellowstone River, the Middle Mississippi River, and in the Wolf Island area of the Lower Mississippi River.

Sturgeon chub populations are currently present in 11 of the 30 tributaries to the Yellowstone and Missouri Rivers where they were historically collected. Factors that have affected sturgeon chubs in the tributaries include construction and operation of dams and reservoirs, water withdrawals primarily for irrigation, and potential water quality impacts associated with energy production and intensive agriculture.

II. LIFE HISTORY AND ECOLOGY OF THE SICKLEFIN CHUB AND THE STURGEON CHUB

The sicklefin chub and sturgeon chub belong to the same genus of fishes (*Macrhybopsis*) in the minnow family (Cyprinidae). In general, they have similar distribution, habitat requirements, and are subject to similar threats. Therefore, they are addressed together in this updated status report.

Taxonomy

The sicklefin chub was first collected from the Missouri River near St. Joseph, Missouri, by Jordan and Meek (1885), but was initially misidentified as a sturgeon chub. Type specimens originated from later collections of Jordan and Evermann (1896) made at the same general area of the Missouri River near St. Joseph, Missouri, and were identified as *Hybopsis meeki*. The sicklefin chub was subsequently placed in the genus *Macrhybopsis* (Mayden 1989).

The sturgeon chub was first collected from the Milk River, a tributary to the Missouri River in Montana, during the Pacific Railroad Surveys of 1853-1855 by Dr. George Suckley (Girard 1856). Jordan and Gilbert (1882) first described the species as *Ceratichthys gelidus*. However, the generic name has been revised several times from *Ceratichthys* to *Hybopsis* (Jordan and Evermann 1896), to *Macrhybopsis* (Cockerell and Allison 1909, Jordan 1920), back to *Hybopsis* (Bailey 1951), and finally back to *Macrhybopsis* (Mayden 1989).

Morphology

The sicklefin chub is usually yellowish or tan colored on the back and silvery-white on the belly. The snout protrudes slightly beyond the mouth. The eyes are reduced and can be partially covered with skin. External taste buds are abundant on the underside of the head, lower body, and pectoral fins. The sicklefin chub also has a single pair of maxillary barbels located at the corner of the mouth. Sicklefin chub have extremely long pectoral fins and a deeply forked caudal fin with a darker lower lobe. The dorsal fin is sickle-shaped. Average adult length ranges from 35 to 100 millimeters (mm) (1.4 to 4.0 inches (in)) and average adult weight ranges from 0.5 to 6.0 grams (g) (0.02 to 0.2 ounce (oz)) (Cross 1967, Eddy and Hodson 1982). The sicklefin chub has a life-span of up to 4 years.

The sturgeon chub is tan to pale green on the back and cream to white on the belly. A few black speckles occasionally are present on the sides and back. It has a long, fleshy snout and subterminal mouth, in which a single pair of maxillary barbels are located at the corners. It has reduced eyes, a streamlined body, a deeply forked caudal fin, and epidermal keels on most scales. Taste buds are abundant on the underside of the head and on the belly and fins. These morphological features are adaptations to life in swift, turbid stream environments (Cross 1967, Pflieger 1975). Average adult length ranges from 35 to 95 mm (1.5 to 3.8 in) and average adult weight ranges from 0.3 to 7.3 g (0.01 to 0.3 oz) (Branson 1963, Branson 1966, Cross 1967, Reno 1969, Eddy and Underhill 1978, Robison and Buchanan 1988, and Werdon 1992). The sturgeon chub is a relatively short-lived species with a life-span of up to 4 years.

Both species are similar in morphology, but they possess distinct morphological characteristics. A unique characteristic of the sturgeon chub is its longitudinally-arranged epidermal keels, which improve hydrodynamic efficiency (Cross 1967). The unique characteristics of the sicklefin chub are the elongated pectoral fins and a sickle-shaped dorsal fin.

Habitat

General habitat requirements for the sturgeon chub and sicklefin chub have been reported by Cross (1967), Pflieger (1975), and the Service (U.S. Fish and Wildlife Service 1993a, 1993b). The sicklefin chub and sturgeon chub evolved in large, free-flowing riverine systems, characterized by swift flows, highly variable flow regimes, braided channels, high turbidity, and sand/fine gravel substrates. Both species generally use similar macrohabitat types and have similar biological requirements.

Studies conducted in the 1990s in Montana (Grisak 1996), North Dakota (Everett 1999), and Missouri (Grady and Milligan 1998) have described habitat characteristics at sicklefin chub collection sites. Sicklefin chubs were collected at depths from 0.1 to 11.0 meters (m) (0.3 to 36 feet (ft), bottom velocities from 0.14 to 1.06 m (0.5 to 3.5 ft) per second, and over a variety of substrate types. While sicklefin chubs have been collected from almost every type of Missouri River habitat type at one time, most fish have been collected in main channel, border channel, and sandbar macrohabitats over sand and fine gravel substrate.

Dieterman (2000) statistically examined 67 variables believed to influence sicklefin chub distribution in the Missouri River, including physical habitat, water quality, flow regime, and predation. This study presented the first quantitative evaluation between sicklefin chub distribution patterns and large-scale Missouri River features. Dieterman found four variables were significant following correction for multiple tests--distance to upstream impoundment, flow constancy, mean segment turbidity, and percent of annual flow in August. Occurrence of sicklefin chub was highest when a segment of the Missouri River was greater than 187 miles (301 km) downstream from a dam; flow constancy was 0.56 or less, indicating an association with river segments having more variable flow regimes; mean summer-early fall turbidity levels were 80 NTUs or greater; and the percent of flow in August was low, less than 10 percent of the total annual flow.

Dieterman (2000) also evaluated habitat use by age-0 and age-1+ sicklefin chub. Three site scale variables differed significantly between juvenile and adult sicklefin chub following correction for multiple tests. Sites where age-1+ sicklefin chub were present were characterized by faster water column velocities, a higher percentage of gravel, and a lower percentage of silt than sites where age-0 fish were collected. Optimum habitat conditions for adult sicklefin chub were analyzed using two methods. The Peeters and Gardeniers (1998) method indicated that optimum conditions during the summer-early fall ranged from 2.20 to 4.80 m (7.2 to 15.7 ft) for depth and 0.6 to 1.10 m (2.0 to 3.6 ft) per second for velocity. Wider optimum habitat conditions were predicted using the Jongman et al. (1987) method, with optimum depth ranging from 1.72 to 5.21 m (5.6 to17.1 ft) and velocity from 0.54 to 1.16 m (1.7 to 3.8 ft) per second.

The majority of sicklefin chub collected by Grisak (1996), Everett (1999), and Hrabik and Herzog (in litt. 2000a) were found over sand substrate. Dieterman (2000) statistically found a significant positive association of age-1+ sicklefin chub with gravel substrates. Grady and

Milligan (1998) collected the greatest percentage of sicklefin chub over organic matter (46.7 percent) and silt (23 percent). Most of these fish were likely age-0 fish. Sturgeon chub also are usually found in main channel and channel border habitats in areas with gravel and/or sand substrate with greatest abundance with fine to medium gravel.

Sturgeon chub and sicklefin chub are often captured together in the Yellowstone, Missouri, and Mississippi Rivers. Welker (2000) collected sturgeon chub in a wide range of depths and current velocities in the Yellowstone/Missouri River confluence area in North Dakota. The highest percentage of sturgeon chub were captured in depths from 2 to 5 m (88 percent) (6.6 to 16.4 ft) and in current velocities from 0.5 to 1.0 m (1.6 to 3.3 ft) per second (81 percent). Most studies describing sturgeon chub collection sites (Reigh and Elsen 1979, Gould 1994, Gelwicks 1996, and Hrabik and Herzog in litt. 2000a) indicate that the primary substrate used by sturgeon chubs is gravel. Welker (2000) reported that sturgeon chub in the Yellowstone/Missouri River confluence area primarily used sand substrate; however, he noted an increasing percent of gravel at sites positively influenced sturgeon chub densities.

In contrast to sicklefin chub, which only occur in large river systems, sturgeon chub also inhabit tributaries to the Yellowstone and Missouri Rivers. Trenka (2000) sampled sturgeon chub in the Montana reach of the Powder River in 1997 and 1998. Nearly all of the 61 sturgeon chub he collected (98.3 percent) were found in bar, inside bend, and secondary channel habitats. Sturgeon chub were primarily collected in shallow water areas, with moderately swift currents. Two-thirds (66.7 percent) of the sturgeon chub taken in seine hauls were from depths between 0.2 to 0.39 m (0.7 to 1.3 ft), and 75 percent were in areas with a current velocity between 0.4 to 0.79 m (1.3 to 2.6 ft) per second. Reigh and Elsen (1979) collected sturgeon chub at 41 sites in the Little Missouri River and 3 sites in the Yellowstone River in North Dakota in 1976 and 1977. Approximately, 93 percent of the sturgeon chub collected were in areas with rock and gravel substrate with moderate current.

Age and Growth

Grisak (1996) summarized age/length relationships for sicklefin chub collected in the Missouri River above Fort Peck Reservoir, Montana, in 1994 and 1995 (Table 1). The oldest fish he collected was age 4. Sicklefin chub collected during this study weighed between 0.6 and 9.6 g (0.02 and 0.34 oz). The heaviest male and female weighed 6.0 and 9.6 g (0.21 and 0.34 oz), respectively. Most specimens weighed between 1 and 6 g (0.03 and 0.34 oz).

Table 1. Length Range by Age Class of Sicklefin Chub Collected in the Missouri River Above Fort Peck Reservoir, Montana, 1994-1995.

Age	Length Range (mm)	Percent of Sampled Population
1	29 - 42 (1.1 - 1.6 in)	29
2	43 - 75 (1.7 - 2.9 in)	33
3	73 - 93 (2.9 - 3.7 in)	34
4	95 - 109 (3.7 - 4.3 in)	4

Everett (1999) evaluated age and growth relation for sicklefin and sturgeon chub collected in the Lower Yellowstone and Missouri Rivers in North Dakota (Tables 2 and 3). The oldest sicklefin and sturgeon chub collected were age 4 and 3, respectively. Stewart (1981) collected one sturgeon chub specimen that was age 4+ from the Powder River in Wyoming.

Table 2. Sicklefin Chub Age and Growth Relationships for Fish Collected in the Lower Yellowstone and Missouri Rivers in North Dakota, 1995.

Age	Length Range (mm)	Weight Range (g)	Percent of Sampled Population
1	39 - 53 (1.5 - 2.1 in)	0.7 - 1.1 (0.02 - 0.04 oz)	5
2	53 - 85 (2.1 - 3.3 in)	0.8 - 4.2 (0.03 - 0.15 oz)	70
3	86 - 99 (3.4 - 3.9 in)	3.3 - 7.7 (0.12 - 0.27 oz)	22
4	107 (4.2 in)	10.1 (0.36 oz)	2

Table 3. Sturgeon Chub Age and Growth Relationships for Fish Collected in the Lower Yellowstone and Missouri Rivers in North Dakota, 1995.

Age	Length Range (mm)	Weight Range (g)	Percent of Sampled Population
1	35 - 50 (1.4 - 2.0 in)	0.6 - 0.9 (0.02 - 0.03 oz)	6
2	51 - 75 (2.0 - 2.8 in)	0.7 - 2.1 (0.02 - 0.07 oz)	68
3	73 - 86 (2.9 - 3.4 in)	1.9 - 6.7 (0.07 - 0.24 oz)	26

Reproduction

The reproductive biology of sicklefin and sturgeon chub is largely unknown. Spawning is believed to occur in the spring as Pflieger (1975) collected young-of-the-year in July from the Missouri River in the State of Missouri. Werdon (1993 a,b) speculated that spawning is likely influenced by water temperature and also may be affected by increasing flows due to snowmelt or precipitation events. Larval *Macrhybopsis* chubs, including either or both sicklefin and sturgeon chubs, were collected in 1996 (Tibbs and Galat 1997). Water temperatures during spawning were estimated based on larval fish development to range from 20.5 to 26.2° C (68.9 to 79.2° F), with peak spawning temperatures ranging from 20.5 to 25.3° C (68.9 to 77.5° F) (Dieterman 2000). Sturgeon chub females in the Powder River in Wyoming became ripe in early June, with the principal spawning activity occurring later in the month and into July (Stewart 1981). Stewart collected no gravid females after July 26 and reported that scales taken from gravid females suggest sexual maturity at age 2+.

While additional research is needed to document the reproductive biology of sicklefin and sturgeon chub, sampling since 1994 indicates that there are stable, self-sustaining populations in widely scattered areas throughout their range. Chub populations are successfully reproducing under a variety of climatic conditions and subsequent flow regimes within the Missouri River Basin and Mississippi River.

Feeding Habits

Almost no information documenting the feeding habits of sicklefin and sturgeon chub has been published. Reigh and Elsen (1979) reported that three sicklefin chubs collected near the mouth of the Yellowstone River in North Dakota contained one black fly pupa (*Simulium* sp.) and pieces of what appeared to be insect exoskeletons, among other unrecognizable material. They also reported that sturgeon chub collected from the Little Missouri River in North Dakota contained insect body parts, but no other identifiable material. Stewart (1981) examined the stomach content of eight sturgeon chub collected from the Powder River in Wyoming. He found pieces of aquatic insects that could not be identified further.

III. HISTORICAL RANGE AND CURRENT DISTRIBUTION, GENERAL

Historical range of fish species is generally based on presence or absence of reliable catch records in peer-reviewed, published literature. Specific citations for field studies documenting sicklefin chub and sturgeon chub populations and experts' opinion on distribution and range are provided in subsequent sections of this report addressing specific river reaches.

In general, the historic record for the sicklefin chub and the sturgeon chub documents presence or absence and the total number of individuals collected. This record provides an incomplete picture of the range of these fish and their populations prior the constructions of dams and other water development activities that have impacted chub habitat. Historically, studies designed to document fish populations primarily focused on sport fish, with limited attention given to native cyprinid populations in the Missouri and Mississippi River basins. No long-term research has been conducted to estimate the size of sicklefin and sturgeon chub populations or determine how chub populations have changed over time.

In 1993, the historical capture data for the sicklefin and sturgeon chub were documented in two status reports prepared by the Service (U.S. Fish and Wildlife Service 1993a, 1993b). This work, combined with the 1994 petition to list the sicklefin and sturgeon chub as endangered, has resulted in an additional emphasis being placed on sampling native cyprinid populations.

Since 1994, when the Service was petitioned to list the sicklefin chub and the sturgeon chub as endangered, a number of field studies have been conducted to sample chub populations. Data available from recent field investigations provide a more complete record of the locations where sicklefin and sturgeon chub occur. Fisheries biologists also have improved the techniques for collecting chubs. Historically, seines of various lengths were used to sample cyprinid populations in shallow water habitat. Since 1994, researchers have found that benthic trawling is a more effective method of collecting sicklefin and sturgeon chub, particularly in water depths over 1 meter. Recent studies using benthic trawls indicate that sicklefin chub and sturgeon chub are more abundant and more widely distributed than indicated in the 1993 status reports, including areas in Montana, North Dakota, South Dakota, Nebraska, and Missouri.

Historically, the sicklefin chub was collected in the Yellowstone River in Montana and North Dakota, the Missouri River from Montana to its confluence with the Mississippi River near St. Louis, and the Mississippi River downstream from the mouth of the Missouri River. Sicklefin chubs also have been collected rarely in the Lower Kansas River in the year following high flows in the Missouri River. Based on reliable catch records, the sicklefin chub historically occurred in the waters bordering or within the following 13 States--Montana, North Dakota, South Dakota, Nebraska, Iowa, Kansas, Missouri, Illinois, Kentucky, Tennessee, Arkansas, Louisiana, and Mississippi. Data from the existing catch records and information available from fisheries biologists have been used to estimate the historic and current distribution of sicklefin chub. Please see Maps 1 and 2 and Tables 4 and 5.

Sicklefin chub habitat was substantially altered by the construction and continuing operation of six multipurpose dams and reservoirs on the Missouri River and channelization of the Lower Missouri River by the Corps. The Missouri River dam and reservoirs were completed between 1937 and 1964 as part of the Pick-Sloan Plan, a multi-purpose flood control and water development project implemented by the Corps and Reclamation. Today, on the main stem Missouri River, approximately 36 percent of the habitat within the range of sicklefin chub has been transformed into reservoir habitat, another 40 percent downstream of the dams has been channelized, and 24 percent of the river habitat has been altered by flow modifications, hypolimnetic releases, and reduced turbidity levels.

Click to View Map 1 - Sicklefin Chub Historical Range (Missouri River Basin)

Click to View Map 2 - Sicklefin Chub Current Range (Missouri River Basin)

Impacts to sicklefin chub populations from reservoir operations and channelization varies. Recent studies conducted in Montana, North Dakota, and Missouri using benthic trawls indicate that sicklefin chub comprise a significant portion of the fish population above Fort Peck Reservoir in Montana, in the Yellowstone/Missouri River confluence area of North Dakota and Montana, and in the channelized reach of the Missouri River in Missouri. At the opposite end of the spectrum, sicklefin chub populations have been extirpated from approximately 800 miles of riverine reaches that have been converted to reservoir habitat and approximately 200 miles of free-flowing reaches below Garrison Dam in North Dakota and Oahe, Big Bend, and Fort Randall Dams in South Dakota. The Service estimates that sicklefin chub currently occupy 54 percent of their historic range in the Missouri River basin.

Sicklefin chub populations also are present in the Middle and Lower Mississippi River. Field work conducted since 1997 by the Missouri Department of Conservation's Open River Field Station (this research center also is known as the Long Term Research Monitoring Station) has provided new information documenting both sicklefin and sturgeon chub populations in this portion of their range. Sicklefin chub habitat in the Middle and Lower Mississippi River has been altered by the construction of dike fields, bendway weirs, and other structures designed to maintain the navigation channel. However, due to the limited number of studies documenting sicklefin chub populations in the Mississippi River, the importance of this population and the full extent of impacts are unknown.

Sturgeon chub have been collected at or near the same locations where sicklefin chub populations have been documented in the Yellowstone, Missouri, and Mississippi Rivers. Sturgeon chub also have been collected historically in a number of the larger tributaries to the Yellowstone and Missouri Rivers. Based on reliable catch records, sturgeon chub have been collected in waters bordering or within the same 13-State range as the sicklefin chub, plus the Powder River drainage in Wyoming and Montana. Table 6 lists the tributaries to the Yellowstone and Missouri Rivers where sturgeon chub populations occur and the tributaries where sturgeon chub populations are believed to be extirpated. Information documenting the extent of the sturgeon chub's historic range in most tributaries is not available. Maps 3 and 4 present a pictural estimate of the historic and current distribution of sturgeon chub.

Construction and operation of the six Missouri River main stem dams and channelization of the Lower Missouri River by the Corps have substantially altered sturgeon chub habitat. Like sicklefin chub, impacts to sturgeon chub from reservoir operation and channelization varies. Fisheries studies conducted since 1994 using benthic trawls indicate sturgeon chub comprise a significant portion of the fish population in the Missouri River above Ft. Peck Reservoir, in the Yellowstone/Missouri River confluence area of North Dakota and Montana, and in Missouri. The sturgeon chub has been extirpated from approximately 800 miles of the Missouri River that has been converted to reservoir habitat and in the 200 miles of free-flowing reaches below Garrison Dam in North Dakota and Oahe, Big Bend, and Fort Randall Dams in South Dakota. Based on the best available survey data, the Service estimates that sturgeon chub currently occupy about 1,155 miles or 55 percent of its historical range in the Missouri River. Viable, self-sustaining populations of sturgeon chub also are found in the Lower Yellowstone River.

Table 4. Estimated Historic Distribution of Sicklefin Chub Populations in the Missouri and Mississippi River Basins.

		RIVER MILES
MISSOURI RIVER BASIN		
MISSOURI RIVER		
Mouth of Cow Creek, Montana, to the Confluence with the Mississippi River		1,950
YELLOWSTONE RIVER		
Mouth of Thirteen Mile Creek to the Confluence with the Missouri River		85
	TOTAL	2,035
MISSISSIPPI RIVER BASIN		
MIDDLE MISSISSIPPI RIVER		
Mouth of the Missouri River to the Confluence with the Ohio River		195
LOWER MISSISSIPPI RIVER		
Mouth of the Ohio River to the Gulf of Mexico		955
	TOTAL	1,150

Table 5. Estimated Current Distribution of Sicklefin Chub Populations in the Missouri and Mississippi River Basins.

	RIVER MILES
<u>MISSOURI RIVER BASIN</u>	
MONTANA	
Missouri River	
Cow Creek to Headwaters of Fort Peck Reservoir	**61**
Fort Peck Dam to North Dakota Border	**110**
Yellowstone River	
Thirteen Mile Creek to North Dakota Border	**68**
NORTH DAKOTA	
Missouri River	
Montana Border to Headwaters of Lake Sakakawea	**34**
Yellowstone River	
Montana Border to the Missouri River	**17**
SOUTH DAKOTA - MISSOURI	
Missouri River	
Gavins Point Dam to St. Joseph Missouri[1]	**370**
MISSOURI	
Missouri River	
St. Joseph, Misosuri, to the Mississippi River	**440**
TOTAL	**1,100**
<u>MISSISSIPPI RIVER BASIN</u>	
MISSOURI - ILLINOIS	
Middle Mississippi River	
Mouth of the Missouri River to the Confluence with the Ohio River	**195**
MISSOURI - LOUISIANA	
Lower Mississippi River	
Mouth of the Ohio River to the Gulf of Mexico[2]	**955**
TOTAL	**1,150**

The Missouri Department of Conservation's Open River Field Station has provided new

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[1] Sicklefin chub occur in low numbers.

[2] Limited information available documenting the sicklefin chub populations.

information documenting both sicklefin and sturgeon chub populations in the Middle and Lower Mississippi River. Sturgeon chub habitat in the Middle and Lower Mississippi River has been altered by the construction of dike fields, bendway weirs, and other structures designed to maintain the navigation channel. However, due to the lack of data documenting sicklefin chub populations in the Mississippi River the importance of this population and the full extent of impacts are unknown.

Sturgeon chub populations are likely extirpated from 19 of 30 tributaries to the Yellowstone and Missouri Rivers.

IV. HISTORICAL RANGE AND CURRENT DISTRIBUTION

UPPER MISSOURI RIVER BASIN

The Upper Missouri River basin includes the main stem Missouri River system and tributaries within the basin from the headwaters in Montana downstream to the Gavins Point Dam (river mile 811) in southeastern South Dakota. Six main stem Corps dams, reservoirs, and inter-reservoir reaches with regulated flows are located along the Upper Missouri River. This portion of the basin includes the tributary rivers and streams in Wyoming, Montana, North Dakota, and South Dakota.

Historically, the sicklefin chub in the Upper Missouri River basin was found in the main stem Missouri River from South Dakota to Montana and in the Lower Yellowstone River in Montana and North Dakota (U.S. Fish and Wildlife Service 1993a). The range of sturgeon chub overlapped the range of the sicklefin chub. In addition, sturgeon chub distribution included 30 turbid tributaries in the Missouri River basin, including the Yellowstone River and several of its tributaries in Wyoming and Montana (U.S. Fish and Wildlife Service 1993b, Montana Department of Fish, Wildlife, and Parks in litt. 2000).

Wyoming: In Wyoming, the sturgeon chub was collected historically in the North Platte, Big Horn, and Powder River drainages. The sturgeon chub is extirpated from the North Platte and is thought to be gone from the Big Horn River, due to the construction of large impoundments and alterations to flow regimes and physical habitat (Mike Welker, Wyoming Game and Fish Department, pers. comm. 2000, and Bill Wilchers, Wyoming Game and Fish Department, in litt. 1997). Sturgeon chub were last collected from the Big Horn River in 1981. Several surveys have been conducted since 1981, including an extensive effort by the Wyoming Game and Fish Department in 2000; however, no sturgeon chub were captured (Mike Welker, pers. comm. 2000).

Click to View Map 3 - Sturgeon Chub Historical Range (Missouri River Basin)

Click to View Map 4 - Sturgeon Chub Current Range (Missouri River Basin)

Table 6. **Historic and Current Distribution of Sturgeon Chub in Tributaries to the Yellowstone and Missouri Rivers.**

YELLOWSTONE RIVER TRIBUTARIES	EXTANT	EXTIRPATED
WYOMING	1. Powder River	1. Big Horn River
	2. Crazy Woman Creek	2. North Platte River
MONTANA	3. Tongue River	3. Sunday Creek
	Powder River	Big Horn River
	4. Sears Creek	
	5. Box Elder Creek	
MISSOURI RIVER TRIBUTARIES		
MONTANA	6. Redwater River	4. Milk River
		5. Teton River
NORTH DAKOTA		6. Little Missouri River
		7. Box Elder Creek
		8. Beaver Creek
		9. Green River
		10. Heart River
SOUTH DAKOTA	7. White River	11. Grand River
	8. Little White River	Little Missouri River
	9. Bear in the Lodge Creek	
	10. Cheyenne River	
NEBRASKA	11. Platte River	12. Niobrara River
		13. Republican River
		14. Loup River
		15. Elkhorn River
		16. Bazile Creek
KANSAS		17. Smoky Hill River
		18. Kansas River
		19. Wakarusa River
		Republican River

A survey of warmwater fishes in the Missouri River drainage in Wyoming (83 streams and 181 locations) was conducted during 1993-1995 (Patton 1997, Patton et al. 1998). Sturgeon chub were found at four locations on the Powder River and at one location on the lower reach of Crazy Woman Creek, a tributary of the Powder River. This survey extended the known range of the sturgeon chub up the Powder River by 50 mi. By comparing 1960s and 1990s survey data and adjusting the data for gear bias between surveys, Patton (1997) concluded the sturgeon chub population in Wyoming is stable, but limited in distribution.

Montana: Considerable data recently have become available on the current distribution of the sturgeon chub and sicklefin chub. The Benthic Fish Study for the Missouri River and Lower Yellowstone River, the largest fish study ever conducted on the Missouri River system, began in 1995. Standard sampling techniques and gears (e.g., gill nets, benthic trawls, bag seines, electroshocking, and trammel nets) were used during three field seasons in 1996-1998 and targeted bottom-dwelling benthic fish, including the sturgeon chub and sicklefin chub. The study, chiefly sponsored by the Corps, but supported by several other Federal agencies, was undertaken by a consortium of river scientists from six U.S. Geological Survey, Biological Resources Division, Cooperative Fish Research Units (Montana, Idaho, South Dakota, Iowa, Kansas, Missouri), and the Montana Department of Fish, Wildlife, and Parks.

Despite six low-head diversion dams for irrigation systems, the lower Yellowstone River appears to support a healthy population of sturgeon chub. Researchers with the Benthic Fish Study collected 230, 285, and 712 sturgeon chub, and 6, 34, and 53 sicklefin chub from the Yellowstone River in 1996, 1997, and 1998, respectively (Dieterman et al. 1997; Young et al. 1998; Mike Ruggles, Montana Department of Fish, Wildlife and Parks, in litt. 1999). These fish were collected from the 71-mile reach of Yellowstone River between the Intake Diversion Dam in Montana downstream to the confluence with the Missouri River in North Dakota.

During 1996-1998, Reclamation conducted investigations of the impacts of irrigation withdrawals and fish entrainment at the Intake Diversion Dam on the Lower Yellowstone River (Hiebert et al. 1999). The diversion is controlled through 11 inscreen sluice conduits. Estimates of fish entrainment were obtained by collecting all fish from 2 to 4 of the 11 conduits with fyke nets. During limited sampling using 2 conduits in 1996, a total of 2,931 fish were collected from the water diversion canal, including 378 sturgeon chub (12.9 percent). No sicklefin chub were collected that year. In 1997, Reclamation intensified the effort to 24 hours per day for 8 days and collected a total of 7,980 fish, including 1,008 sturgeon chub (12.7 percent), and 2 sicklefin chub. This extended the known range for the sicklefin chub upstream in the Yellowstone River. Entrainment netting in 1998 collected a total of 4,529 fish, including 744 sturgeon chub (16.5 percent), and 0 sicklefin chub. Estimates of total entrainment were determined by extrapolating the monthly average entrainment rates over the irrigation season. Hiebert (et al. 1999) projected that approximately 289,000 ± 113,000 sturgeon chub were entrained in the irrigation canal system.

Both the sturgeon chub and sicklefin chub are known to occur in the Missouri River above Fort Peck Reservoir (Grisak 1996). Grisak's field work highlights that different fishery collection methods can yield substantially different results. During the 1994-1995 field seasons, benthic trawls and bag seines were used in July and August to sample a 100-mile stretch of the Missouri River upstream from the headwaters of Fort Peck Reservoir (Grisak 1996). During 141 seine hauls at 23 random sites 5,095 fish were collected, including 4 sicklefin chub (0.08 percent of the catch) and 8 sturgeon chub (0.16 percent of the catch). Sturgeon chub and sicklefin chub ranked 14th and 15th in abundance, respectively. In contrast to the seine collection data, sicklefin and sturgeon chub were the second and third most common species collected in benthic trawl tows. During 1994 and 1995, Grisak collected 1,376 fish with a benthic trawl, including 302 sicklefin chub (21.9 percent of the catch) and 260 sturgeon chub (18.9 percent of the catch). The benthic trawl permitted collections at sites with deeper water. The mean depth at trawl sites where sicklefin chubs were collected was 3.41 m (11.2 ft) (1.37 to 6.41 m - 4.5 to 21.0 ft), as compared to an average depth of 0.50 m (1.6 ft) (0.19 to 0.86 m - 0.6 to 2.8 ft) at sites sampled with seines. Grisak did not collect sicklefin chub in the upper two segments of his study area (Judith River confluence - river mile 1982 to Grand Island - river mile 1930) where water depth generally does not exceed 2.4 m (7.9 ft). All sicklefin chub were collected in the lower three segments of the study area (Grand Island to the headwaters of Fort Peck Reservoir - river mile 1883). Maximum water depth ranged from 5.5 to 11.6 m (18 to 38 ft) in this area.

Data from the Benthic Fish Study indicate that 43, 161, and 377 sturgeon chub, and 21, 109, and 137 sicklefin chub were collected from a 70-mile reach of river immediately above Fort Peck Reservoir in 1996, 1997, and 1998, respectively (Dieterman et al. 1997; Young et al. 1998; Lee Berstedt, Montana Cooperative Fishery Research Unit, in litt. 1999). In general, sampling efforts increased in successive years and most chubs were caught with benthic trawls.

In recent years, Tews (1994), Liebelt (1996), Dieterman et al. (1997), Young et al. (1998), and Ruggles (in litt. 1999) reported distributional data for the sicklefin chub and sturgeon chub on the Missouri River segment between Ft. Peck Dam (river mile 1771) and the Yellowstone River (river mile 1582). Their collective data indicate that both species appear to be absent from the 11-mile segment of river from Fort Peck Dam downstream to the Milk River (river mile 1760). Recent fish surveys in the Milk River tributary by the Montana Department of Fish, Wildlife, and Parks in 1997 and 1998 did not document the presence of the sturgeon chub (Mike Ruggles, in litt. 1999).

Sturgeon chub are found in increasing numbers from the Milk River downstream to the Yellowstone River. During the 3-year period from 1996-1998, 5, 9, and 14 sturgeon chub were collected in the 59-mile Missouri River segment from the Milk River to Wolf Point (river mile 1701). Collections increased to 37, 48, and 40 sturgeon chub in the 199-mile Wolf Point to Yellowstone River segment of the Missouri River during the same time period (Dieterman et al. 1997; Young et al. 1998; Mike Ruggles, in litt. 1999).

In general, sicklefin chub are less numerous than sturgeon chub in the inter-reservoir reach between Fort Peck and Lake Sakakawea (river mile 1552). They were not found in the Milk River to Wolf Point segment. They are found in the Wolf Point to Yellowstone segment, but only in the lower reaches from the Redwater River confluence downstream. In 1996-1998, Benthic Fish Study researchers collected 6, 18, and 35 sicklefin chub from this reach of river (Dieterman et al. 1997; Young et al. 1998; Mike Ruggles, in litt. 1999).

The Montana Department of Fish, Wildlife, and Parks is conducting a study to evaluate the pallid sturgeon reintroduction program in the Missouri River above Fort Peck Reservoir. As part of this initiative, benthic trawl samples were collected in 1999 and 2000. In August of 1999, 308 benthic trawl tows yielded 561 sicklefin chub and 218 sturgeon chub. Sicklefin chub were the most common species collected, comprising 41.5 percent of the total catch, and sturgeon chub were the third most prevalent species, comprising 16.1 percent of the catch (Gardner 2000a). In August 2000, 145 sturgeon chub (32.0 percent of the catch) and 23 sicklefin chub (5.1 percent of the catch) were captured in 105 benthic trawl tows. Sturgeon chub and sicklefin chub were the second and third most common fish sampled by trawling in 2000 (Gardner 2000b).

Based on survey data collected during the past 24 years, the Montana Department of Fish, Wildlife, and Parks believes that the sturgeon chub and sicklefin chub are more widely distributed in Montana than previously described (Larry Peterman, Montana Department of Fish, Wildlife, and Parks, in litt. 1995; Patrick Graham, Montana Department of Fish, Wildlife, and Parks, in litt. 1997 and 2000). The Montana Department of Fish, Wildlife, and Parks summarized the records compiled in the Montana River Information System for sicklefin and sturgeon chub. Using collection records from 1977 to the present, sturgeon chub occur in 1,100 miles of streams in Montana, including the Missouri River and two tributaries (Redwater and Teton Rivers) and the Yellowstone River and four tributaries (Box Elder Creek, Powder River, Sears Creek, and Tongue River). Sicklefin chub populations, which were first documented in Montana in 1979, occur in 231 miles of the Yellowstone and Missouri Rivers. The Montana Department of Fish, Wildlife, and Parks also believes that researchers would increase their success in collection of these chub species by using trawling techniques and targeting different habitats.

North Dakota: With increased efforts in recent years to further document the distribution and abundance of the sturgeon chub and sicklefin chub, several researchers have surveyed the confluence area of the Yellowstone and Missouri Rivers in northwestern North Dakota since 1992 and documented populations of both species (Tews 1994; Welker 2000; Jason Lee, North Dakota Game and Fish Department, in litt. 1995; Greg Power, North Dakota Game and Fish Department, in litt. 1995 and 1997; Steve Krentz, U.S. Fish and Wildlife Service, pers. comm. 1995; Everett and Scarnecchia 1996; Liebelt 1996; Dieterman et al. 1997; Young et al. 1998; Mike Ruggles, in litt. 1999). Both species have been collected with benthic trawls and seines in the Lower Yellowstone River from the mouth upstream to the Montana border (river mile 0-14), as well as from the Missouri River from Lake Sakakawea upstream to the Montana border (river mile 1552-1585) (Welker 2000, Everett 1999)

During a pallid sturgeon study, Tews (1994) collected 47 sturgeon chub between river mile 2 and river mile 51 on the Yellowstone River by seining. Sturgeon chub were the second most common species collected by that method. In 1994, the North Dakota Game and Fish Department collected 30 sicklefin chub from the Missouri River west of Williston, North Dakota (Jason Lee, pers. comm. 1995).

In 1995, the University of Idaho and the North Dakota Game and Fish Department initiated a study to assess the distribution, relative abundance, and relative density of sturgeon chub and sicklefin chub along three segments of the Yellowstone and Missouri Rivers (Everett and Scarnecchia 1996). The study area included the Yellowstone River near its confluence with the Missouri River, and segments of the Missouri River near Williston and Bismarck, North Dakota. A total of 2,726 fish were collected using a benthic trawl and seine, including 64 sicklefin chub (2.3 percent) and 31 sturgeon chub (1.1 percent). No sicklefin or sturgeon chub were collected in the Bismarck segment. Both species were collected throughout 94 percent of the area sampled in the Williston and Yellowstone segments. Everett (1996) reported a substantial difference in the catch results produced by benthic trawl tows and seine hauls. Sicklefin chubs comprised 7.9 percent of the benthic trawl catch and 0.4 percent of the catch with seines. Sturgeon chub showed a similar pattern, comprising 3.2 percent of the benthic trawl catch and 0.4 percent of the fish collected by seining. Over 60 percent of the sicklefin and sturgeon chub sampled by Everett were collected from the main channel in water depths where seines are not usable. The mean depth at sicklefin and sturgeon chub collection sites was 6.8 and 2.5 meters, respectively. Sicklefin and sturgeon chub were the second and third most common cyprinids, respectively, from the Williston and Yellowstone study areas.

During the Benthic Fish Study in 1996-1998, 11, 16, and 1 sturgeon chub, and 28, 7, and 21 sicklefin chub were collected from the reach of river from the Missouri/Yellowstone River confluence downstream to the headwaters of Lake Sakakawea (Dieterman et al. 1997; Young et al. 1998, Tim Welker, in litt. 1999). Most of the specimens were collected with the benthic trawl. Sturgeon chub and sicklefin chub each comprised 1-2 percent of the total target fish catch in each year.

Additional non-standard benthic trawl and bag seine sampling was conducted by Welker (2000) in the confluence area of the Missouri and Lower Yellowstone Rivers. Welker sampled four river segments in the period July-September 1997 and 1998. The objective of this study was to obtain information on the distribution and habitat use of sicklefin chub, sturgeon chub, and other selected cyprinids. Welker collected 3,033 fish using seines to sample shallow channel border habitat (depths up to 1.5 meters) and a benthic beam trawl to collect fish in deep water habitat of the main channel. Table 7 summarizes collection results for sicklefin and sturgeon chub. Welker's collections were taken during the summer with mean water temperatures from 19.0 to 21.6° (66.2 to 70.9° F) for the four study segments (range 13.9 to 27.6° C - 57.0 to 81.7° F). The majority of sicklefin chub (92 percent) and sturgeon chub were captured in deep water habitat.

The work of Welker, Grisak (1996), and others highlights that the results of seine collections made during the summer in large rivers may not accurately represent the status of sicklefin and sturgeon chub populations.

Table 7. Number and Relative Abundance of Sicklefin and Sturgeon Chub in Samples Collected Near the Confluence of the Yellowstone and Missouri Rivers in 1997 and 1998 (Welker 2000).

	STURGEON CHUB		SICKLEFIN CHUB		TOTAL FISH
	#	% of catch	#	% of catch	#
BORDER CHANNEL					
(Seine)	**24**	**0.01**	**12**	**0.005**	**2,627**
MAIN CHANNEL					
(Benthic Trawl)	**131**	**32.3**	**135**	**33.2**	**406**
TOTAL	**155**	**5.1**	**147**	**4.8**	**3,033**

Fisher (1999) evaluated the importance of backwaters to native fish downstream of the Missouri/Yellowstone Rivers confluence in 1997 and 1998. As part of this study, Fisher found no direct evidence of physical inhabitance of backwater habitats during any life stage of sicklefin or sturgeon chub. He collected 21 young-of-year sturgeon chub and 5 sicklefin chub using a seine to sample shallow water habitat adjacent to sandbars in the main channel of the Missouri River (Fisher, pers. comm. 1999).

During August 1999 and 2000, the Montana Department of Fish, Wildlife, and Parks sampled the Missouri River from Williston, North Dakota, to the headwaters of Lake Sakakawea using a benthic trawl (James Liebelt in litt. 1999 and 2000). Liebelt collected 1,193 fish in 1999, including 132 sturgeon chub (11.1 percent of the catch) and 103 sicklefin chub (8.6 percent of the total catch). Sturgeon chub and sicklefin chub were the second and third most common fish collected, respectively. In August 2000, 44 sturgeon chub and 63 sicklefin chub were collected in the same study area. The sicklefin chub and sturgeon chub ranked third and fourth in abundance, respectively.

Farther downstream in North Dakota, sturgeon chub were found historically in the Little Missouri River, Box Elder Creek, Beaver Creek, Grand River, Green River, Heart River, and White Earth Creek (Reigh 1978, Reigh and Elsen 1979, Reigh and Owen 1979, U.S. Fish and Wildlife Service 1993b). Two sturgeon chub were collected from the Heart River in 1987 (Greg Power, in litt. 1995). The sturgeon chub has not been collected in recent years and is considered extirpated from these streams.

Sturgeon chub were collected at 41 collection sites throughout the length of the Little Missouri River in North Dakota in 1976 and 1977 and at locations in the Lower Yellowstone River (Reigh 1978, Reigh and Elsen 1979). The North Dakota Game and Fish Department collected 55 sturgeon chub, including 2 young-of-the-year, at 4 sample sites in the Little Missouri River during August 1984 (Greg Power, in litt. 1995). However, sturgeon chub were absent from surveys on the Little Missouri in 1990 (Werdon 1992), 1993 (Peterka 1993, Kelsch 1994) and in 1995 (Greg Power, pers. comm. 1997). In August 1997, the Service and the North Dakota Game and Fish Department jointly surveyed 38 sites on the Little Missouri River from the North Dakota/South Dakota border to Lake Sakakawea to establish a baseline inventory of the fish community, but did not collect any sturgeon chub (Wade King, in litt. 1998).

The 85-mile inter-reservoir segment of the Missouri River from the Garrison Dam (river mile 1389) downstream to the headwaters of Lake Oahe (river mile 1304) was sampled in 1996-1998 as part of the Benthic Fish Study. However, no sturgeon chub and no sicklefin chub were captured with the benthic trawl, bag seine, or any other fish collecting gear (Dieterman et al. 1997; Young et al. 1998; Tim Welker, in litt. 1999). Both species are likely extirpated from this reach of river.

South Dakota: Few historical records of the sturgeon chub exist in South Dakota. One collection was made in the 1890s by Evermann and Cox (1896) on the White River. As part of a systematic survey of rivers and streams in the 1950s by Bailey and Allum (1962), sturgeon chub were collected at two locations on the Grand River, at two locations on the Missouri River (one below Fort Randall Dam and one below Gavins Point Dam near Yankton, South Dakota), and at two locations on the White River. In the mid-70s, Bich and Scalet (1977) seined 25 sites on the Little Missouri River in Harding County and found sturgeon chub at 5 locations.

Additional sturgeon chub were not collected in South Dakota until 1993. In 1993 and 1994, the Service and South Dakota Department of Game, Fish, and Parks personnel used seines to document the presence of sturgeon chub at 10 locations on the White River, Little White River, and Cheyenne River (Douglas Hofer, South Dakota Game, Fish, and Parks, in litt. 1995). The species had been collected previously at one of these locations in 1950 by Bailey and Bailey, and in 1951 by Gibbs and Bartel (Bailey and Allum 1962). In 1994, the South Dakota Department of Game, Fish, and Parks conducted a limited seining effort at four sites on the Little Missouri River where Bich and Scalet had collected the species in 1976; however, they did not locate any sturgeon chub (Douglas Hofer, in litt. 1995).

A cursory survey of 35 headwater streams and 4 rivers west of the Missouri River in South Dakota was conducted in 1994 to document the presence and distribution of native fish. Forty-six sites were sampled using bag and wall seines, modified dip nets, and standard metal minnow traps. The survey confirmed the presence of the sturgeon chub in the Cheyenne, White, and Little White Rivers, and extended the range to the Bear-in-the-Lodge Creek, a tributary to the White River (Cunningham et al. 1995).

In 1995, the South Dakota Cooperative Fish and Wildlife Research Unit and the South Dakota Department of Game, Fish, and Parks embarked upon a cooperative effort to intensively survey fish and habitats in the State's major western streams and tributaries. The Upper Moreau River was sampled in 1995 and 1996, and the Belle Fourche and Cheyenne Rivers and tributaries were sampled in 1996 and 1997 by graduate student researchers. Fish populations were sampled using a variety of gear including seines, trap nets, hoop nets, and fyke nets. Chubs and other small fish were primarily collected with seines. No sturgeon chub were collected from either the Belle Fourche River (Doorenbos 1998) or Upper Moreau River (Loomis 1997). A total of 26 sturgeon chub were collected with seines at 5 of 9 reaches on the Cheyenne River between Angostura Reservoir and Lake Oahe (Hampton 1998). The sturgeon chub specimens represented less than 1 percent of the total fish collected (3,896). Although more sturgeon chub were collected in 1997 (15) than in 1996 (11), the species is considered rare (Charles Berry, South Dakota Cooperative Fish and Wildlife Research Units, pers. comm. 1997).

In 1997, the South Dakota Department of Game, Fish, and Parks funded additional surveys on the Cheyenne, Little Missouri, Little White, and White Rivers, and confirmed the presence of the sturgeon chub in all but the Little Missouri River (Cunningham and Hickey 1997). A total of 115 sturgeon chub were found at 10 of 12 sites on the White River. Sturgeon chub also were found at one location each on the Cheyenne River and Little White River. Although the Little Missouri River was intensively surveyed from the Wyoming border to the North Dakota border, sturgeon chub were not collected and appear to be extirpated. Cunningham and Hickey (1997) indicated the reason for extirpation is unclear, but speculate a possible relationship to drought, oil and gas development, and changes in turbidity which warrant research.

In 1998, the fishery resource of the White River again was surveyed. Ninety sturgeon chub were collected from 9 of 11 sample reaches and comprised about 3.6 percent of the total collection of 2,524 fish (Dave Fryda, South Dakota Cooperative Fish and Wildlife Research Unit, in litt. 1999). Cunningham (in litt. 1999) also seined the White River at the Badlands Bombing Range in 1998 and 1999. Approximately 50 sturgeon chub were collected from 2 sites on July 22, 1998, and about 25 specimens were netted from 3 locations on May 16, 1999. Based on this work and previous sampling efforts, Cunningham concluded that sturgeon chub are abundant in the White River.

The sicklefin chub was documented in South Dakota in 1952 (Bailey and Allum 1962) in the Missouri River at five widely separated locations (from the mouth of the Grand River in Corson County to a location below Yankton). The South Dakota Department of Game, Fish, and Parks believes that historical populations of the sicklefin chub in South Dakota were restricted primarily to reaches of the turbid Missouri River and now considers the sicklefin chub to be extirpated from the State because the main stem reservoirs and remaining riverine reaches no longer provide habitat for this species (Douglas Hofer, in litt. 1995).

The Missouri River from Fort Randall Dam (river mile 880) downstream to the mouth of the Niobrara River (river mile 845) and from Gavins Point Dam (river mile 811) to Ponca, Nebraska (river mile 753), was sampled by Benthic Fish Study researchers in 1996-1998. During the 3-year study, only one sicklefin chub (1996) and no sturgeon chub were collected in about 900 hours of effort with various gears (Dieterman et al. 1997; Young et al. 1998; Brad Young, South Dakota Cooperative Fish and Wildlife Research Unit, in litt. 1999). The one sicklefin chub was collected in a benthic trawl sample taken approximately 6 miles southeast of Burbank, South Dakota, on the western edge of Union County.

MIDDLE MISSOURI RIVER BASIN

The Middle Missouri River basin includes the main stem Missouri River from Gavins Point Dam (river mile 811) in southeastern South Dakota downstream through Iowa and Nebraska to Rulo, Nebraska (river mile 498), near the Nebraska/Kansas State line, and its tributaries, primarily the prairie streams west of the Missouri River.

Nebraska - Iowa: Historical records indicate that sturgeon chub and sicklefin chub were present in the Missouri River in Iowa and Nebraska. Historical abundance data for these species do not exist, although sturgeon chub were reported as abundant in the Missouri River at Sioux City, Iowa, in the late 19th century (U.S. Fish and Wildlife Service 1993b). Most catch records from the 1940s to the present document the capture of a single specimen. During the past 60 years sturgeon chub have been collected in the Missouri River from waters bordering Cass, Dixon, and Thurston Counties, Nebraska, and Mills County, Iowa (Larry Hutchinson, Nebraska Game and Parks Commission, in litt. 1999; Harlan and Speaker 1969). During the same time period, sicklefin chub have been documented in waters bordering Cass, Dakota, Dixon, Knox, Otoe, Richardson, Thurston, and Washington Counties, Nebraska, and Fremont, Harrison, Mills, Pottawattamie, and Woodbury Counties, Iowa (Hesse 1993a; Larry Hutchinson in litt. 1999). Records from the University of Michigan Museum of Zoology indicate that both species occurred in the Missouri River from the Niobrara River to Platte River in Nebraska, in the early 1940s and early 1950s (Douglas Nelson, University of Michigan Museum of Zoology, in litt. 1992).

Since the early 1950s, both the sturgeon chub and sicklefin chub have been rarely collected in the Middle Missouri River. Seining and electrofishing of unchannelized and channelized segments of the river in South Dakota, Nebraska, and Iowa in 1976 failed to capture any sturgeon chub and sicklefin chub (Kallemeyn and Novotny 1977). Hesse (1993a, 1993b, 1994) summarized a number of surveys and reported the results of extensive seining in the Nebraska reach of the

Missouri River from 1970 to 1993. He collected 45,500 small fish using seines during the 24-year study period. The only sicklefin and sturgeon chub collected were single specimens of each species that were captured in the far southeast corner of Nebraska in 1988.

Hesse (in litt. 2000) has continued annual sampling along the nearly 400 miles of the Missouri River that forms the eastern boundary of Nebraska. During the period from 1994 to 1999, 32,650 fish were netted using seines and winged trapnets. Sicklefin and sturgeon chub were the rarest species captured, with a total of two sturgeon chub netted in 1994 and one sicklefin chub collected in 1998 using seines.

Stasiak (1990) reported two sturgeon chub impinged at the Fort Calhoun Power Plant (river mile 646) in 1977 and 1978. The Service (U.S. Fish and Wildlife Service 1993a), citing others, reported the power plant impingement of a total of eight sicklefin chub in 1975, 1980, 1981, and 1982. Except for the 1975 record from river mile 646, all of these sicklefin chub were from the river downstream of the Platte River confluence (river mile 595). From the late 1970s to 1996, no sturgeon chub were collected in the Missouri River above the Platte River confluence. The standardized surveys of the Benthic Fish Study in 1996-1998 found few sturgeon chub (three in 1996 and two in 1997) and no sicklefin chub in the Missouri River segment between the Big Sioux River (river mile 740) at Sioux City and the Platte River confluence (Dieterman et al. 1997; Young et al. 1998; Mark Pegg, Iowa Cooperative Fish and Wildlife Research Unit, in litt. 1999).

Neither species have been collected during sporadic seining of the river near Sioux City, Iowa, over the past 20 years (Rod Tondreau, Western Iowa Technical University, pers. comm. 1995). In the early 1980s, Iowa State University conducted a Statewide fish survey and a Missouri River fish survey for the Corps. Both of these surveys sampled potential sturgeon and sicklefin chub habitat, but no fish were captured (Bruce Menzel, Iowa State University, pers. comm. 1995). Werdon (1992) sampled three Missouri River historical sturgeon chub collection sites in Woodbury County, Iowa, and Thurston and Dixon Counties, Nebraska, during 1989 and 1990. She did not collect either sicklefin or sturgeon chub at these locations.

Stasiak (1990) summarized the literature and historical records pertaining to sturgeon and sicklefin chub in the Nebraska reach of the Missouri River and conducted systematic collections from Sioux City, Iowa, to Rulo, Nebraska, in 1989. A total of 3,800 fish, representing 30 species, were collected using seines. No sicklefin or sturgeon chub were collected during this study. Stasiak concluded sicklefin and sturgeon chub populations are very rare in the Missouri River in Nebraska. In the reach of river between the Platte River confluence and the Nishnabotna River confluence (river mile 595-542), Stasiak (1990) reported two sturgeon chub impinged at a power plant in 1974 and 1982 near river mile 556. More recently, two sturgeon chub were collected at Brownville in 1994 following the 1993 Missouri River flood (Larry Hesse, Nebraska Game, Fish and Parks, pers. comm. 1995). Six sturgeon chub and one sicklefin chub were collected by Benthic Fish Study researchers during 1996 (Dieterman et al. 1997). Neither species were found in 1997 (Young et al. 1998), but one additional sturgeon chub was collected in this reach of the Missouri River in 1998 (Mark Pegg, in litt. 1999).

During 1999, the Nebraska Game and Parks Commission (1999) monitored fish populations in channelized and unchannelized sections of the Missouri River bordering Nebraska using a variety of gear including hoop nets, electrofishing equipment, gill nets, seines, trammel nets, and a semi-balloon otter trawl. Three sturgeon chub were collected at the Hamburg Bend mitigation site during the spring of 1999. An additional five sturgeon chub were collected in the fall in the dike field at the Tobacco Island mitigation site. The Nebraska Game and Parks Commission reports that the five sturgeon chub collected at Tobacco Island represent 23 percent of all sturgeon chub taken from the Nebraska section of the Missouri River since 1941. The sturgeon chub collected adjacent to Tobacco Island were collected with the otter trawl in mean water depths ranging from 3.1 to 5.8 m (10.2 to 19.0 ft). One sicklefin chub was taken in a benthic trawl sample at the Goose Island control site. Current records suggest that both the sturgeon chub and sicklefin chub exist in very low numbers in the channelized Iowa/Nebraska reach of the Missouri River below the Platte River confluence.

Occurrence records of the sturgeon chubs collected in Nebraska's tributaries to the Missouri River prior to the 1950s are found in Evermann and Cox (1896), Bailey and Allum (1962), and Reno (1969). They reported that the sturgeon chub occurred at scattered locations in the lower Niobrara River, the Republican River, Loup River, Elkhorn River, Platte River, and Bazile Creek. Between 1984 and 1988, the Nebraska Department of Environmental Quality collected more than 70,000 small fish from 350 stream sites across Nebraska; however, no sturgeon chub were collected (Bazata 1991).

In 1989-1990, Werdon (1992) resurveyed historic sites documented by Bailey and Allum (1962) and Evermann and Cox (1896) on the Platte River, but did not relocate sturgeon chub. Werdon (1992) also failed to relocate sturgeon chub at three sites previously documented on the Republican River by Bailey and Allum (1962); one site documented on the Loup River; a site on the eastern Elkhorn River; and a site noted by Evermann and Cox (1896) on Bazile Creek in Knox County.

Sturgeon chub were collected from the Platte River in Dodge County in 1987 (Peters et al. 1989) and Sarpy County in 1991 (Thomas Labedz, University of Nebraska State Museum, in litt. 1992) in low numbers. During extensive sampling of the lower Platte River in 1987, two sturgeon chub were collected near Fremont in Dodge County; five more specimens were collected further downstream in Sarpy County in 1991 (Rowe 1992, Larry Hutchinson, in litt. 1999). On September 19, 2000, the Missouri Department of Conservation and the University of Nebraska collected three sturgeon chub in the Platte River using a benthic trawl, approximately 12 miles upstream of the confluence with the Missouri River (Hrabik in litt. 2000). These were the first specimens collected in the Platte River since 1991. Hrabik suggests that sturgeon chub in the Platte River are uncommon, but may not be as rare as previously suspected.

Outside of the main stem Missouri River, no recent records of sicklefin chub in Nebraska exist. Earlier, Morris (1960) reported collecting sicklefin chub from the Platte River near North Bend and Schuyler, Nebraska, in 1959. However, Stasiak (1990) reported that these specimens were probably misidentified sturgeon chub.

LOWER MISSOURI RIVER BASIN

The Lower Missouri River basin includes the main stem Missouri River and associated tributaries in Kansas and Missouri, downstream of Rulo, Nebraska (river mile 498), to the mouth of the river (river mile 0) north of St. Louis.

Kansas: Historically, the sturgeon chub was a component of the fish fauna of the Missouri and Lower Kansas Rivers in Kansas. The sicklefin chub was present in the Missouri River and rarely captured from the Lower Kansas River (Cross 1967). The last known sturgeon chub and sicklefin chub captured from the Lower Kansas River occurred in 1979 and 1994, respectively (U.S. Fish and Wildlife Service 1993b; Kate Shaw, University of Kansas History Museum, pers. comm. 1995). Prior to the 1994 collection of the sicklefin chub, the next most recent collection record was from 1952 (Cross et al. 1982). Both the 1952 and 1994 collection were presumably the result of migration during flood flows on the Kansas River in 1951 and 1993, respectively. In the Lower Missouri River basin, it is likely that sicklefin chub populations presently occur only in the main stem Missouri River.

In Kansas, numerous field collections were completed in 1992 on the Missouri River between White Cloud, Kansas, and Leavenworth, Kansas, and on the Kansas River from Lawrence, Kansas, to the confluence with the Missouri River. A total of eight sturgeon chub were captured from seven localities on the Missouri River, and no sturgeon chub were captured from the Kansas River. No sicklefin chub were captured from either river (Thomas Wenke, Fort Hays State University, pers. comm. 1993 and 1995). A survey of the Kansas, Republican, and Smoky Hill Rivers in 1991-1992, on or near Fort Riley, Kansas, did not find sturgeon chub. The last collection of the species from this locale was 1964 (Wenke et al. 1993). Werdon also unsuccessfully sampled this locale for sturgeon chub in 1991 (U.S. Fish and Wildlife Service 1993b). In August 1994, three reaches on the Lower Kansas River were sampled for small fishes. No sturgeon chub or sicklefin chub were captured (Vernon Tabor, U.S. Fish and Wildlife Service, in litt. 1994). The reach of the lower Kansas River where sturgeon chub were last captured in 1979, was sampled again in 1997 and 1999. No sturgeon chubs were collected.

A collection locale for both species on the Kansas River at Lawrence, Kansas, has been sampled several times annually since 1951 by staff from the Division of Fishes, University of Kansas Museum of Natural History. Historical fish collections from this locale date to the late 1800s. The last capture of sturgeon chub and sicklefin chub from this area was 1972 and 1994, respectively (Frank Cross, University of Kansas, retired, pers. comm. 1995; Kate Shaw, University of Kansas Natural History Museum, pers. comm. 1995).

Missouri: In 1945, Fisher (1963) established 11 collection sites in the Lower Missouri River from the Iowa-Missouri State line to the confluence of the Missouri River with the Mississippi River. Pflieger and Grace (1987) described Fisher's work as the first thorough survey of fish in the Lower Missouri River. The purpose of this study was to document the fish community before further impoundment of the Missouri River. Only Fort Peck Dam in Montana was in place when Fisher's study was conducted. Fisher primarily used seines to collect small fish from April through October 1945. He collected 24,600 fish, including 66 sicklefin chub and 23 sturgeon chub.

No systematic surveys were conducted during the 1970s, but sturgeon chub and sicklefin chub were collected from northwest and central Missouri by various collectors (U.S. Fish and Wildlife Service 1993a, 1993b). Grace (1985) used seines to sample shallow water areas surrounding two sand islands in the Lower Missouri River. The islands, located at river mile 177.4 and 169.8, were sampled at approximately monthly intervals from July 1982 to October 1983. Sicklefin and sturgeon chub each comprised 1.5 percent of the total catch. Grace noted that catch varied widely by season. Catch rates for sicklefin chub were high during early September 1982 and February through June 1983. During this time, they comprised between 6.9 and 12.7 percent of the catch. During the period from July to October 1983, no sicklefin chub were collected. The catch rates for sturgeon chub were highest from December to June, representing between 8.1 to 24.2 percent of the catch. Few sturgeon chub were collected during the summer.

Data collected in 1982-83 (Grace and Pflieger 1985) indicate that the species' distribution in this reach of the Missouri River remained similar to the 1946-1969 period for sturgeon chub and the 1905-1969 period for the sicklefin chub. Grace and Pflieger (1985) collected 376 sturgeon chub from 7 of 13 sampling sites, although most populations were concentrated in the lower river below central Missouri. The sampling sites were located along the length of the Missouri River in the State of Missouri. Sturgeon chub were collected at three of the eight pre-1969 collection sites. They also reported 590 sicklefin chub from 9 of 13 sampling sites, which corroborated earlier collection locations and river reaches. Sampling effort was not reported.

Pflieger and Grace (1987) used the results presented by Fisher (1963), data they collected (Grace and Pflieger 1985) and studies conducted by other biologists to evaluate how the relative abundance and distribution of fish has changed in the Lower Missouri River from 1940 to 1983. They reported the percent composition of large (≥150 mm total length - 5.9 in) and small (<150 mm total length) fish for the time periods 1940-45, 1962-72, 1978-83. Pflieger and Grace (1987) concluded that both sicklefin and sturgeon chub increased in abundance in the Missouri River below Kansas City. They speculated the Lower Missouri River may be the last stronghold of the sturgeon chub and sicklefin chub.

Gelwicks et al. (1996) revisited historic collection sites that had been sampled by Fisher, Pflieger, and Grace to determine the distribution and relative abundance of sicklefin and sturgeon chub. Thirteen historic collection sites from the Iowa-Missouri border to the mouth of the Missouri River were seined from October 31 to November 15, 1994. Gelwicks collected

6,560 fish, including 3,586 small fish, representing 17 species. The collection contained 163 sicklefin chub and 114 sturgeon chub. Sicklefin chub were captured at all 13 collection sites and sturgeon chub were found at 11 of 13 sites. Gelwicks also collected 18 speckled chub and sturgeon chub hybrids. Table 8 summarizes the results of the studies that have evaluated the relative abundance of small fish in the Lower Missouri River. The table is based on data presented by Pflieger and Grace (1987) and Gelwicks (et al. 1996).

Table 8. Percent Composition of Sicklefin Chub and Sturgeon Chub in the Small Fish Population of the Lower Missouri River 1940-1994.

	1940 - 1945	1962 - 1972	1978 - 1983	1994
SICKLEFIN CHUB	**0.7**	**2.1**	**2.8**	**4.5**
STURGEON CHUB	**0.1**	**0.2**	**0.8**	**3.2**

Gelwicks' sampling indicated an increase in the distribution and abundance of sicklefin and sturgeon chub. It should be noted that while Gelwicks used seines to be consistent with previous studies, his collections occurred in November with water temperatures ranging from 11 to 13° C (51.8 to 55.4° F). Work conducted by Grace (1985), the Missouri Department of Conservation, and others suggest that sicklefin and sturgeon chub may be more commonly found in shallow water areas when the water temperature is less than 15° C (59° F) and thus more readily collected with seines.

In July and August 1997, Grady and Milligan (1998) also sampled historic collection sites on the Lower Missouri River. Nine of 13 historic sites and 2 new sites were seined and trawled. High water conditions, which persisted throughout the 1997 field season, prevented sampling at four of the historic collections sites. Sixty sicklefin chub and 29 sturgeon chub were collected. All sturgeon chub captured during this study and 59 of 60 sicklefin chub were collected in benthic trawl tows.

Grady and Milligan (1998) compared their data to long-term Missouri River data sets from 1944 to 1997, including those reported in Pflieger (1975), Grace and Pflieger (1985), Pflieger and Grace (1987), and Gelwicks et al. (1996). The primary purpose of this study was to evaluate if populations of sicklefin chub, sturgeon chub, and other selected species of cyprinids were declining. Grady and Milligan analyzed the existing data sets to determine the probability of collecting chubs (i.e., presence or absence) over time. They found that the probability of collecting sicklefin chubs in the Lower Missouri River increased from 1945 to 1997. During the same time period, the probability of capturing sturgeon chub remained stable. Grady and Milligan cautioned that although the sturgeon chub population in the Lower Missouri River appears stable, it has declined dramatically throughout most of its range due primarily to changes in the river's channel, turbidity, and hydrograph.

Grady and Milligan (1998) used the most complete long-term data set characterizing sicklefin and sturgeon chub populations throughout their range. Efforts to statistically evaluate the abundance of chubs over time were hampered by the lack of sampling effort data from previous collections. Their study highlights the limitations associated with comparison of historical and current data sets collected during different years, river conditions, and with different sampling protocol. Furthermore, most of the studies have not accounted for differences in gear selection, especially between benthic trawls and seines. Under certain conditions, benthic trawls have become a reliable gear to collect both chub species.

Frank Cross (pers. comm. 1995) suggests that a stable or slightly increasing population of the sturgeon chub in the Lower Missouri River since the 1960s may be related to changes in substrate as sediments are trapped in main stem and tributary reservoirs, and concurrently, the amount of fine gravel, rather than sand, increases in the lower river. In addition to substrate changes, the abundance of sicklefin chub, and to a lesser degree, sturgeon chub, progressively increases downstream, paralleling an increase in the abundance of sandbars, shallow-water habitat, warmer water temperatures, higher turbidity, a more natural hydrograph due to tributary influence and thus, an increase in the frequency of higher river stages and floods during the species' spawning period.

The Service's Columbia Missouri Fishery Resource Office conducted monitoring surveys in a 7-mile reach of the lower Missouri River (river mile 213-219) from 1997 to 1999. A total of 480 sicklefin chub were collected using a benthic trawl (440) and Wisconsin-type mini-fyke nets (40). Seines also were extensively used throughout the 7-mile monitoring area; however, neither species were collected. Over 97 percent of the sicklefin chubs captured were from Jameson Island (river mile 219), a sandbar/wing dike complex and Lisbon Chute (river mile 217), a naturally formed 2-mile side channel containing point/mid-channel sandbars. During the same study, 13 sturgeon chub were captured in benthic trawl tows (Louise Mauldin, U.S. Fish and Wildlife Service in litt. 2000).

The floods of 1993 and 1995 significantly changed the physical character of portions of the Lower Missouri River floodplain when levees failed, and scour holes developed, resulting in the connection of the river and floodplain, which is important to the survival of many native Missouri River species. Analysis of scour holes by the University of Missouri in 1994-1997 documented the use of these habitats by both sicklefin and sturgeon chub and their high value as nursery habitat for larval, juvenile, and young-of-year fish (John Kubisiak, Missouri Cooperative Fish and Wildlife Research Unit, in litt. 1997; John Tibbs, Missouri Cooperative Fish and Wildlife Research Unit, in litt. 1997; Doug Dieterman, Missouri Cooperative Fish and Wildlife Research Unit, in litt. 1999).

A total of 23 sicklefin chub and one sturgeon chub were collected by Kubisiak during 1,214 seine hauls in the Lower Missouri River (river mile 262-160) from April through September in 1995 and 1996. All fish were collected in scour holes either continuously or seasonally connected to the river. Tibbs provided evidence of reproduction in the Lower Missouri River in 1996. He collected 9 juvenile sicklefin chubs and 133 sicklefin and/or sturgeon chub larvae from 9 scour holes connected to the Missouri River between river mile 300 and 160 from July through September 1996. Gelwicks also documented sicklefin/sturgeon chub larvae at a scour hole near river mile 261 in 1996 and 1997 (Doug Dieterman, in litt. 1999).

In 1996, 1997, and 1998, researchers with the Benthic Fish Study collected 7, 15, and 9 sturgeon chub and 11, 7, and 1 sicklefin chub, respectively, in various river segments between Rulo, Nebraska (river mile 498), and the Grand River (river mile 250) (Dieterman et al. 1997; Young et al. 1998; Pat Braaten, Kansas Cooperative Fish and Wildlife Research Unit, in litt. 1999). During the same study period, they collected 2, 9, and 4 sturgeon chub and 9, 37, and 46 sicklefin chub, respectively, from several locations between Glascow, Missouri (river mile 220), and the mouth of the Missouri (river mile 0) (Dieterman et al. 1997; Young et al. 1998; Doug Dieterman, in litt. 1999). In general, more sturgeon chub and sicklefin chub were collected in 1997 and 1998 than in 1996. Most of the chubs collected were taken by a benthic trawl, and the higher numbers in 1997 and 1998 may reflect the increased sampling effort in those years. The Benthic Fish Study confirmed that sturgeon chub and sicklefin chub increase in abundance progressively downstream and are most common in the lower 130 mi of the Missouri River below the Osage River confluence (river mile 130.4). Although analyses are preliminary, age-growth relationship data (1997-1999) for the sicklefin chub from the Benthic Fish Study suggest that successful recruitment is occurring in those portions of the Missouri River system where the sicklefin chub is still found (Pat Braaten, in litt. 1999).

Etnier (David Etnier, University of Tennessee, in litt. 1996) documented reproduction in the Lower Missouri River below river mile 93 during September and October 1996. He collected 210 sicklefin chub and 81 sturgeon chub from 8 lower river sites. Most of these fish were young-of-year or juveniles. Their capture followed the high 1996 spring flows on the Lower Missouri River. Etnier considered both species as the most abundant cyprinids in his seine hauls. On April 7, 2000, the Missouri Department of Conservation Open River Field Station collected fish samples at Pelican Island. This site is located in the lower Missouri River, approximately 15 miles upstream of the confluence with the Mississippi River. They collected 451 sturgeon chub and 30 sicklefin chub in four benthic trawl tows. The highest number of chubs were collected over clean gravel substrate, approximately 65 m (213 ft) off shore in water depths averaging 1.1 m (3.6 ft). Yearling sturgeon chub and sicklefin chub also were collected near shore with seines in water less than 0.75 m (2.5 ft) deep (Hrabik, Missouri Department of Conservation, in litt. 2000).

MIDDLE MISSISSIPPI RIVER

Collection records for the Middle Mississippi River (Missouri River confluence to the Ohio River confluence) provide an incomplete picture of sicklefin and sturgeon chub populations and how they have changed with time. In general, few records exist, which may reflect little sampling effort or low populations. Researchers familiar with the distribution and relative abundance of fish in the Middle Mississippi River have until recently considered both species to be rare (Pitlo et al. 1995). Recent sampling efforts using an experimental benthic trawl suggest that sicklefin and sturgeon chub may be more common than previously believed.

Missouri - Illinois: Several records exist from the University of Michigan Museum of Zoology (Douglas Nelson, in litt. 1992) for both the sturgeon chub and sicklefin chub, and from Bailey and Allum (1962) for the sturgeon chub in the Middle Mississippi River in Illinois and Missouri from the late 1930s and early 1940s. As is the case with most historical survey data for these chub species, relative abundance data does not exist for the Mississippi River. These records indicate that sturgeon chub were collected from near Chester, Illinois, river mile 110, downstream to the river's confluence with the Ohio River. Smith (1979) reported records from Madison and Union Counties, Illinois, but dates of collection are unknown. Madison County is in the uppermost reach of the Middle Mississippi River near St. Louis and the Missouri River confluence. Records from the University of Kansas Natural History Museum list the sicklefin chub in the Mississippi River in St. Charles, Perry, Scott, and Mississippi Counties, Missouri, in the early 1960s (Kate Shaw, pers. comm. 1995).

Klutho (1983) collected 5,480 fish seining shoreline habitat (depths up to 1.5 m - 4.9 ft) at 2 locations near Grand Tower, Illinois, from April 1978 to February 1983. He classified the sturgeon chub as rare and the sicklefin chub as common at Grand Tower, Illinois. During this study, 61 species were collected, including 5 juvenile sturgeon chub and 65 sicklefin chub. The sicklefin chub ranked 13th in abundance and represented slightly over 1 percent of the total catch. Statistical analysis revealed that the presence of sicklefin chub was correlated with high water levels, sand substrate, and water temperatures ranging from 2 to 9° C (35.6 to 48.2° F). Klutho found sicklefin chub were most prevalent in shallow water during the late winter and early spring. He hypothesized that sicklefin chub may move inshore to avoid being washed downstream during period of high flows.

Grace and Pflieger (1985) surveyed 16 sites on the Mississippi River bordering Missouri, including 10 locations along the Middle Mississippi River. They collected 54,900 fish, representing 84 species using seines and electrofishing equipment. Small fish collections, using seines to sample shallow water habitat, were conducted between June 27 and September 15, 1983. No sicklefin or sturgeon chub were collected in samples taken from the Mississippi River.

In Illinois, both chub species are considered to be rare; however, the sicklefin chub is reported to be more common than the sturgeon chub (Smith 1979). Since 1986, the Illinois Department of Conservation has seined 33 sites between Lock and Dam 22 and Thebes, Illinois, 3 times each summer to monitor reproduction and recruitment of riverine fishes. No sturgeon chub or sicklefin chub have been collected since monitoring began (Butch Atwood, Illinois Department of Conservation, pers. comm. 1995). Sicklefin chub are occasionally found at Grand Tower during annual qualitative sampling; the fish are suspected to be young-of-year and no adults are collected (Brooks Burr, pers. comm. 1995).

In 1996, Southern Illinois University conducted a study for the Illinois Department of Natural Resources to determine the present distribution and abundance of the sturgeon chub and sicklefin chub in Illinois, and among other objectives, to document historical changes in distribution and abundance (Piller et al. 1996). Seines were used to provide a basis for comparison of historical and current data. Eleven sites (both historical and new localities with potential suitable habitat) were sampled from September 1995 to July 1996; however, neither species was collected. Both species were previously present in collections from the early 1980s and mid-1990s following floods, but absent during many intervening years. They concluded, based on gear compatibility, that both species had declined in distribution and abundance in the Middle Mississippi River.

Etnier (David Etnier, in litt. 1996) collected two sturgeon chub and one sicklefin chub in the Middle Mississippi River at three sites from Scott County, Missouri, to the mouth of the Missouri River during the fall of 1996.

The Missouri Department of Conservation Open River Field Station has used a variety of gear since 1991 to sample fish populations in the Middle Mississippi River (Hrabik, in litt. 2000a). In 1997, they added small mesh netting and made other modifications to a standard slingshot-balloon trawl. The modified or experimental benthic trawl has allowed researchers to more effectively sample small fish, including sicklefin and sturgeon chub. The experimental benthic trawl has permitted fishery biologists to collect small fish in deep water habitat where seining or other collection methods can not be used or are ineffective.

During the 9-year period from 1991 to 1999, the Open River Field Station did not collect sturgeon chub in the Middle Mississippi River using a variety of gear, including seines, minnow fyke nets, and standard trawling equipment. However, during the past 4 field seasons (1997-2000), 227 sturgeon chub were collected using the modified or experimental benthic trawl. The number of sicklefin chub collected also has substantially increased. From 1991 to 1999, 55 sicklefin chub were captured using various gear. Since 1997, 209 sicklefin chub have been collected with the experimental benthic trawl in the Middle Mississippi River (Hrabik and Herzog, Missouri Department of Conservation, in litt. 2000a,b).

Hrabik (in litt. 1993 and 1997) had previously indicated that no conclusive evidence existed to suggest that sicklefin chub and sturgeon chub in the Middle Mississippi River were members of a viable, self-sustaining population. Prior to employing the experimental benthic trawl in 1997, Open River Field Station biologists considered both species to be waifs or transient fish from the Lower Missouri River. Trawling data collected during the past four field seasons confirm the presence of a viable population of both sicklefin and sturgeon chub in the Middle Mississippi river. Based on the short-term data set collected during the past 4 years, Hrabik and Herzog (in litt. 2000a, b) indicated that sturgeon chub are uncommon, but not rare, and their numbers are steady to slightly increasing. Sicklefin chub are uncommon and perhaps borderline rare in the Middle Mississippi River. Collections made during the past four field seasons suggest that sicklefin chub numbers are slightly decreasing. However, additional data is needed to establish reliable population trends for sicklefin and sturgeon chub.

LOWER MISSISSIPPI RIVER

Missouri - Kentucky - Tennessee - Arkansas - Mississippi - Louisiana: Few historical records exist for sturgeon chub and sicklefin chub from the Lower Mississippi River from the mouth of the Ohio River to the Gulf of Mexico. Records are more sparse on the Lower Mississippi River system than the Middle Mississippi River and have been reported only from the main stem Mississippi River and not from tributaries. The lack of records for sicklefin chub and sturgeon chub from the Lower Mississippi River may be due, in part, to a reduced sampling effort and limited trawling by comparison to the effort expended on the Missouri and Middle Mississippi Rivers.

The most current fishery study documenting the presence of sicklefin and sturgeon chub is ongoing work being conducted by the Missouri Department of Conservation Open River Field Station. During 2000, the Wolf Island Chute, a 4.3-mile area located approximately 24 miles downstream from the mouth of the Ohio River was sampled on three occasions. Complete data summarizing these collections are not currently available; however, based on initial field assessments, Hrabik and Herzog (in litt. 2000a) believe a viable population of both sicklefin and sturgeon chub exist in the Wolf Island area. Both species are less abundant in the Lower Mississippi River than the Middle Mississippi River, despite what appears to be an abundance of adequate habitat.

Other occurrence records for sicklefin and sturgeon chub in the Lower Mississippi River document small collections (one to three fish) over the last 60 years. Three pre-1980 records exist for the sicklefin chub from the Lower Mississippi River. Etnier and Starnes (1993) reported a 1940 record from the Missouri shore of the Mississippi River at Cottonwood Point, Pemiscot County, across the river from Dyer County, Tennessee. Burr and Warren, Jr. (1986) reported a 1944 University of Michigan Museum of Zoology record from the Mississippi River at the Ohio River confluence, Alexander County, Illinois, for both the sturgeon chub and sicklefin chub. Records from the University of Kansas Natural History Museum (Kate Shaw, pers. comm. 1995) document the occurrence of the sicklefin chub from the Lower Mississippi River (Mississippi County, Missouri) in the early 1960s.

Etnier and Starnes (1993) reported two records of sturgeon chub from near the Hatchie River confluence in Tipton County, Tennessee. These records are apparently the same records mentioned by Robison and Buchanan (1988) and the same as a 1981 record cited in U.S. Fish and Wildlife Service (1993b). Robison and Buchanan (1988) also reported one collection (one specimen) from Mississippi County, Arkansas, which is on the opposite shore from Tipton County.

In addition to these records, Robison and Buchanan (1988) reported a 1980 Mississippi River sicklefin chub record of two small specimens from near Blytheville, Mississippi County, Arkansas. They also noted collection of the first record (1980, three specimens) of this species in Tennessee, from the Mississippi River in Lauderdale County (across the river from Mississippi County, Arkansas). No more recent collections of the sicklefin chub or sturgeon chub have been reported in Arkansas (Henry Robison, University of Southern Arkansas, pers. comm. 1995; Cindy Osborne, Arkansas Natural Heritage Commission, pers. comm. 1997). In 1994, sicklefin chub were captured near the mouth of the Obion River (Dyer and Lauderdale Counties, Tennessee), about 19 km (12 mi) downstream of Cottonwood Point, by Dr. David Etnier, University of Tennessee at Knoxville (Ron Cicerello, Kentucky State Nature Preserves Commission, pers. comm. 1995). The 1944 record from Missouri, the 1980 records from Blytheville and Lauderdale Counties, and Etnier's 1994 record, all come from the same vicinity.

Extensive sampling of main channel, side channel, and inlet habitats around seven sand islands in the Lower Mississippi River in Missouri and Kentucky by seining during May-July of 1993 and 1994 failed to capture sturgeon chub, but captured one juvenile sicklefin chub (John Tibbs, in litt. 1995). This specimen was collected near river mile 835 in Fulton County, Kentucky.

The sturgeon chub has not been reported from Mississippi and only four collection records of the sicklefin chub exist for the State. In the spring of 1973, two sicklefin chub were collected from the Vicksburg Power Plant intake screen and a single specimen was caught in the Mississippi River from waters bordering Claiborne County, Mississippi (Guillory 1979). During the fall of 1973, two additional sicklefin chub were captured as part of the Grand Gulf Nuclear Power Plant survey. They were collected from the Mississippi River using a 16-foot otter trawl (Todd Slack, Mississippi Museum of Natural History, pers. comm. 2000). Ross (1991) reported a single record of sicklefin chub in Mississippi from near Vicksburg. No more recent collections exist (Steve Ross, pers. comm. 1995 and 1997). Ross categorized the conservation status of the sicklefin chub in Mississippi as rare (usually collected as single individuals) and peripheral (a species whose main distribution is outside of Mississippi and is only represented in the State by occasional waifs). In Louisiana, the sturgeon chub is rare and represented by one specimen, which was collected in the Mississippi River in West Feliciana Parish (Henry Bart, Tulane University, pers. comm. 1995). No records of the sicklefin chub have been reported for Louisiana.

V. SICKLEFIN CHUB STATUS SUMMARY

Since 1993, when the Service completed a Sicklefin Chub Status Report, surveys have been conducted throughout most of this species' historic range. These studies indicate that sicklefin chub are more common and more widely distributed than previously believed. The efficiency of sampling techniques have dramatically improved with the use of benthic trawls that have been modified to collect small fish. Benthic trawls have permitted sampling in deep water habitats where seines, the traditional cyprinid collection method, are ineffective or cannot be used.

Collection records for sicklefin chub indicate that this species historically occurred in 70 miles of the Lower Yellowstone River, 1,950 miles of the main stem Missouri River, and 1,150 miles of the Mississippi River, below the mouth of the Missouri River. Based on field studies conducted during the past decade, sicklefin chub currently occupy approximately 1,090 miles in the Missouri River drainage or 54 percent of its historic range.

Information documenting the presence of sicklefin chub in the Mississippi River is limited by comparison to the Missouri River data set. Field studies conducted by the Missouri Department of Conservation since 1997 have documented viable populations of sicklefin chub in the Middle Mississippi River and in the Wolf Island area of the Lower Mississippi River. Historic collections of sicklefin chub in the Lower Mississippi River below Wolf Island are rare and generally document the presence of an individual fish. Sufficient data does not exist to determine if the Lower Mississippi River provided important habitat for sicklefin chub.

Recent studies using benthic trawls indicate that sicklefin chub are a significant part of the fishery at three locations in the Missouri River drainage--above Fort Peck Reservoir in Montana; the Yellowstone/Missouri River confluence area in North Dakota and Montana; and the lower Missouri River in Missouri. Grisak (1996) used both seines and a benthic trawl to sample the fish population in the Missouri River above Fort Peck Reservoir in 1994 and 1995. He found sicklefin chubs comprised 21.9 percent of the benthic trawl catch and only 0.08 percent of the catch with seines. Sicklefin chubs were the second most common species collected in benthic trawl tows. In 1999 and 2000, Gardner (2000a,b) sampled the same general area as Grisak. The sicklefin chub was the most common species collected in 1999 (41.5 percent of the catch) and the third most common species collected in 2000 (5.1 percent of the catch). Welker (2000) used both seines to sample shallow border channel habitat and a benthic trawl to sample deep water habitat in the Yellowstone/Missouri River confluence area in 1997 and 1998. Sicklefin chub were the most common species collected in benthic trawl tows, comprising 33.2 percent of the trawl catch. By contrast, only 12 sicklefin chub were collected in seine hauls (0.005 percent of the catch using seines). Liebelt (in litt. 1999) sampled the Missouri River above the headwaters of Lake Sakakawea in 1999. Sicklefin chub were the third most common species collected, making up 8.6 percent of the catch. Grady and Milligan (1998) sampled the Missouri River in Missouri in 1997. They collected 3,934 fish in seine hauls, including one sicklefin chub. By contrast, sicklefin chubs were the second most common species collected with a benthic trawl (8.4 percent of the catch).

Construction of six dams and reservoirs on the main stem Missouri River from 1937 to 1964 as part of the Pick Sloan Plan and their continued operation is the major factor that has impacted sicklefin chub populations. Completion of the dams converted 800 miles of turbid, riverine habitat to lentic systems. Sicklefin chubs, which are highly adapted to conditions found in large, turbid river systems, have been extirpated from the reservoirs and the free-flowing reaches below Garrison, Oahe, Big Bend, and Fort Randall Dams in North and South Dakota. Sicklefin chub likely become easy prey for sight-feeding piscivorous fish in the relatively clear water conditions found in these areas. Sicklefin chub are found in low numbers in the Missouri River from Gavins Point Dam downstream to the Missouri border.

VI. STURGEON CHUB STATUS SUMMARY

Historically, the sturgeon chub occurred throughout 2,100 miles of the main stem Missouri River and 1,150 miles of the main stem Mississippi River. The species also was found in the Yellowstone River in Montana and North Dakota and 30 tributaries to the Yellowstone and Missouri Rivers. The sturgeon chub occurred in portions of four tributaries in Wyoming, nine in Montana, five in North Dakota, six in South Dakota, six in Nebraska, and four in Kansas. Tributaries such as the Powder River, which provides sturgeon chub habitat in both Wyoming and Montana, are included in the tallies for both States. Other tributaries that historically provided sturgeon in two States include the Big Horn, Little Missouri, and Republican Rivers.

Sturgeon chub currently occupy approximately 1,155 miles or about 55 percent of its former range in the Missouri River. The species also continues to be found in 11 of 30 tributaries to the Yellowstone and Missouri Rivers that were documented as providing sturgeon chub habitat. As with the sicklefin chub, information documenting sturgeon chub populations in the Mississippi River is limited by comparison to the Missouri River data set. Field studies conducted by the Missouri Department of Conservation since 1997 have documented a viable population of sturgeon chub in the Middle Mississippi River and in the Wolf Island area of the Lower Mississippi River (Hrabik and Herzog 2000a,b). Historic collections of sturgeon chub below Wolf Island are rare and do not provide adequate information to assess if this area historically provided important sturgeon chub habitat.

The distribution of sturgeon chub in the main stem Missouri and Mississippi Rivers is similar to that of the sicklefin chub. Both species are highly adapted for conditions found in free-flowing rivers with high turbidity levels in the main channel. Like the sicklefin chub, sturgeon chub comprise a significant portion of the Missouri River fish community above Fort Peck Reservoir in Montana, in the Yellowstone/Missouri River confluence area in Montana and North Dakota, and in the Lower Missouri River in Missouri.

Recent studies using benthic trawls designed to collect small fish from deep water areas of the main channel have increased information about the distribution and relative abundance of sturgeon chub. Grisak (1996) conducted the first studies using a benthic trawl with small mesh netting to specifically collect cyprinids and other small fish in the Missouri River. He sampled the Missouri River above Fort Peck Reservoir in 1994 and 1995 and found that sturgeon chub comprised 18.9 percent of the benthic trawl catch and only 0.16 percent of the catch with seines. In Grisak's study, sturgeon chub were the third most common species collected in benthic trawl tows. In 1999 and 2000, Gardner (1999, 2000) sampled the same general area as Grisak.

Gardner collected 218 sturgeon chub (16.1 percent of the catch) in August 1999 and 145 sturgeon chub (32.0 percent of the catch) in August 2000 using a benthic trawl. Welker (2000) used both seines and a benthic trawl to sample the fish population in the Yellowstone/Missouri River confluence area in North Dakota. Sturgeon chub were the second most common species collected (32.3 percent of the catch) in benthic trawl samples taken in the main channel. Shallow border channel areas also were sampled with seines. Sturgeon chubs were rare in seine samples, representing less than 0.01 percent of the catch. Liebelt (in litt. 1999) sampled a reach of the Missouri River from Williston, North Dakota, downstream to the headwaters of Lake Sakakawea in August 1999. Sturgeon chubs were the second most common species collected, representing 11.1 percent of the catch in benthic trawl tows. In Missouri, Grady and Milligan (1998) sampled the Lower Missouri River in 1997. They collected 3,934 fish with seines; however, no sturgeon chub were captured. Sturgeon chub ranked fourth in abundance for fish collected in benthic trawl tows (4.1 percent of the catch).

VII. DISTINCT POPULATION SEGMENTS

The Service and the National Marine Fisheries Service have adopted criteria (61 FR 4722) for listing, delisting, and reclassifying unique stocks under the ESA. To constitute a distinct population segment, a stock or group of stocks must be--(1) discrete (i.e. spatially separate) from other stocks of the taxon), (2) significant (e.g. ecologically unique for the taxon; extirpation would produce a significant gap in the taxon range; the only surviving native stock of the taxon; or there is substantial genetic divergence between the stock and other stocks of the taxon, and (3) the status of the stock must warrant protection under the ESA.

Sicklefin and sturgeon chub populations on the Missouri River basin may be effectively isolated by the Missouri River main stem dams. Sicklefin and sturgeon chub populations meet the criterion for discreetness. As a result, the second criterion, that of significance (e.g., genetically or ecologically unique) must be addressed.

Dieterman (2000) examined sicklefin chub collected throughout their range in the Missouri and Lower Yellowstone Rivers, measuring 18 phenotypic traits and using multi-variance spatial techniques to explore patterns of spatial variation that might suggest phenotypically distinct populations. Determan found that intra-segment variation in sicklefin chub populations phenotypic traits currently exceeds inter-segment variation. This research indicates that phenotypically distinct populations in the Missouri River do not exist, despite river regulation.

Similar studies to evaluate phenotypic traits of sturgeon chub have not been conducted. However, given the short time that the Missouri River main stem dams have been in place, the Service does believe that genetically or ecological populations of sturgeon chub have developed. Testing to evaluate genetic variation within sicklefin and sturgeon chub populations have not been conducted.

The Service found no morphological, physiological, or ecological data during this status review process indicating unique adaptations of individual stocks or assemblages of sicklefin or sturgeon chub within the range of these species. Chub populations could meet the discreetness criterion; however, there is no evidence supporting the second criterion, that genetically or ecologically significant stocks have developed. Therefore, single populations of sicklefin chub and sturgeon chub are recognized for the purposes of this updated status review.

VIII. LAND OWNERSHIP

Within the wide geographic range of the sturgeon chub and sicklefin chub, ownership and management of the rivers, tributaries, and adjacent uplands varies by State and waterway. Ownership of the uplands adjacent to the Missouri River and its tributaries is primarily private, but also includes a mixture of Federal, State, tribal, and municipal lands. Management of reservoir elevations and annual operations on the lower six Missouri River main stem reservoirs and dams is the responsibility of the Corps. Reclamation has similar responsibilities for the Canyon Ferry Reservoir, the uppermost reservoir on the main stem Missouri River in Montana, and many of the tributary dams, reservoirs, and low-head diversion dams. The Corps also has primacy for operation and maintenance of the Missouri River Bank Stabilization and Navigation Project and a number of tributary reservoirs, especially in the Kansas and Osage River basins.

IX. PREVIOUS FEDERAL ACTION

The sicklefin chub and sturgeon chub first received listing consideration when the two species were included in the September 18, 1985, Animal Candidate Review for Listing as Endangered or Threatened Species (50 FR 37958) as category 2 candidate species for listing. Category 2 status comprised taxa for which information indicated that a proposal to list as endangered or threatened is possibly appropriate, but for which conclusive data on biological vulnerability and threats are not currently available to support proposed rules. The Service initiated individual status reviews in 1992 and subsequently published status reports on each species in August 1993 (U.S. Fish and Wildlife Service 1993a, 1993b).

On July 11, 1994, the Service reclassified both the sicklefin chub and sturgeon chub as category 1 candidate species and announced this reclassification in a revised animal notice of review on November 15, 1994 (59 FR 58982). Category 1 status comprised taxa for which the Service had substantial information on biological vulnerability and threats to support proposals to list the taxa as endangered or threatened species. As of February 26, 1996, the Service no longer classifies candidate species by category. The chubs are now simply termed a candidate species, and each has a listing priority of 2.

On August 8, 1994, the Service received a petition from the Environmental Defense Fund, which was jointly signed by American Rivers, Mni Sose Intertribal Water Rights Coalition, National Audubon Society, and the Nebraska Audubon Council, to list both the sicklefin chub and sturgeon chub as endangered. The petitioners asserted that these species should be listed as endangered species because of their inability to adapt to human-induced alterations of the Missouri River. They indicated that sicklefin chub and sturgeon chub are physically adapted through evolution to live in turbid, swift-flowing rivers. Alterations described by the petitioners include impoundments, channelization, and removal of snags. The petitioners indicated that these alterations have detrimentally impacted the fishes' spawning and feeding habitat by changing the natural hydrograph and water temperatures and halting sediment movement, which reduced turbidity, and reducing the amount of organic matter transported by the Missouri River (Hesse 1994).

Following a review of the petition, its supporting documents and data, and other available information about the status, distribution, abundance, and threats to the sicklefin and sturgeon chub, the Service published a notice in the January 18, 1995, Federal Register of a 90-day petition finding (60 FR 3613). The Service found that substantial information existed to indicate that listing the sicklefin chub and sturgeon chub may be warranted.

Section 4(b)(3)(B) of the ESA, requires that, for any petition to revise the List of Endangered and Threatened Wildlife and Plants that contains substantial scientific and commercial information, the Service make a finding within 12 months of the date of receipt of the petition on whether the petitioned action is--(a) not warranted, (b) warranted, or (c) warranted but precluded from immediate proposal by other pending proposals of higher priority. Completion of the 12-month finding and a potential listing proposal was delayed by a Congressional moratorium in 1995 and 1996 on listing packages, Service backlog of listing actions and low funding priorities for the chubs in 1996 and 1997, and Service and State comments requesting that data from several comprehensive fish surveys (1995-1998) throughout the chubs' historical range be incorporated into the listing package. The Service prepared an initial draft 12-month finding in August 1995 and updated the draft finding in 1997 and 1999.

On April 6, 2000, the Montana Rivers Coalition Inc. filed a 60-day notice of intent to sue because of the Service's failure to complete a 12-month finding on a petition to list the sicklefin chub and the sturgeon chub, as required by Section 4 of the ESA. This action led to a final stipulated settlement agreement being signed and entered by the United States District Court, Missoula, Montana, on October 6, 2000. The settlement agreement stipulated that the Service shall submit for publication in the Federal Register a 12-month determination for the sicklefin chub and the sturgeon chub on or before April 12, 2001.

X. SUMMARY OF FACTORS AND THREATS AFFECTING THE SPECIES

Section 4(a)(1) of the ESA and regulations (50 CFR Part 424) promulgated to implement the listing provisions of the ESA set forth the procedures for adding species to the Federal "List of Endangered and Threatened Wildlife and Plants." A species may be determined to be an endangered or threatened species due to one or more of the five factors described in Section 4(a)(1). Each of the five factors will be addressed for sicklefin chub and then sturgeon chub. The information presented for sicklefin chub populations applies to the sturgeon chub as their range and habitat use overlap. The range of sturgeon chub extends further than sicklefin chub, including tributaries to the Yellowstone and Missouri Rivers. The discussion of the factors affecting sturgeon chub primarily focuses on additional factors unique to this species.

SICKLEFIN CHUB

A. The Present or Threatened Destruction, Modification, or Curtailment of the Species' Habitat or Range.

Missouri River Main Stem Dams: Destruction and alteration of big-river ecological functions and habitat once provided by the Missouri and Mississippi Rivers are believed to be the primary cause of declines in the habitat and range of the sicklefin chub. The physical and chemical elements of channel morphology, flow regime, water temperature, sediment transport, turbidity and nutrient inputs once functioned within the big-river ecosystem to provide habitat for sicklefin and other native species. Today on the main stem Missouri River, approximately 36 percent of riverine habitat within the sicklefin chub's historic range, has been transformed from river to lake by construction of six massive earthen dams by the Corps between 1937 and 1964 (U.S. Fish and Wildlife Service 1993c). Another 40 percent of the river downstream of dams has been channelized. An additional 24 percent of river habitat has been altered by changes in water temperature and flow caused by dam operations.

Missouri River aquatic habitat downstream of the six main stem dams has been and continues to be altered by reductions in sediment and organic matter transport/deposition, flow modification, hypolimnetic releases, and narrowing of the river through channel degradation. Those activities have adversely impacted the natural river dynamics by reducing the diversity of bottom contours and substrate, slowing accumulation of organic matter, reducing overbank flooding, changing seasonal flow patterns, severing flows to backwater areas, and reducing turbidity and water temperature (Hesse 1987). The Missouri River dams also are believed to have adversely affected sicklefin chub by fragmenting habitats and effectively isolating populations. The reaches below the main stem dams also have been affected by a proliferation of bank stabilization projects in the past 10 years. Cumulatively, these projects may adversely affect aquatic habitat by increasing river velocities and river bed degradation; and reducing sediment input into the system, sandbar formation, and shallow water habitat.

The pattern of flow velocity, volume, and timing of the pre-development rivers provided the essential life requirements of native large-river fish like the sicklefin and sturgeon chub, pallid sturgeon, and paddlefish. Hesse and Mestl (1993) found a significant relationship between the density of paddlefish larvae and two indices (timing and volume) of discharge from Fort Randall Dam. They concluded that when dam operations caused discharge to fluctuate widely during spring spawning, the density of drifting larvae was lower, and when annual runoff volume was highest, paddlefish larval density was highest. Hesse and Mestl (1987) also modeled these same two indices of discharge from Fort Randall Dam with an index of year-class strength. They demonstrated significant negative relationships between artificial flow fluctuations in the spring and poor year-class development for several native and introduced fish species; river carpsucker (*Carpiodes carpio*), shorthead redhorse (*Moxostoma macrolepidotum*), channel catfish (*Ictalurus punctatus*), flathead catfish (*Pylodictis olivaris*), sauger (*Stizostedion canadense*), smallmouth buffalo (*Ictiobus bubalus*), and bigmouth buffalo (*I. cyprinellus*). The sample size of sturgeon was too small to model in that study; however, a clear relationship existed between poor year-class development in most native species studied and the artificial hydrograph.

Modde and Schmulbach (1973) found that during periods of low dam releases, the secondary subsidiary channels, which normally feed into the river channel, become exposed to the atmosphere and thus cease to contribute littoral benthic organisms into the drift. Schmulbach (1974) states that use of sandbar habitats were second only to cattail marsh habitats as nursery grounds for immature fishes of many species.

Construction and the continuing operation of the main stem dams on the Missouri have significantly altered the Missouri River ecosystem and the habitat historically used by sicklefin and sturgeon chub. The degree of impact to chub populations varies, depending upon location with the system. In locations where the Missouri River is free-flowing and carries relatively high levels of turbidity, sicklefin and sturgeon chub comprise a substantial portion of the population collected in benthic trawl samples. In reaches of the river system that have been converted to reservoir habitat or short, free-flowing inter-reservoir reaches with relatively low turbidity levels, chub populations have been extirpated.

Missouri River Bank Stabilization and Navigation Project: Historically, the main channel of the Missouri River changed course, relocating over 610 m (2,000 ft) in some years. The river transported large amounts of sediment that created braided channels in the meandering river. The braided channel restricted navigation and periodically flooded bottomland farms and communities along the river.

Authorization of the Rivers and Harbors Act between 1912 and 1945 established a program to channelize the Missouri River from Sioux City, Iowa, to the mouth of the Missouri River near St. Louis, Missouri. This program, known as the Missouri River Bank Stabilization and Navigation Project (BSNP), created one stabilized channel from the numerous small channels. The project consists mainly of revetments along the outside bends and transverse dikes along the inside bends to force the river into a single active channel that is self-sustaining. Officially completed in 1981, the existing project extends from Sioux City, Iowa, to the mouth of the Missouri River (735 mi) and maintains a 2.7-m deep (9-ft deep) by 91-m wide (300-ft wide) channel. The Corps conducts annual maintenance surveys and activities to ensure the continued integrity and function of the existing BSNP structures.

Morris et al. (1968) found that channelization of the Missouri River reduced the surface area by approximately 67 percent. Funk and Robinson (1974) calculated that the length of the Missouri River between Rulo, Nebraska, and its mouth (~500 RM) had been reduced by 8 percent, and the water surface area had been reduced by 50 percent following channelization.

Since 1974, the Corps has implemented measures to modify the channel maintenance structures and improve fish and wildlife habitat. The Corps has restored some side-channel connections and increase habitat diversity in the channelized Lower Missouri River by notching dikes or otherwise modifying channel structures (Burke and Robinson 1979). The Corps estimates that approximately 2,600 notches have been constructed. Notching dikes or revetments can increase channel width and diversity, and create substantial shallow-water/sandbar complexes at certain river stages. After the 1993 flood, revetment repairs that allowed continued riverine connection to off-channel scour holes and chutes also have helped maintain habitat diversity and value, particularly for riverine fishes.

Channelization of the Missouri River to create a self-sustaining navigation channel has reduced habitat diversity and adversely affected fish and wildlife habitat. Sicklefin and sturgeon chub populations exist in low numbers from Gavins Point Dam to St. Joseph, Missouri. Research studies conducted in the Missouri River in Missouri from the 1940s to the 1990s indicate that the relative percentage of sicklefin and sturgeon chub in small fish collections has increased. Unfortunately, baseline data characterizing chub populations in the Missouri River prior to the authorization and initial construction of the BSNP do not exist.

In 1986, Congress authorized mitigation for fish and wildlife resources lost due to the construction, operation, and maintenance of the BSNP. Please see ongoing regulatory and conservation action for further discussion of the BSNP mitigation plan.

Mississippi River Channelization: Construction activities to create and maintain a navigation channel in the Middle Mississippi River have been ongoing since 1927. Approximately 111 miles of stone dikes, 169 miles of rock revetment, and 16 miles of bendway weirs have been constructed to narrow the channel for navigation. This work, which alters or removes shallow, sandbar habitat used by chubs, is about 66 percent complete and is scheduled to be finished in 2014. Most side channels and islands were cut off from the main channel by closing structures. Wing dikes have reduced average width from about 1,615 m (5,300 ft) in 1888 to about 975 m (3,200 ft) in 1968, for a total reduction of about 40 percent (Rasmussen 1979). Currently, about 14,569 hectares (ha) (36,000 acres (ac)) of unvegetated sandbar habitat remain (U.S. Army Corps of Engineers 1997). Future plans call for constricting the river top width to 457 m (1,500 ft) between the distal ends of the wing dikes.

Levee construction on the Lower Mississippi River, from the Ohio River to the Gulf of Mexico, has eliminated the river's major natural floodway and reduced the area of the floodplain connected to the river by more than 90 percent (Fremling et al. 1989). Fremling et al. (1989) also report that levee construction isolated many floodplain lakes and raised river banks. As a result of levee construction, 15 meander loops were severed between 1933 and 1942.

We find that sicklefin and sturgeon chub habitat has been reduced by efforts to constrict and control the Missouri and Mississippi Rivers with reservoirs, stabilized banks, jetties, dikes, levees and revetments. However, segments of the Missouri and Mississippi Rivers continue to support self-sustaining population of sicklefin chub, and future construction should not reduce the existing habitat to levels that would eliminate viable populations. Studies conducted in Montana, North Dakota, and Missouri using benthic trawls indicate that sicklefin and sturgeon chub comprise a significant portion of the fish population in segments of the Yellowstone and Missouri Rivers. Recent studies conducted by the Missouri Department of Conservation have documented viable populations of both sicklefin and sturgeon chub in the Middle Mississippi River and in the Wolf Island area of the Lower Mississippi River.

Water Depletions: Water depletion projects for municipal, agricultural, and energy related industrial uses within the Missouri River basin may impact sturgeon chub and sicklefin chub populations and habitats. Areas of concern are water depletion projects in the Upper Missouri River and Lower Yellowstone River basin in Montana and North Dakota, and the Platte River in Nebraska. Much of the flow of the Platte River has been depleted and other water development projects have been proposed or are under construction in the Colorado and Wyoming portions of the basin. The Lower Platte River has experienced substantial depletion of flows during high runoff periods over the past century (Williams 1978, Eschner et al. 1983).

The Lower Yellowstone River and Missouri River upstream and downstream of the confluence of these two rivers, collectively known as the Mon-Dak irrigation frontier, are subject to considerable water depletion projects for irrigation purposes. In 1997, Reclamation (1999) conducted an inventory of major water diversions on the Lower Yellowstone River between Billings, Montana, and the North Dakota State line. Reclamation documented six low-head dam irrigation diversion projects, four irrigation diversions using lateral dikes, seven pumped irrigation diversion projects, and a number of industrial and municipal water diversions in this reach of river. The Mon-Dak region currently supports over 171,000 ac under irrigation.

In March 1998, Montana Governor Marc Racicot established the Vision 2005 Task Force on Agriculture. The Task Force was developed to address the goal of doubling agriculture's economic value by the year 2005. The Task Force established the goal of increasing irrigation in eastern Montana by 500,000 ac. Expansion of irrigation to meet the task force goal will require an unquantified amount of water depletions from the Yellowstone and Missouri Rivers and their tributaries. Planning is currently ongoing for the West Crane Sprinkler Irrigation Project southwest of Sidney, Montana. The Richland County Conservation District is seeking authorization to divert 24,000 acre-feet from the Yellowstone River to irrigate 12,000 ac of existing dry land agriculture. As the overall amount, timing and locations of these potential depletions are currently unknown, we cannot evaluate the impact of these proposals on aquatic resources at this time.

Irrigation diversion dams and other types of river diversion structures in the upper basin have the greatest potential to directly take (kill or harm) sicklefin and sturgeon chubs. Six low-head dam diversion structures are located on the Lower Yellowstone River below Billings, Montana, and five structures are found on the Tongue River, a tributary to the Yellowstone River. Some of the structures are administered and operated by Reclamation, while others are privately owned and operated. Irrigation diversion structures may be located near shallow water habitats for chubs, typically withdraw large volumes of available river flow, and may set up currents that pull or attract fish out of the river and into the diversion canals. Fish become entrained into the diversion canals during the irrigation season, cannot escape, and either die in the irrigated fields or canal after the diversions are completed for the season.

Reclamation (Hiebert et al. 2000) evaluated fish entrainment rates at the intake diversion structure on the Lower Yellowstone River. Studies were conducted during 1996, 1997, and 1998 irrigation by netting 2 to 4 of the 11 unscreened conduits in the diversion structure. Estimates of total entrainment were calculated by extrapolating the monthly average entrainment ratios over the full irrigation season. Reclamation projected that approximately 289,000 ± 113,000 sturgeon chub were entrained during the 3-year study. The projected losses for 1996, 1997, and 1998 were 52,000 ± 39,000, 75,000 ± 18,000, and 163,000 ± 56,000 sturgeon chub.

The diversion dams are generally low-head dams, but effectively act as barriers to upstream migration of native fish that evolved in a low gradient river system. In the late summer, diversions into canals and water withdrawal from the Tongue River may contribute to chronic dewatering of the Tongue River and impacts to the Tongue and Yellowstone fish communities (Backes et al. 1997).

B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes.

No evidence exists that overutilization of the sicklefin chub is occurring for any purpose. Collection of this species occurs at low levels during scientific investigations and for educational purposes. Sicklefin chub are not pursued by fishermen. Though not selectively harvested as a bait species, accidental removal of individual sicklefin chub from the wild may occur during legal harvest of bait fish for personal use throughout most of the chubs' historical range. Accidental removal also could occur from illegal harvest in Tennessee, Kentucky, and Kansas where sicklefin chub are protected from take. They also could be collected accidentally by legal commercial harvest of bait fish in Montana, North Dakota, South Dakota, Iowa, Missouri, Illinois, Kentucky, Arkansas, Mississippi, and Louisiana. Regardless, regulated collection for scientific and educational purposes, or accidental take associated with personal or commercial harvest of bait fish has a negligible effect on sicklefin chub populations.

C. Disease or Predation.

No studies have been conducted to evaluate the impacts of disease on sicklefin chub. As a result, the significance of this threat is not known, but it is believed to be low. Fishery biologists who collected sicklefin chub during the past 60 years have presented no evidence to suggest that disease presents a threat to sicklefin chub.

The extent of predation on sicklefin chub, both historical and present, is poorly documented. The sturgeon chub and sicklefin chub evolved with a number of piscivorous riverine fishes, including sauger, pallid sturgeon, and channel catfish. Some predation by these species on the sturgeon and sicklefin chub undoubtedly occurred, but the extent is unknown. Since the construction of water resource development projects on the Missouri River and its tributaries, riverine habitat has been lost due to impoundments, and turbidity levels have been reduced significantly in the remaining riverine or stream reaches. These factors, combined with the introduction of other piscivorous fishes more suited to the modified habitat conditions and decreased turbidities, have altered the fish communities and may contribute to the vulnerability of the sturgeon and sicklefin chub to localized predation. The piscivorous walleye (*Stizostedium vitreum*), white bass (*Morone chrysops*), skipjack herring (*Alosa chrysochloris*), and northern pike (*Esox lucius*) either have been introduced to the river systems or have become much greater in abundance in response to changed instream conditions.

Gardner and Berg (1982) reported that sturgeon and sicklefin chub are preyed upon heavily by sauger in the Missouri River in Montana. Sicklefin and sturgeon chub combined were the second most common food item in saugers collected from August to November 1980 in a reach of the river above Fort Peck Reservoir. They were found in 21 percent of the fish collected for stomach analysis. Elser (et al. 1977) evaluated the stomach contents of 43 sauger and 13 burbot (*Lota lota*) collected in the Yellowstone River in 1975 and 1976. Sturgeon chub were found in the stomachs of the sauger (4.7 percent) and one burbot (7.7 percent). Other researchers (Pflieger and Grace 1987) speculated that predation likely has increased over historic levels due to habitat alterations, greater water clarity, and escape of sight feeding piscivores into formerly unoccupied stream habitats. Some local predation is likely to occur at the confluence of tributary streams occupied by chubs and main stem reservoirs occupied by predators.

Sicklefin and sturgeon chub populations evolved with piscivorous fish in the Missouri River Basin and the Mississippi River. The best commercial and biological information available indicates that predation by piscivorous fish is not a threat to the continued existence of the sicklefin and sturgeon chub in locations where turbidity levels and flow conditions are adequate to support their populations.

D. Inadequacy of Existing Regulatory Mechanisms.

The degree or lack of protective classification for the sturgeon chub and sicklefin chub varies widely throughout its range. Several national and State professional conservation societies and environmental departments within various State governments unofficially have classified the sturgeon chub and sicklefin chub as either threatened or endangered, species of special concern, rare, on a watch list, deemed in need of management, or transient. However, these designations do not provide any legal protection to either chub species. Only a few States provide a legal status or regulate protection of the sturgeon chub and sicklefin chub.

Unofficially, the sturgeon chub is classified in Wyoming as a State species of concern and is considered under the State's Mitigation Policy and for planning purposes (Robert Pistono, in litt. 1995; Bill Wichers, in litt. 1997). In Montana, the sturgeon chub and sicklefin chub are considered species of special concern (Hunter 1994). In North Dakota, the Dakota Chapter of the American Fisheries Society (1994) considers the sturgeon chub as threatened and the sicklefin chub as endangered. In Nebraska, both the sturgeon chub and sicklefin chub are considered species of special concern (Ross Lock, Nebraska Game and Parks Commission, pers. comm. 1995). The species have not been classified in Iowa (Daryl Howell, Iowa Department of Natural Resources, pers. comm. 1995). The sturgeon chub and sicklefin chub are considered rare in Missouri (Missouri Department of Conservation 1995).

In South Dakota, the sturgeon chub and sicklefin chub officially are classified as threatened (Eileen Dowd Stukel, South Dakota Department of Game, Fish, and Parks, in litt. 1997). However, the South Dakota Department of Game, Fish, and Parks believes the sicklefin chub may be extirpated from the State (Douglas Hofer, in litt. 1995). Both chubs receive legal protection in the State of Kansas where the sturgeon chub and sicklefin chub are classified officially as threatened and endangered, respectively. Take of either species is prohibited and provisions allow for habitat protection and designation of critical habitat (Kansas Department of Wildlife and Parks 1992). The sturgeon chub is listed as endangered in the State of Illinois. The State prohibits the take of the sturgeon chub and provides some habitat protection (Sue Lauzon, pers. comm. 1995, and in litt. 1997). Kentucky has restrictions on collections of both chubs (Wayne Davis, Kentucky Department of Fish and Wildlife Resources, pers. comm. 1995) and Tennessee prohibits the take or possession of either chub, or the knowing destruction of habitats (Bob Hatcher, Tennessee Wildlife Resource Commission, pers. comm. 1995). In Louisiana, neither species is protected on a "State List." The Louisiana Department of Wildlife and Fisheries considers sicklefin and sturgeon chub as transient species (John Roussel, Louisiana Department of Wildlife and Fisheries, in litt 2000). In light of the low numbers of sturgeon chub and sicklefin chub in these States, the effectiveness of the various regulations is difficult to assess.

Missouri River Bank Stabilization and Navigation Project - Fish and Wildlife Mitigation: Congress authorized mitigation for fish and wildlife resources lost due to the construction, operation, and maintenance of the BSNP, within the States of Missouri, Kansas, Iowa, and Nebraska, in Section 601(a) of the Water Resources Development Act of 1986 (WRDA 86, Public Law 99-662). The Corps supported that authorization with the April 24, 1984, Chief of Engineers' report, "Missouri River Fish and Wildlife Mitigation, Iowa, Nebraska, Kansas, and Missouri." That report, based on a May 1981 Feasibility Report and Environmental Impact Statement completed by the Missouri River Division, documented the estimated loss of 522,000 ac (211,410 ha) of aquatic and terrestrial habitat in and along the Missouri River between 1912 and 2003 attributable to the BSNP. Based on those losses, the 1984 report also described various measures to compensate for these losses and recommended a plan to restore, preserve, or develop 48,100 ac (19,480 ha) of land (U.S. Army Corps of Engineers 1984). Project construction is to include land acquisition and habitat development on 29,900 ac (12,109 ha) of land and habitat development on 18,200 ac (7,371 ha) of existing public lands within the four affected States. Although several mitigation alternatives were proposed, the selected alternative, when fully implemented, would compensate only an estimated 3 percent of lost aquatic acres and 7 percent of lost terrestrial acres attributable to the BSNP. Like the BSNP, the Mitigation Project is completely federally funded (i.e., construction, operation, and maintenance).

If fully implemented, the Mitigation Project will preserve and restore 3,200 ac (1,296 ha) of aquatic habitat, and 44,900 ac (18,184 ha) of terrestrial habitat through development of habitat on public lands and acquisition and development on private lands. Funding began in Fiscal Year 1992. As of April 2000, approximately 79 percent of the originally authorized land acquisition acreage has been acquired (23,549 ac out of 29,900 ac originally authorized). Land acquisition is complete in Kansas and Nebraska, and is likely to be completed in Iowa and Missouri in the next couple of years. Of these acquired lands, approximately 18 percent (4,295 ac [1,739 ha]) have been developed for fish and wildlife. Habitat development of public lands as of April 2000, is 2,504 ac (1,014 ha) of the 18,200 ac (7,371 ha) authorized, or about 14 percent.

Conceptual aquatic habitat objectives for mitigation sites call for reclaiming and reconnecting filled-in chutes and backwaters, and preventing future sedimentation. Terrestrial habitat development will depend on the existing habitats types, and for public land, existing management objectives. Habitat development may involve dredging of filled-in wetlands, enlarging wetlands, side channel openings/closure, bank stabilization, dike and levee construction, pumping, reforestation, timber stand improvement, food plot establishment and native re-vegetation. Restoration of floodplain habitats such as mature bottomland forests will take many years before significant habitat benefits will begin to accrue to the Missouri River ecosystem. Restoration of other habitats like emergent wetlands, shallow water areas, and chutes should result in more immediate benefits to the river ecosystem.

On most existing public lands, terrestrial habitats are likely to remain isolated from the river by levees. On acquired lands, the value of the Mitigation Project to the riverine environment will depend on its potential for restoring main channel and off-channel habitat, and reconnecting floodplain habitats to the river during the spring flood pulse. Areas with extensive levee protection and no connected aquatic and wetland habitats such as chutes, sloughs, side channels, or temporary and seasonal wetlands will have less value to the riverine/floodplain ecosystem.

Recently, Section 334 of the WRDA of 1999 reauthorizes the Missouri River Bank Stabilization and Navigation Fish and Wildlife Mitigation Project and increases the amount of lands, and interests in land, to be acquired for the project by 118,650 ac (48,053 ha). To determine the cost of this project modification, Section 334 (b)(1) also directs the Corps to conduct a study within 180 days in conjunction with the States of Nebraska, Iowa, Kansas, and Missouri. That report was completed in April 2000. The Corps is awaiting congressional action to implement the expanded mitigation project.

Based on conceptual plans for restoration projects in the four States, the agencies anticipate the expanded project could potentially provide approximately 7,000 ac (2,835 ha) of shallow water, sandbar habitat (under existing hydrologic conditions) which will benefit native fish populations including sicklefin and sturgeon chub. Monitoring programs to evaluate fish populations are needed to quantify benefits of project-related shallow water, sandbar habitat to the listed and candidate endangered species. The expanded mitigation project also is expected to provide approximately 20,000 ac (8,100 ha) of additional wetland habitat and 92,000 ac (37,260 ha) of additional terrestrial habitat in the Missouri River floodplain.

Preliminary monitoring data for selected mitigation and control sites in Nebraska and Missouri are currently available. The Nebraska Game and Park Commission has initiated a monitoring program at chute restoration, backwater, and dike modification/removal mitigation sites and at control sites in the Missouri River. In the spring of 1999, three sturgeon chub were collected with seines at the Hamburg Bend mitigation site. Benthic trawl samples were taken at the Tobacco Island mitigation site and the Goose Island control site during the fall of 1999. Five sturgeon chubs were collected from the Tobacco Island area, representing 23 percent of all sturgeon chub collected in the Nebraska reach of the Missouri River since 1941. One sicklefin chub was taken at Goose Island. This was the first sicklefin chub collected in the Nebraska reach of the Missouri River since 1988 (Nebraska Game and Park Commission 2000). Nebraska Game and Park Commission will continue its monitoring efforts in the coming year to evaluate the Missouri River BSNP - Fish and Wildlife Mitigation Project.

In Missouri, the Service sampled a 7-mile reach of the Lower Missouri River around the Jameson Island (River Mile 219) and Lisbon Chute (River Mile 217) mitigation areas using a benthic trawl, mini-fyke nets, and seine. During the period from 1997 to 1999, 480 sicklefin chub and 13 sturgeon chub were collected (Louise Mauldin, U.S. Fish and Wildlife Service in litt. 2000). Work conducted by the University of Missouri between 1994 and 1997 documented the use of scour holes by sicklefin and sturgeon chub and their high value as nursery habitat for larval, juvenile and young-of-year fish (John Kubisiak, Missouri Cooperative Fish and Wildlife Research Unit, in litt. 1997; John Tibbs, Missouri Cooperative Fish and Wildlife Research Unit, in litt. 1997; Doug Dieterman, Missouri Cooperative Fish and Wildlife Research Unit, in litt. 1999).

E. Other Natural or Manmade Factors.

Hybridization: Fishery biologists have noted the presence of a small percentage of chub hybrids in collections from the Missouri River in Missouri. Grace and Pflieger (1985) collected one speckled chub x sturgeon chub and one sturgeon chub x sicklefin chub in a sample of 18,400 fish collected near Easley, Missouri (river mile 177.3 to 169.9) in 1982 and 1983. Gelwicks et al. (1996) reported 18 speckled chub x sturgeon chub hybrids in collections made in 1994. The hybrids were found at 7 of 13 collection sites in the Missouri River, from the Iowa-Missouri border in the confluence of the Missouri and Mississippi Rivers near St. Louis. Hybridization has not been reported at other locations within the range of the sicklefin and sturgeon chub. While the extent of hybridization and potential impacts to sicklefin and sturgeon chub populations appears to be minor at this time, future studies should monitor and report on the presence of hybrids.

Pollution/Contaminants: Although it does not appear that pollution has directly contributed to reduction of the species range, pollution may be an exacerbating threat. Pollution of the Missouri River by organic wastes from towns, packing houses, and stockyards was evident by the early 1900s and continued to increase as populations grew and additional industries were established along the river (Whitley and Campbell 1974). Due to the presence of a variety of pollutants, numerous fish-harvest and consumption advisories have been issued over the last decade or two from Kansas City, Missouri, to the mouth of the Mississippi River.

Riverine habitats important to sicklefin and sturgeon chub are subject to acute and chronic water quality impacts and contamination associated with oil development and transport of crude oil products. In the past 10 years, oil pipelines crossing the lower reaches of the Chariton River and Gasconade River, two major tributaries of the Missouri River in Missouri, have ruptured and spilled large amounts of crude oil, which eventually reached the main stem. In June 1995, an oil spill that occurred into an irrigation canal near the confluence reached the Missouri River. Barge accidents have occurred on the Mississippi River and resulted in the release of contaminants. Because of the volume of flow and dilution factor in the Yellowstone, Missouri, and Mississippi Rivers, the potential for oil spills and release of other contaminants to impact large areas of sturgeon and sicklefin chub habitat is considered low, and direct impacts to chubs minor. State and Federal agencies have programs in place to address spills of oil and other contaminants. These programs minimize any impacts that a spill might have on habitat for fish and wildlife. For example, the State of Missouri's Department of Natural Resources and the Environmental Protection Agency oversee cleanup activities related to oil spills. We also work with our partners to minimize impacts of spills on fish and wildlife.

Invasive Species: Impacts to native fish and wildlife populations is ever increasing due to the introduction of non-native species that have the capacity to cause irreparable damage. A major contributor to the depletion and extinction of native species, second only to habitat loss, is the introduction of species into new environments. The threats to native populations include--displacement of native species through competition for habitat or forage, decreasing the amount of biological diversity necessary to maintain a viable native population, impacting water quality, reducing habitat quality for native populations and influencing the biomass of fragile populations.

In the Missouri River below Gavins Point Dam, these effects are currently happening at an ever increasing rate. Exotic fish species have entered the Missouri River via the Mississippi River and are expanding upstream into suitable habitats. A number of Asian carp species are currently established in this section of the river. These include the bighead, black, silvery, and grass carp. Changes to Missouri River fish populations seem to be occurring already. Anecdotal information from commercial anglers indicates that Asian carp have become a dominant by-catch and are actually replacing desired species. Currently, no data exist to document that chubs are being impacted directly by invasive species. However, if Asian carp populations continue to expand, the diversity of species supported by the Missouri and Mississippi River ecosystems, including chubs, will likely be negatively impacted.

In the Missouri River basin above Gavins Point Dam, a potential also exists for undesirable exotic species to impact fish populations, although this threat has not occurred to date. Eventually, some exotic species likely will be introduced. Introduction of exotic invasive species has the potential to impact native populations, such as the two chub species, through direct and indirect competition with aquatic plants and animals, thereby decreasing available habitat.

Impingement: The Service (1993a, 1993b), Stasiak (1990), and Hesse et al. (1982) reported the impingement of very low numbers of both sicklefin chub and sturgeon chub at once-through-cooled power plant intakes on the Missouri River in Nebraska. Between 1973 and 1977, one sicklefin chub and two sturgeon chub were impinged at the Fort Calhoun Nuclear Station (river mile 646). Impingement sampling frequency at Fort Calhoun was twice daily from May through September, and once daily from October through April. At Cooper Nuclear Station (river mile 556) only one sturgeon chub was reported impinged between 1974 and 1977, with five randomly selected diurnal and nocturnal sampling times per week. Only two sicklefin chub and one sturgeon chub were impinged at the

Iatan Power Plant intake (river mile 411), another once-through-cooled plant, in northwest Missouri during 12, 24-hour surveys between October 5 and December 31, 1980 (Geo-Marine, Inc. 1981). The water withdrawal rate during the Iatan study was about 550 cubic-feet-per-second, or roughly 1 to 2 percent of the river flow. A 4-year study of power plant impingement, entrainment, and water temperature effects to Middle Missouri River adult fish communities did not detect changes in the adult fish populations because of power plant operations. Overall impacts to the river's aquatic communities were considered minimal (Hesse et al. 1982).

Impingement and entrainment impacts to chub populations from municipal and industrial water intakes throughout the species' ranges are unknown due to lack of data. Compared to once-through-cooled power plants, these intakes withdraw insignificant amounts of water in comparison to river flow, especially those along the main stem Missouri and Mississippi Rivers. However, power plant, municipal, industrial, and irrigation intakes sited in river segments with less altered habitats, such as occur in the Upper Missouri and Yellowstone Rivers in Montana and upper basin tributaries, have greater potential to impinge and/or entrain chubs.

Quantifiable power plant, municipal, and industrial intake threats to chubs along the Lower Missouri River in Missouri and the Middle Mississippi and Lower Mississippi Rivers are unknown due to the lack of data. The larger populations of both species in the Missouri River below Kansas City increases the probability that intakes, especially those at once-through-cooled power plants that remove higher percentages of the river flow, accidentally will take individuals of both species.

XI. SUMMARY OF FACTORS AND THREATS AFFECTING STURGEON CHUB

A. The Present or Threatened Destruction, Modification, or Curtailment of the Species' Habitat or Range. Issues for the sturgeon chub are the same as previously described for the sicklefin chub. Please see sicklefin chub summary, page 51.

B. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes.

No evidence exists that overutilization of the sturgeon chub is occurring for any purpose. Collection of this species occurs during scientific investigations and for educational purposes. They are not pursued by fishermen. Though not selectively harvested as a bait species, accidental removal of individual sturgeon chub from the wild may occur during legal harvest of bait fish for personal use throughout most of the chubs' historical range. Accidental removal also could occur from illegal harvest in Tennessee, Kentucky, Kansas, and Illinois, where sturgeon chub are protected from take. They also could be collected accidentally by legal commercial harvest of bait fish in Montana, North Dakota, South Dakota, Nebraska, Iowa, Missouri, Kentucky, Arkansas, Mississippi, and Louisiana. Regardless, regulated collection for scientific and educational purposes, or accidental take associated with personal or commercial harvest of bait fish has a negligible effect on sturgeon chub populations.

C. **Disease or Predation. Issues for the sturgeon chub are the same as previously described for the sicklefin chub.** Please see sicklefin chub summary, page 55.

D. **Inadequacy of Existing Regulatory Mechanisms. Issues for the sturgeon chub are the same as previously described for the sicklefin chub.** Please see sicklefin chub summary Page 55.

E. **Other Natural or Manmade Mechanisms.**

Drought: Severe drought, combined with the construction of the Missouri River main stem dams and reservoirs, may be a factor influencing sturgeon chub populations in some tributaries. Historically, the impacts of severe drought on sturgeon chub populations in tributary streams was likely mitigated by refugia habitat offered by the Missouri River. Prior to impoundment, the Missouri River provided habitat to sustain populations during severe drought and a source of chubs to recolonize tributaries following drought. Today, approximately 750 miles of refugia habitat, from the headwaters of Lake Sakakawea to Gavins Point Dam, has been degraded or converted to reservoir habitat and no longer provides suitable habitat to sustain sturgeon chub populations.

For example, severe drought in the Missouri River basin in the late 1980s and early 1990s may have contributed to the extirpation of sturgeon chub from the Little Missouri River in North Dakota and South Dakota. Kelsch (1994) sampled a number of sites in the Little Missouri River where Reigh and Elsen (1979) had collected sturgeon chub in 1976 and 1977; however, Kelsch did not collect sturgeon chub. He hypothesized that the closure of the Garrison Dam in 1953 and the subsequent formation of Lake Sakakawea eliminated the Missouri River as refugia habitat for sturgeon chub during periods of severe, prolonged drought. During prolonged drought, which occurred in the region of the Little Missouri River between 1987 and 1993, sturgeon chub may have been unable to persist in the Little Missouri River during periods of intermittent flow, clearing water, and silty substrate conditions that periodically occurred. During historic periods of severe drought, the unimpounded Missouri River may have served as a refuge, continuously providing turbid, flowing conditions necessary for sturgeon chub survival.

Due to fragmentation of habitats, tributaries like the Little Missouri River that flow into reservoirs are now isolated from riverine habitat with sturgeon chub, therefore preventing natural recolonization from downstream riverine reaches. These isolated streams may need to be augmented and stocked with chubs captured from other stable populations (Dryer et al. 1997). However, the introduced populations, if successful, will be subject to the same impacts during the next protracted drought.

Sturgeon chub populations evolved with periods of extended drought as a natural factor influencing their habitat. While construction of the dams on the Missouri River coupled with drought may have resulted in the loss of sturgeon chub populations in some tributaries, we do not believe this is a significant factor affecting existing populations.

Coalbed Methane Production: Coalbed methane development in northeastern Wyoming and southeastern Montana poses a potential threat to sturgeon chub populations and habitat in the Powder and Tongue River basins (David Felley, U.S. Fish and Wildlife Service, Cheyenne, Wyoming, pers. comm. 2000, and Lou Hanebury, Fish and Wildlife Service, Billings, Montana, pers. comm. 2000). Methane locked in coal beds is extracted by using modified water well drilling rigs to establish wells and then pumping water out of the formation to reduce the hydrostatic pressure. Wells typically produce mostly water at first (12-15 gallons per minute), but over time the amount of water declines and gas production increased as the bed is dewatered. The water is either discharged on the surface or injected into underground aquifiers. At this point, studies have not been conducted to determine infiltration and evaporation losses or the amount of discharge water reaching the Powder River.

Coalbed methane production in Wyoming is a rapidly expanding industry. In the Powder River basin, approximately 3,000 active wells exist and over 11,000 additional wells have been permitted for drilling. Industry estimates indicate that up to 75,000 wells are possible over the next 60 years in the Powder River Basin. In Montana, coalbed methane production has occurred to a limited degree, with approximately 170 wells currently producing. The Montana Department of Environmental Quality has placed a moratorium on the permitting of new wells until an environmental impact study is complete.

Coalbed-methane-produced water that meets Wyoming water quality standards is typically discharged into intermittent drainages and surface waters. The Wyoming Department of Environmental Quality initially thought the existing standards concerning water produced during oil and gas production were adequate; however, given the magnitude of rapid expansion of coalbed methane production, this position is being reviewed (Wyoming State Engineer's Office, State Water Forum Meeting, October 27, 2000).

Potential water quality issues associated with the production of coalbed methane have been identified by the Service's Cheyenne, Wyoming, Field Office. Groundwater samples taken from Powder River basin coal seams have trace element concentrations exceeding the aquatic chronic criteria for arsenic, cadmium, copper, iron, lead, selenium, and zinc. Coalbed-methane-produced water in the Powder River drainage is generally higher in sodium and total dissolved solids and has a higher sodium adsorption ratio than water produced at methane wells in the Belle Fourche drainage. The Service is collecting water discharged at producing wells to evaluate trace element concentrations and assess potential impacts to fish and wildlife resources.

Coalbed methane production has the potential to impact sturgeon chub populations in the

Powder and Tongue Rivers drainages; however, at this time sufficient information is not available to determine the significance of this threat. Field studies evaluating discharges throughout the year are needed to document water quantity and quality effects and their significance to fish and wildlife resources.

Yellowstone River Basin Low Head Dams: The lowhead dams on the Yellowstone River and its tributaries have been identified as being barriers to native fish species which migrate for spawning purposes. Normally, these lowhead dams also are associated with a water withdrawal system for irrigation or municipal water uses which subsequently entrain fish species, thereby affecting recruitment and survival to the populations.

Reclamation's lowhead dam located at Intake, Montana, has been identified as a structure to be privatized and transferred to the Lower Yellowstone Irrigation District. As part of this transfer, the Service and Reclamation entered into a formal consultation under the ESA. Effects of this structure were identified for the pallid sturgeon, as well as for the sturgeon chub. Reclamation decided to incorporate "fish friendly" changes into the structure as part of the transfer process. Due to uncertainties in the actual design changes needed to be effective for pallid sturgeon, Reclamation has decided to identify that changes are needed and convene experts in fish passage and sturgeon to develop an alternative that will best suit this situation. Reduction of entrainment losses at Intake is expected to be very beneficial to sturgeon chubs.

Several other structures on the Yellowstone and its tributaries above Intake may offer opportunities in the future to make improvements for fish passage and reduce entrainment losses. Improvements at any of these sites could benefit the chubs.

XII. ONGOING REGULATORY AND CONSERVATION ACTIONS

Missouri River Biological Opinion: In November of 2000, the Service completed a biological opinion (U.S. Fish and Wildlife Service, 2000) under Section 7 of the ESA on the Corps' Operation of the Missouri River Main Stem System, the related operation of the Kansas River Tributary Reservoirs, and the Operations and Maintenance of the Missouri River Bank Stabilization and Navigation Project. The habitat covered in this consultation includes the Missouri River from the headwaters of Fort Peck Reservoir to its confluence with the Mississippi River at St. Louis. Among the four species covered in this consultation is the pallid sturgeon. The pallid sturgeon inhabits large, turbid rivers and in much of its range, the pallid sturgeon uses habitats similar to the sicklefin and sturgeon chub.

The Service found that to avoid jeopardizing the continued existence of the pallid sturgeon, restoration of a portion of suitable riverine and aquatic habitat, and hydrologic conditions on river segments between Fort Peck and the headwaters of Lake Sakakawea, and the river below Gavins Point Dam to its confluence with the Mississippi is necessary. As part of the consultation, the Service developed a Reasonable and Prudent Alternative (RPA) that includes actions for the pallid sturgeon and the ecosystem in general. The alternative is designed to return some semblance of practical "form and function" of a river system to appropriate sections of the Missouri and Kansas Rivers. This alternative, in part, includes flow enhancement, temperature modifications, and habitat restoration/creation in key sections of the river.

An integral part of this process is adopting an adaptive management approach. Adaptive management allows regular modification of management actions based on new information from the endangered species and habitat monitoring program and changing environmental conditions. An agency coordination team will guide development and implementation of future river management and habitat development activities. The Corps is currently working on an Implementation Plan for the RPA. When all or parts of this RPA are implemented, the Service expects a beneficial impact to the ecology of the river and, in particular, its indigenous species, including sicklefin and sturgeon chub populations.

The current emphasis on the Missouri River is to restore enough environmental integrity to the river to avoid jeopardizing its species. This emphasis, if implemented, is expected to have a significant beneficial effect on both the chubs through habitat restorations/creations, improved temperature regimes, and beneficial and stimulating flow modifications in sections of river above Lake Sakakawea and below Gavins Point Dam.

Little Missouri River Sturgeon Chub Reintroduction: In 1997, the Service developed a reintroduction plan to re-establish extirpated populations of the sturgeon chub in the Little Missouri River (Dryer et al. 1997). Through a partnership with the North Dakota Game and Fish Department, the Montana Department of Fish, Wildlife, and Parks, National Park Service, U.S. Forest Service, and Reclamation, sturgeon chub have been captured at or near the Intake Diversion Structure on the Lower Yellowstone River in Montana during the past three field seasons. This work resulted in releasing 302, 473, and 201 sturgeon chub into the Little Missouri River at the South Unit of Theodore Roosevelt National Park in 1998, 1999, and 2000, respectively (U.S. Fish and Wildlife Service 1998 and Wade King, Fish and Wildlife Service, pers. comm. 2000). Test netting conducted in the Little Missouri River after reintroduction has yielded no sturgeon chub. The Service is currently evaluating this initiative. Sturgeon chub collected at the Intake Diversion Structure also have been provided to Gavins Point National Fish Hatchery and the Bozeman Fish Technology Center to develop propagation techniques.

XIII. FINDINGS AND CONCLUSIONS

The Service has compiled and analyzed the available data on sicklefin and sturgeon chub populations throughout their range. We found that sicklefin and sturgeon chub are highly adapted for conditions found in turbid, free-flowing river systems. The historic range of the sicklefin chub included the Lower Yellowstone River, the Missouri River, and the Mississippi River below the confluence with the Missouri River. The range of the sturgeon chub overlapped the sicklefin chub and included 30 tributaries to the Yellowstone and Missouri Rivers. Sturgeon chub also ascended further upstream in the Yellowstone and Missouri Rivers than sicklefin chubs. We also found the literature documenting sicklefin and sturgeon chub provide an incomplete picture of population levels, range, habitat use, and biology. Information documenting chub baseline conditions (prior to the construction on the Missouri River main stem dams) is limited to a few records documenting the presence of these species.

In 1993, the Service issued status reports for the sicklefin chub and sturgeon chub. The reports indicated the range and populations of sicklefin and sturgeon chub have been substantially reduced. In August 1994, the Service was petitioned to list the sicklefin and sturgeon chub as endangered. These actions helped to focus attention on two species that had been largely overlooked throughout much of their range. While major information gaps remain concerning feeding habits, reproduction, seasonal habitat use, and other aspects of sicklefin and sturgeon chub biology, substantially greater emphasis has been placed on documenting chub populations and their habitats during the past 7 years.

At the same time as the petition to list the sicklefin and sturgeon chub as endangered was filed, fishery biologists modified the gear used to sample cyprinid populations. Until 1993, researchers primarily relied on seines to collect small fish in the Missouri and Mississippi Rivers. Seines allowed sampling in shallow water sandbar and border channel habitats, usually not exceeding 1.5 m (4.9 ft) in depth. Grisak (1996) was the first to use a benthic trawl, modified to catch small fish, to characterize the fish population in the Missouri River. Grisak's work above Fort Peck Reservoir in Montana during 1994 and 1995 and the results of subsequent field investigations using benthic trawls have provided new information on the range and relative abundance of the sicklefin and sturgeon chub. He collected 5,095 fish, using seines to sample shallow-water sites (0.19 to 0.86 m - 0.6 to 2.8 ft). Sicklefin and sturgeon chub were rare in seine hauls, comprising 0.08 and 0.16 percent of the total catch, respectively. Sturgeon chub ranked 14th in abundance and sicklefin chub ranked 15th in seine hauls. In comparison, Grisak collected 302 sicklefin chub (21.9 percent of the catch) and 260 sturgeon chub (18.9 percent of the catch) using a benthic trawl. Sicklefin and sturgeon chub were the second and third most common species collected in benthic trawl tows. The mean depth at trawl sites where sicklefin chubs were collected was 3.41 meters.

Field studies, conducted since the 1993 status reports were issued, indicate that sicklefin chub and sturgeon chub are more widespread and occur in greater numbers than previously believed. Researchers in Montana (Gardner 2000a,b), North Dakota (Liebelt, in litt. 1999, Everett 1999, Welker 2000), and Missouri (Grady and Milligan 1998, Hrabik and Herzog, in litt. 2000a,b) have collected substantially greater numbers of sicklefin and sturgeon chub using trawling techniques. Recently, new locations supporting sicklefin and sturgeon chub populations, such as the Wolf Island area of the Lower Mississippi River also have been identified.

While recent studies documenting sicklefin and sturgeon chub populations are encouraging, the range of these species has been substantially reduced. The major factors impacting sicklefin and sturgeon chub populations are the construction and continued operation of the six main stem dams on the Missouri River built as part of the Pick Sloan Plan between 1937 and 1964, the loss of habitat associated with the Bank Stabilization and Navigation Project in the Middle and Lower

Missouri River, and navigation projects on the Middle and Lower Mississippi River. The dams altered the physical and chemical elements of channel morphology, flow regime, water temperature, sediment transport, turbidity, and nutrient input that provided habitat for sicklefin chub, sturgeon chub, and other native fish species. Today, approximately 36 percent of the Missouri River's riverine habitat has been converted to reservoirs, 40 percent has been channelized, and the remaining 24 percent has been altered by changes in water temperature, turbidity levels, and flow conditions caused by dam operations. Sicklefin chub currently occupy approximately 54 percent of its historic range in the Missouri River basin. Sturgeon chub are currently found in about 55 percent of its former range in the Missouri River. Sturgeon chub also occur in 11 of the 30 tributaries to the Yellowstone and Missouri Rivers where historic catch records exist.

Sicklefin and sturgeon chub populations have been eliminated from over 800 miles of the Missouri River that has been impounded, and approximately 200 miles of inter-reservoir reaches between Garrison Dam in North Dakota and Gavins Point Dam in South Dakota. These species also are found in low numbers in the Middle Missouri River, below Gavins Point Dam to St. Joseph, Missouri. Collectively, the results of field investigations indicate viable, self-sustaining populations of sicklefin and sturgeon chub continue to occur in a portion of their historic range, while in other areas these species have been extirpated or exist in low numbers.

The ESA defines a threatened species as any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. An endangered species is defined as any species which is in danger of extinction throughout all or a significant portion of its range. A species may be determined to be threatened or endangered due to one or more of five factors described in Section 4(a)(1) of the ESA. The decision concerning whether a species warrants listing requires an evaluation of past actions and measures in the foreseeable future that affect the species.

As discussed previously, the principal factor impacting sicklefin and sturgeon chub populations is the construction and operation of the dams on the main stem Missouri River, operation and maintenance of the Bank Stabilization and Navigation Project in the Missouri River, and the navigation channel on the Mississippi River. Water depletion projects, impoundments, entrainment, and drought impacted sturgeon chub populations in the Yellowstone River and tributaries to the Yellowstone and Missouri Rivers. The threats posed by the dams and reservoirs have been in place for over 35 years. Despite the loss of over 1,000 miles of suitable habitat in the Missouri River, sicklefin and sturgeon chubs continue to be found in good numbers where habitat conditions, flow patterns, and turbidity levels resemble conditions prior to the construction of the main stem dams. Likewise, the wide-spread extant chub populations provide evidence that these species retain viable populations in spite of impacts of water depletions, entrainment and drought.

Sicklefin and sturgeon chub are short-lived species, with a small percentage of their populations reaching age 4+. While little is known about sicklefin and sturgeon chub reproduction, these species have successfully propagated with the major identified threats in place since 1964, when the Big Bend Dam in South Dakota, the last major flood control component of the Pick-Sloan Plan, was completed. Sicklefin and sturgeon chub have successfully reproduced under a variety of flow conditions in the Missouri River, including periods of extended drought and persistent high water levels.

There are potential impacts associated with coalbed methane production in Wyoming and Montana, future water impoundment and depletion projects on the Yellowstone River, its tributaries, and tributaries to the Missouri River, and Asian carp population in the Lower Missouri and Mississippi Rivers. However, based on the information currently available, we do not believe that magnitude of these threats are sufficient to endanger the population viability of these species throughout their range.

On the basis of the available information, the Service concludes that neither the sicklefin chub nor the sturgeon chub are likely to become threatened or endangered in the foreseeable future. Stable, self-sustaining populations of sicklefin and sturgeon chub exist in widely scattered areas of their range. Chub populations continue to successfully reproduce with principal factors impacting chub habitat, the Missouri River main stem dams having been in place for over 35 years. Therefore, listing the species is not currently warranted. This conclusion is based on the best available information summarized in this document.

Our not warranted finding is based on the current status of these species, and upon our analysis of such future threats that are known at this time. The Service is encouraged that ongoing and planned conservation measures will benefit sicklefin and sturgeon chubs and their habitats in the foreseeable future. As discussed previously, these projects include the Missouri River BSNP Fish and Wildlife Mitigation Plan, conservation measures identified in the Missouri River Biological Opinion, and section 7 consultation to minimize fish entrainment at the intake diversion structure in the Yellowstone River. The Service believes that the identified conservation measures provide priority actions to improve habitat conditions for sicklefin and sturgeon chub.

The Service recommends that Federal and State natural resource agencies, tribal groups, universities, conservation organizations, and other concerned entities continue to monitor, protect, and restore sicklefin and sturgeon chub populations throughout their range. Information in the literature describing the feeding habits, reproduction, seasonal habitat use, predator prey relations, and other aspects of sicklefin and sturgeon chub biology is limited. To protect and enhance sicklefin and sturgeon chub populations and their habitat, additional research and monitoring is needed to guide habitat rehabilitation plans and reintroduction efforts.

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U.S. FISH & WILDLIFE SERVICE
REGION 6

CONTAMINANTS PROGRAM

Selenium in a Wyoming Grassland Community Receiving Wastewater from an In Situ Uranium Mine

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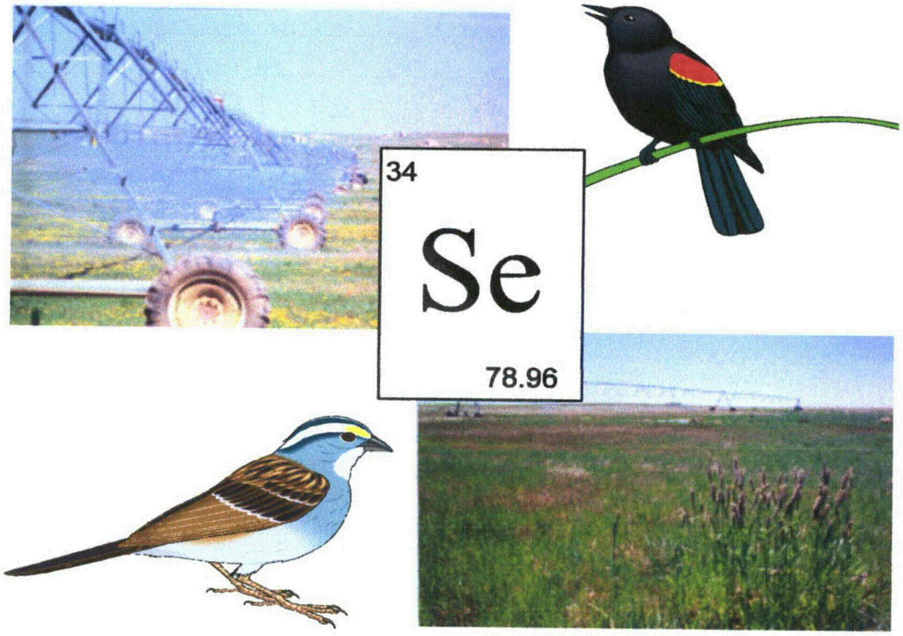
Project #: 98-6-6F37-FC



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ABSTRACT

Water, soil, vegetation, grasshoppers (Family Acrididae), bird eggs and bird livers collected at a 23.5 hectare (58 acres) grassland irrigated with wastewater from an in situ uranium mine (Study Area) and a reference site in 1998 were analyzed for selenium and other trace elements. Bird surveys were conducted at the irrigated grassland at the in situ uranium mine to determine species use, relative abundance and behavior. We observed 23 species of birds using the Study Area. Western meadowlarks (*Sturnella neglecta*), red-winged blackbirds (*Agelaius phoeniceus*), lark buntings (*Calamospiza melanocorys*) and horned larks (*Eremophila alpestris*) were the most common avian species using the Study Area and were observed feeding and drinking at this site. Meadowlarks, red-winged blackbirds and lark buntings were observed nesting at the Study Area. Selenium concentrations in the uranium mine wastewater applied onto the grassland ranged from 340 to 450 µg/L. Selenium in the upper 15 cm (6 in) of soil from the irrigated grassland at the mine ranged from 2.6 to 4.2 µg/g dry weight (dw). Mean selenium concentrations in soil and water were 5 and 15 times higher at the Study Area than at the reference site. Selenium concentrations in grasses and grasshoppers ranged from 6.8 to 24 µg/g and 11 to 20 µg/g dw, respectively. Selenium in red-winged blackbird eggs and livers collected from the Study Area ranged from 13.2 to 22 µg/g and 33 to 53 µg/g dw, respectively, and concentrations were well in excess of toxic thresholds. Mean selenium concentrations in grasses, grasshoppers, and bird eggs and livers were 5.8 to 30 times higher at the Study Area than at the reference site. Elevated selenium concentrations in water, soil, grasshoppers, and red-winged blackbird eggs and livers collected from the Study Area demonstrate that selenium is being mobilized and is bioaccumulating in the food chain.

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INTRODUCTION

High concentrations of waterborne selenium can be produced with in situ mining of uranium ore as uranium-bearing formations are usually associated with seleniferous strata (Boon 1989). Boon (1989) reported that uranium deposits in Converse County, Wyoming can contain up to 4,500 µg/g (ppm) of selenium. In situ mining of uranium is done by injecting a leaching solution of native ground water containing dissolved oxygen and carbon dioxide into the uranium-bearing formation through injection wells. The leaching solution oxidizes the uranium and allows it to dissolve in the ground water. Production wells intercept the pregnant leaching solution and pump it to the surface. The leaching solution also dissolves selenium present in the formation. The uranium is extracted from the pregnant leaching solution and the water is reinjected into the ore-bearing formation. Water is recycled through the mining process several times and then is disposed of through deep-well injection, evaporation ponds or land application through irrigation after treatment for removal of uranium and radium.

The Highland Uranium Project near Douglas, Wyoming has reported waterborne selenium concentrations from 1,000 to 2,000 µg/L (ppb) in their in situ mining wastewater (information from permit filed at the Wyoming Department of Environmental Quality, Land Quality Division, Cheyenne, WY). The Wyoming Department of Environmental Quality (WDEQ) has permitted the mine to dispose of wastewater through land application. Wastewater is stored in holding ponds and is applied onto a grassland with center pivot irrigation systems. At full capacity the holding ponds are 2 to 13 hectares (ha) (5 to 32 acres) in size. The larger of the two ponds has never reached full capacity (Bill Kearney, Environmental Superintendent, Power Resources, Glenrock, WY, Personal communications, March 1, 2000). Currently, the mine has two center pivots in operation. The center pivots have been operational since 1989 and1995, respectively.

The effects of selenium on fish and aquatic migratory birds have been well documented (Eisler 1985, Ohlendorf et al. 1986, Hamilton et al. 1990, Ohlendorf et al. 1988, Skorupa and Ohlendorf 1991, Lemly 1993, Saiki and Ogle 1995). Selenium concentrations >2 µg/L in water are known to impair waterbird reproduction and survival due to the high potential for dietary toxicity through food chain bioaccumulation (Lemly 1993). To protect waterfowl, shorebirds, and other wildlife from adverse effects, waterborne selenium concentrations should be • 2 µg/L (Skorupa and Ohlendorf 1991; Lemly 1993). Waterborne selenium concentrations >3 µg/L exceed the bioaccumulation threshold for wildlife. Food organisms can bioaccumulate selenium from the water and supply a toxic dose of selenium to wildlife; however, the selenium concentration may not affect the health of the food organism (Lemly 1993).

Selenium enters the food chain almost entirely through vegetation and dietary plant selenium is readily absorbed by animals (up to 100%). This fact pertains to not only macrophytic vegetation but microscopic algae and phytoplankton, both of which serve as a principal food source for invertebrates (Ohlendorf et al 1993). Aquatic invertebrates also bioaccumulate selenium and can contain concentrations 2 to 6 times those found in aquatic plants. Selenium can concentrate in the food chain more than 300,000 times the concentration in the water (Besser et al. 1993). For example, the Kendrick irrigation project, located west of Casper, Wyoming has documented deformities and poor reproductive success in American avocets (*Recurvirostra americana*) and eared grebes (*Podiceps nigricollis*) resulting from elevated selenium concentrations. The median concentration of dissolved selenium in water samples from two closed basin ponds were 38 and 54 µg/L (See et al. 1992). Due to the bioaccumulation of selenium in food items from these ponds, aquatic birds suffered from impaired reproduction (See et al. 1992).

Impacts to waterfowl feeding on selenium contaminated food sources can occur in seven days (Heinz et al. 1990). Ingestion of water containing selenium concentrations as low as 2.2 mg/L can cause immune suppression in waterfowl (Fairbrother and Fowles 1990). During migration, birds are very stressed and become much more susceptible to the effects of environmental contaminants (Peterle 1991). Fairbrother and Fowles (1990) found selenium concentrations >10 µg/g in the livers of mallards (*Anas platyrhnchos*) given water with 2.2 mg/L selenium in the form of selenomethionine. Biological effects thresholds (dry weight) for sensitive aquatic birds such as waterfowl are 10 µg/g for liver tissue and 3 µg/g for eggs (Lemly 1993 and Heinz 1996). Selenium concentrations above these thresholds can cause impaired reproduction or mortality.

Little information is available on selenium bioaccumulation and toxicity in grassland species of passerine birds. Ohlendorf and Hothem (1995) and Santolo G.M. Santolo (G.M. Santolo, CH2M Hill, Sacramento, CA. Personal Communications, August 1999) report data on grassland species of passerine birds collected at Kesterson National Wildlife Refuge. Research on selenium mobilization and bioaccumulation in terrestrial communities has focused primarily on vegetation and ungulates. Forage species such as grasses can accumulate elevated levels of selenium in high selenium soils associated with uranium mining (Hossner et al. 1992). Raisbeck et al. (1996) found immune suppression in pronghorn (*Antilocapra americana*) fed an alfalfa-grass hay diet containing 15 µg/g of selenium. Acute poisoning has been documented in sheep (*Ovis aries*) fed plant material containing 3.2 to 12.8 µg/g of selenium (Eisler 1985).

This study was designed to: determine selenium concentrations in water, soil, terrestrial invertebrates, vegetation, birds and bird eggs; determine pathways of selenium in the food chain; and document potential adverse effects to migratory birds resulting from selenium bioaccumulation.

STUDY AREA

The Highland Uranium in situ mine is located in Converse County, Wyoming and is operated by Power Resources, Inc. (PRI). The mine is located approximately 40 km (25 miles) north of Douglas and 38 km (24 miles) northeast of Glenrock (Figure 1). The Satellite # 1 purge storage reservoir is approximately 2 ha (5 acres) in size with a maximum depth of 3.05 m (10 feet). The reservoir holds approximately 61,675 m^3 (50 acre-feet) of wastewater. The center pivot irrigates 23.5 ha (58 acres) of grassland. The irrigator is a low profile system with 106.68 cm (42-inch) drop pipes and is 263.8 m (865 feet) in length. The irrigator completes a rotation every 21.8 hours and applies approximately 0.68 cm (0.27 inches) of wastewater on the grassland per revolution. A small berm from 15 to 30 cm (six to 12 inches) high encircles the irrigated area to ensure that the wastewater remains on site. The irrigated area is nearly flat and is dominated by grasses such as brome (*Bromus tectorum*); foxtail barley (*Hordeum jubatum*); blue grama (*Bouteloua gracilis*); common buffalo grass (*Buchloe dactyloides*); western wheatgrass (*Agropyron smithii*); and needle and thread (*Stipa*

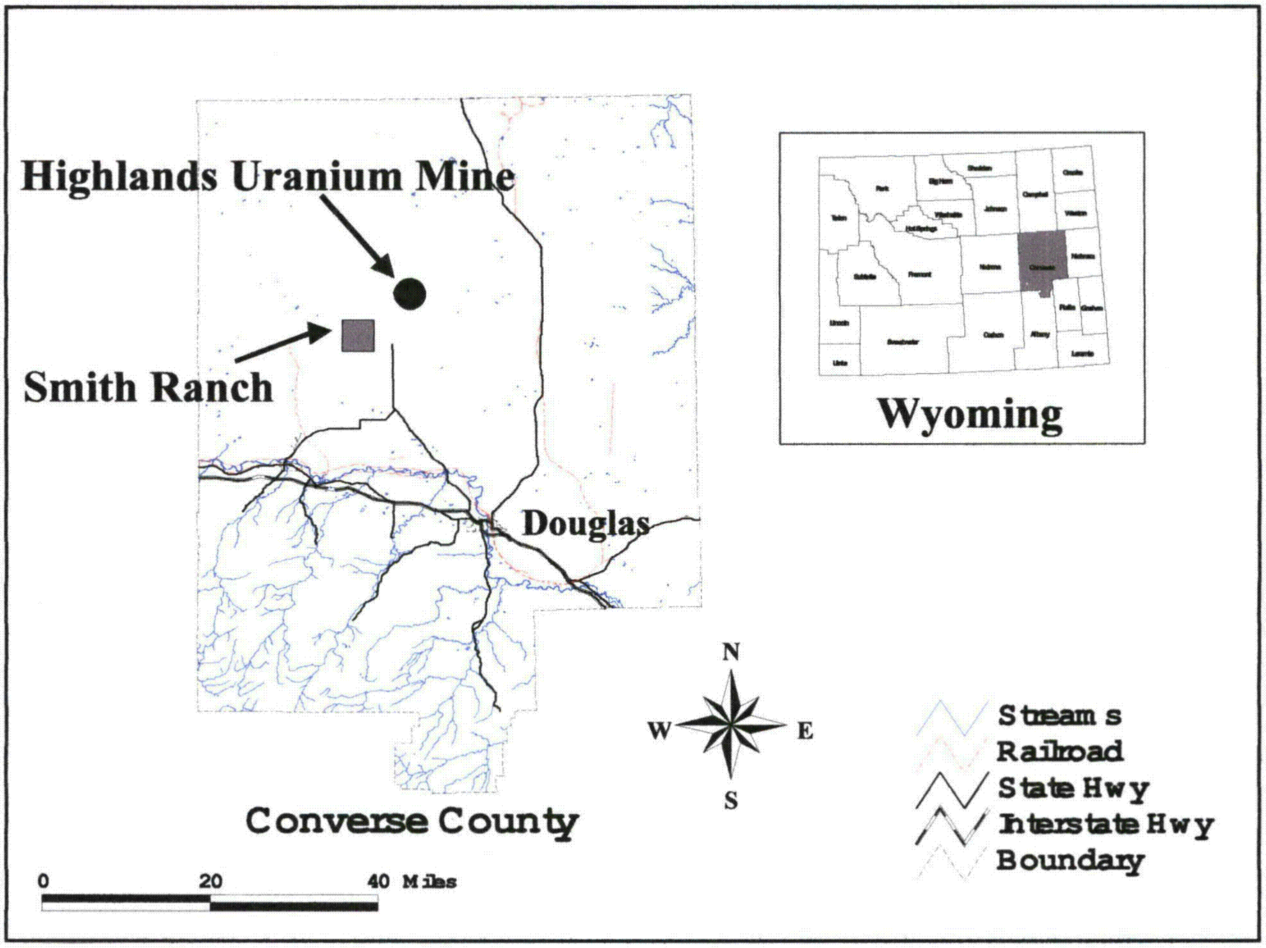

Figure 1. Location of the Highland in-situ uranium mine (Study Area) and the Smith Ranch (Reference Site), Converse County, Wyoming.

comata). Soils in the irrigated area consist of clay and clayey-loam Bidman and Ulm soils. These soils are slowly to moderately permeable.

The area receives an average of 30 cm (12 inches) of precipitation per year of which 45 percent falls during the months of May, June and July. The evaporation rate is 159.7 cm (62.9 inches) per year. Temperatures range from -40 °F in the winter to 100 °F in the summer. The prevailing winds are from the west and southwest with predominant speeds ranging from 17 to 33 km (11 to 21 miles) per hour.

Satellite # 1 purge storage reservoir and irrigation area 1 at the Highlands uranium mine (Study Area) were selected for this study. Satellite # 1 has operated since 1989. A center pivot irrigated area located at the Smith Ranch, approximately 16 kilometers (km) (10 miles) southwest of the Highlands uranium mine, was selected as the reference site (Reference Site). Alfalfa (*Medicago sativa*) is irrigated at the Reference Site. The radius of the irrigated area is 274.5 meters (m) (900 feet).

METHODS

Bird Surveys

Surveys were conducted once a week between 0800 and 1200 (MST), between May 5 and September 3, 1998. Stations were placed 200 m (658 ft) apart and 200 m out from the center pivot. Stations were marked with easily visible stake wire flags to avoid creating perch sites for birds which could influence results. Surveys were preformed by one of two observers or both observers together. Counts lasted 5 minutes, ten minutes if abundance was low at each station. All birds observed (seen or heard) within 75 m (246 ft) of a count station were identified by species. Additionally birds beyond 75 m were identified by species and noted on the data sheets as outside the area. Birds observed using the purge storage reservoir were also recorded.

Nesting Study

Twenty songbird nest boxes each were set up at the Study Area and the Reference Site. Nest boxes were checked weekly, recording nest condition, number of eggs, live young, dead young and presence/absence of adults. Nestlings were visually examined for anomalies. Songbird nests were located using random passes with a hand-held drag-line and through incidental flushes of females from nests. Each nest located was flagged ten m (32.8 ft) out from the nest in alignment with the center pivot of the irrigation system. Nest locations were flagged 10 m away to avoid detection of nests by predators. For each nest located, clutch size was recorded and one egg was randomly collected. Eggs were dissected and embryos aged and examined for deformities. The egg contents were submitted for trace elements analysis.

Trace Element Study

Vegetation, soil, water and terrestrial invertebrate samples were collected from the Study Area and the Reference Site. All equipment used to collect water, sediment, and soil samples was rinsed with deionized water and acetone prior to collection of each sample. Eight water samples were collected from the purge storage reservoir, the center pivot and from standing water within the irrigated grassland of the Study Area during June and August 1998 (Table 3). Two water samples were collected from the center pivot at the Reference Site in July and August 1998. Water samples were collected using 1-liter chemically-clean polyethylene jars with teflon-lined lids. The pH of the water samples collected for chemical analysis was lowered to approximately 2.0 with laboratory-grade nitric acid. Five soil samples were collected at each site in June 30 and July 1, 1998 to a depth of 15 cm (6 in) with a stainless steel spoon and placed in whirl-pak bags and frozen as soon as possible. Vegetation samples were clipped using chemically-cleaned scissors and placed in whirl-pak bags. Five grass samples (foxtail barley, and brome) and one dandelion (*Taraxacum officinale*) sample were collected from the Study Area. Four grass samples (brome and Kentucky bluegrass (*Poa pratensis*)) and one alfalfa sample were collected from the Reference Site. Pondweed (*Potamogeton* spp.) from the purge storage reservoir was collected by gloved hand and placed in whirl-pak bags. Five sediment samples were collected at the purge storage reservoir to a depth of 15 cm (6 in) with a stainless steel spoon and placed in whirl-pak bags and frozen within an hour after collection. Terrestrial invertebrates were collected using a sweep net, sorted to family and placed in chemically-clean 40 milliliter glass jars with teflon-lined lids. All samples were frozen within an hour after collection. Six composite samples of grasshoppers (Family Acrididae) from the Study Area and five from the Reference Site were submitted for trace element analyses.

Six red-winged blackbirds each were collected from the Study Area and the Reference Site, using a 20-gauge shotgun and steel shot. Bird livers and gizzards were dissected from the carcasses. The gizzard contents were removed and placed in chemically-clean glass vials and the livers in whirl-pak bags and frozen within an hour after collection. Six liver samples each from the Study Site and the Reference Site were submitted for trace element analysis. Two samples of red-winged blackbird gizzard contents were submitted to the laboratory for trace element analysis.

Water, sediment and biota samples were submitted to the Environmental Trace Substances Laboratory (ETSL) at Columbia, Missouri, under contract with the Service's Patuxent Analytical Control Facility (PACF) at Laurel, Maryland, for trace element analyses. Trace element analysis included scans for: arsenic, mercury, and selenium using atomic absorption spectroscopy. Inductively Coupled Plasma Emission Spectroscopy was used to scan a variety of elements including boron, barium, copper, lead, vanadium and zinc. Mercury samples were digested under reflux in nitric acid. Other samples were digested under reflux in nitric and perchloric acids. PACF conducted Quality Assurance/Quality Control on all samples analyzed by ETSL. Seven samples were lost in preparation at ETSL and included: two red-winged blackbird (*Agelaius phoeniceus*) livers and two European starling (*Sturnus vulgaris*) eggs from the Study Area, and three red-winged blackbird livers from the Reference Site. All analytical data for soil, sediment, and biota are reported in dry weight.

Statistics

Statistical analysis was performed using Systat statistical software. The Kruskal-Wallis One-Way Analysis of Variance test was used to compare selenium concentrations between the Study Area and the Reference Site. The probability level determining significance was $P<0.05$.

RESULTS

Bird Surveys

Field work was completed between May 28 and September 3, 1998. We observed 626 birds during point count surveys at the Study Area. During 90 point/count/days we observed 385 individuals comprising 14 species within the 75 m (246 feet) fixed point count radius. Western meadowlarks (*Sturnella neglecta*) were the most numerous species followed by the red-winged blackbird, lark bunting (*Calamospiza melanocorys*) and horned lark (*Eremophila alpestris*), respectively (Figure 2).

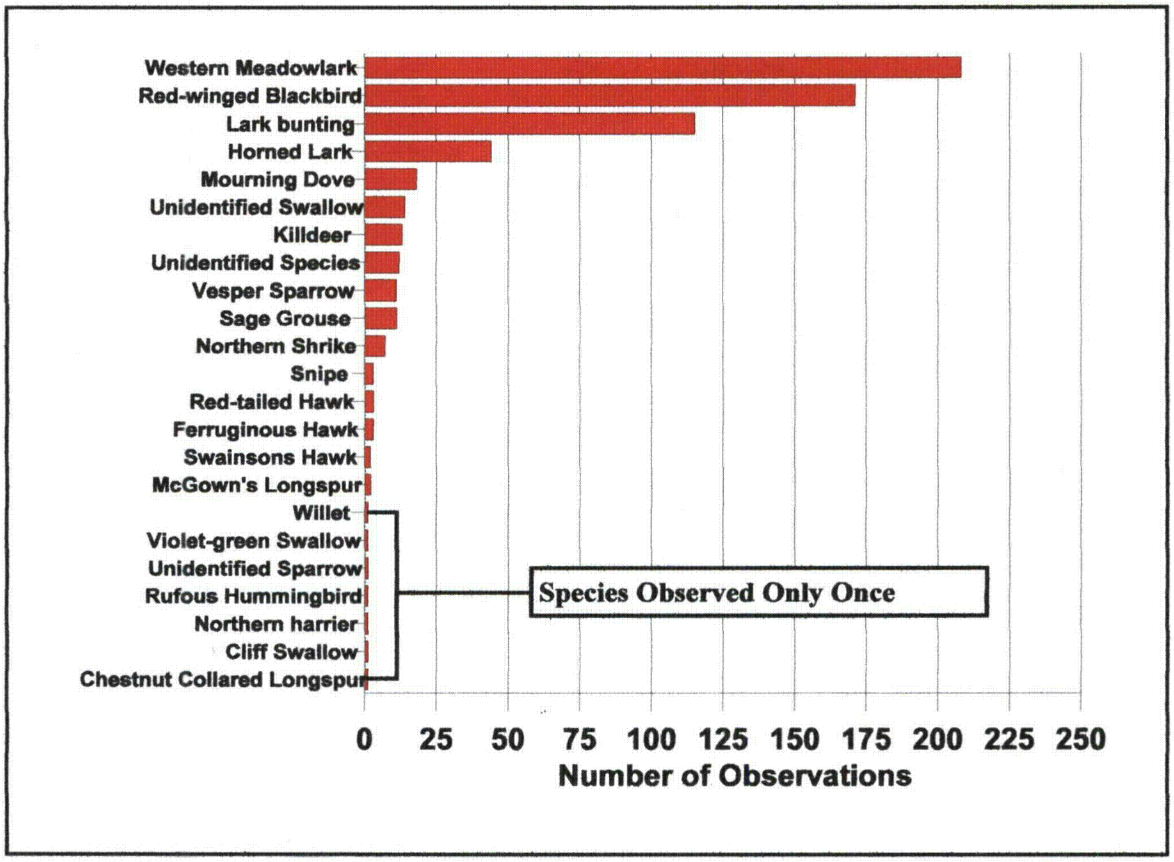

Figure 2. Species observed at the uranium mine irrigated area and the number of observations for each species.

The remaining 10 species accounted for only 17 percent of the observations. Of the 241 birds observed beyond the 75 m point count radius, only 9 individuals comprising six new species were observed. Peak observations for red-winged blackbirds, and lark buntings were in June; whereas, horned lark and western meadowlark numbers remained consistent throughout the survey period (Figure 3). Birds flying over and landing at the Study Area were the most frequent behaviors observed during the surveys followed by perching (Figure 4). Birds were also observed feeding and

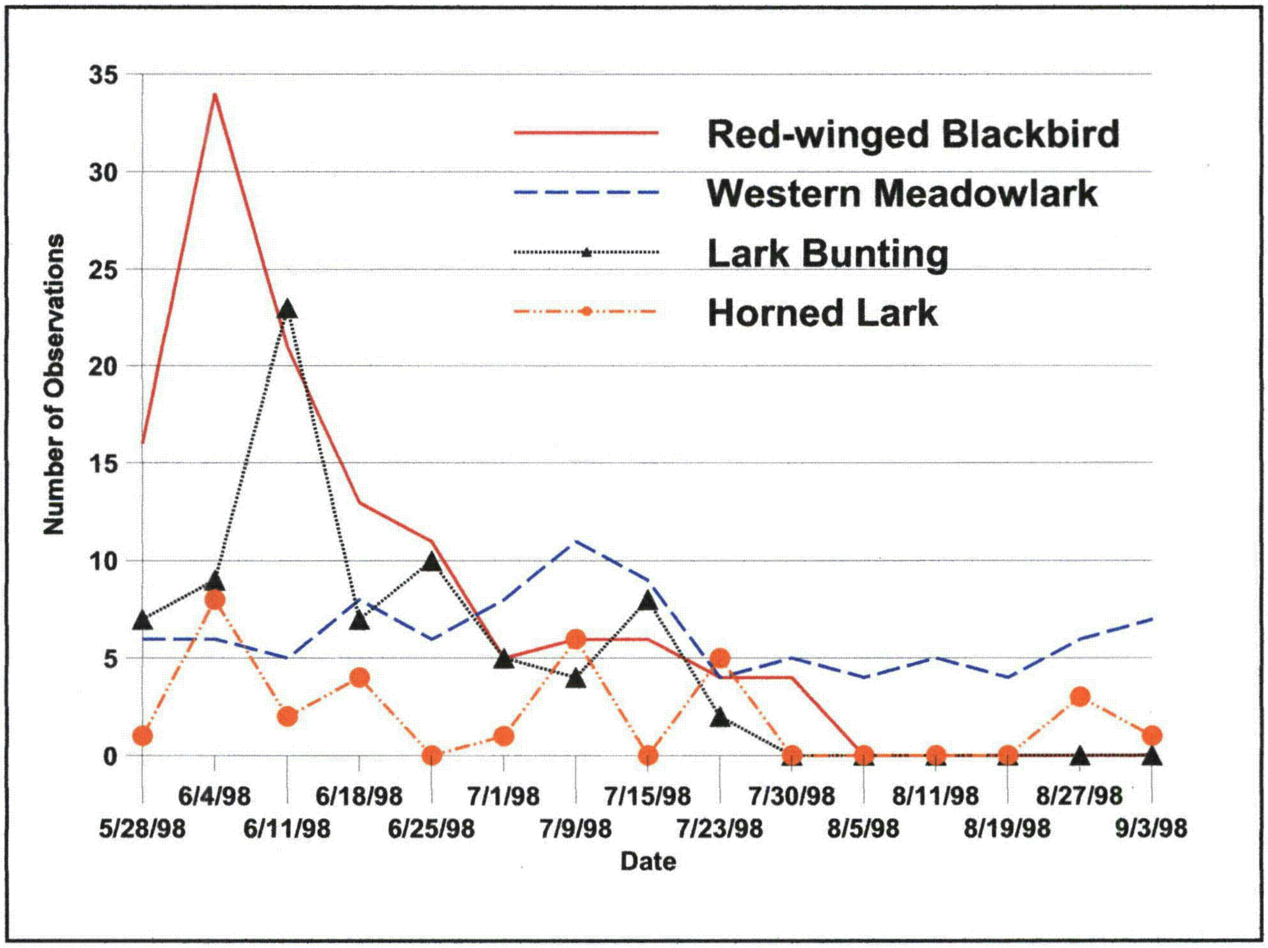

Figure 3. Daily counts of the four most common bird species observed at the uranium mine irrigated area, May 28, 1998 to September 3, 1998.

drinking at the Study Area. Nine birds were observed using the purge storage reservoir between May and September and included: two eared grebes; two gadwalls (*Anas strepera*); one hooded merganser (*Lophodytes cucullatus*); one sandpiper (Family Scolopacidae); two black terns (*Chlidonias niger*); and one mallard.

Nesting Data

Searches for ground-nesting native species at the Study Area revealed nesting by red-winged blackbirds as well as western meadowlarks and lark buntings. The nest boxes had little to no use by European starlings; therefore, the study focused on collecting eggs and livers from red-winged blackbirds at both the Study Area and the Reference Site since their nests were the most numerous.

Nine red-winged blackbird nests were monitored at the Study Area and 13 were monitored at the Reference Site. Red-winged blackbird nests at the Study Area were located in tall bunch grass as well as in a small stand of cattails (*Typha* sp.) growing in ponded water. The nests at the Reference Site were located in a cattail marsh immediately adjacent to the irrigated alfalfa field. Of the nine

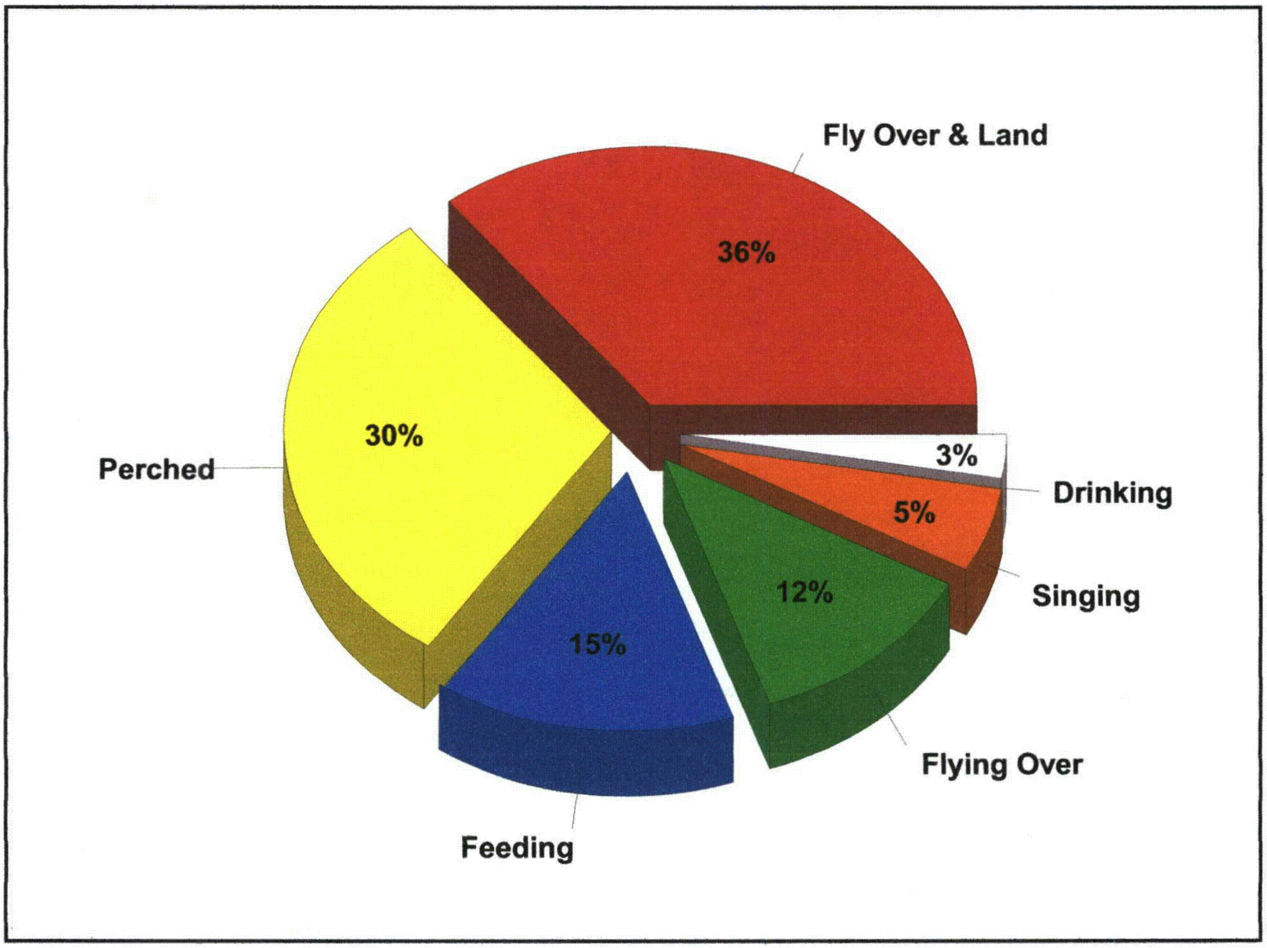

Figure 4. Behaviors observed during the bird surveys at the uranium mine irrigated site.

nests at the Study Area, six were abandoned and the fate of three was unknown. A late snowstorm on June 4 probably caused the abandonment of three nests. The cause of abandonment in two nests was unknown and one nest failed due to disturbance by the investigators. At the Reference Site five nests successfully hatched and six were presumed to have hatched. One nest was abandoned and the fate of four eggs in one nest was unknown. One of six red-winged blackbird eggs collected from the Study Area was infertile. The remaining five red-winged blackbird eggs collected at the Study Area were in the early stages of incubation (1 to 4 days). Incubation stages in red-winged blackbird eggs collected at the Reference Site ranged from 1 to 11 days; all eggs except one were fertile. Two western meadowlark nests were monitored at the Study Area; however, the fate of the eggs was unknown. The eggs at these nests could have been taken by a predator. One egg collected from one of the two meadowlark nests was fertile, the embryo appeared normal and was in the mid-stages of incubation (7 days). Of the two starling eggs collected at the Study Area, one was fertile and the other infertile. Both embryos were one day old. The one starling egg collected at the Reference Site was fertile. No abnormalities were observed in embryos collected from the Study Area and the Reference Site; however, it should be noted that all eggs were in early to mid-stages of incubation making it difficult to determine if the embryos were malformed (Table 1).

Table 1. Selenium concentrations (in µg/g dry weight) in songbird eggs collected from the in-situ uranium mine grassland (Study Site) receiving wastewater via irrigation and from a Reference Site.

<table>
<tr><td colspan="6"><i><u>Study Site</u></i> - <i>Red-winged Blackbird</i></td></tr>
<tr><th>Sample #</th><th>Fertile</th><th>Incubation Stage[1]</th><th>Viable Embryo</th><th>Malformation Observed</th><th>Se (µg/g)</th></tr>
<tr><td>PRIRBE01</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>15</td></tr>
<tr><td>PRIRBE02</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>20</td></tr>
<tr><td>PRIRBE03</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>15</td></tr>
<tr><td>PRIRBE04</td><td>No</td><td>Early</td><td>Unknown</td><td>No</td><td>13</td></tr>
<tr><td>PRIRBE05</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>22</td></tr>
<tr><td>PRIRBE06</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>19</td></tr>
<tr><td colspan="6"><i><u>Reference Site</u></i> - <i>Red-winged Blackbird</i></td></tr>
<tr><td>REFRBE01</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE02</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE03</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE04</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE05</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE06</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE07</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>2</td></tr>
<tr><td>REFRBE08</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE09</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td>REFRBE12</td><td>Unknown</td><td>Early</td><td>Unknown</td><td>No</td><td>4</td></tr>
<tr><td colspan="6"><i><u>Study Site</u></i> - <i>European Starling</i></td></tr>
<tr><td>PRISTE03</td><td>Yes</td><td>Early</td><td>No</td><td>No</td><td>7</td></tr>
<tr><td>PRISTE05</td><td>No</td><td>Early</td><td>No</td><td>No</td><td>8</td></tr>
<tr><td colspan="6"><i><u>Reference Site</u></i> - <i>European Starling</i></td></tr>
<tr><td>REFSTE01</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>3</td></tr>
<tr><td colspan="6"><i><u>Study Site</u></i> - <i>Western Meadowlark</i></td></tr>
<tr><td>PRIWME01</td><td>Yes</td><td>Early</td><td>Yes</td><td>No</td><td>18</td></tr>
</table>

[1] Early = 1 - 4 days; Mid = 5 - 8 days; Late = > 8 days

Trace Elements

Arsenic and boron were elevated in pondweed samples collected from the purge storage reservoir. Arsenic concentrations ranged from 1.7 to 3.7 µg/g. Boron concentrations in pondweed ranged from 26 to 236 µg/g dry weight. A water sample collected from pooled water in the irrigated field at the Study Area had an elevated zinc concentration of 7,410 µg/L. No analytical anomalies were reported by PACF. The source of the zinc is unknown.

Selenium concentrations in soil, grasses, grasshoppers, and red-winged blackbird eggs and livers collected from the Study Area were significantly higher than the concentrations found at the Reference Site (P<0.05)(Figure 5 and Table 2). Mean selenium concentrations in water from the Study Area were 19 times higher than those from the Reference Site.

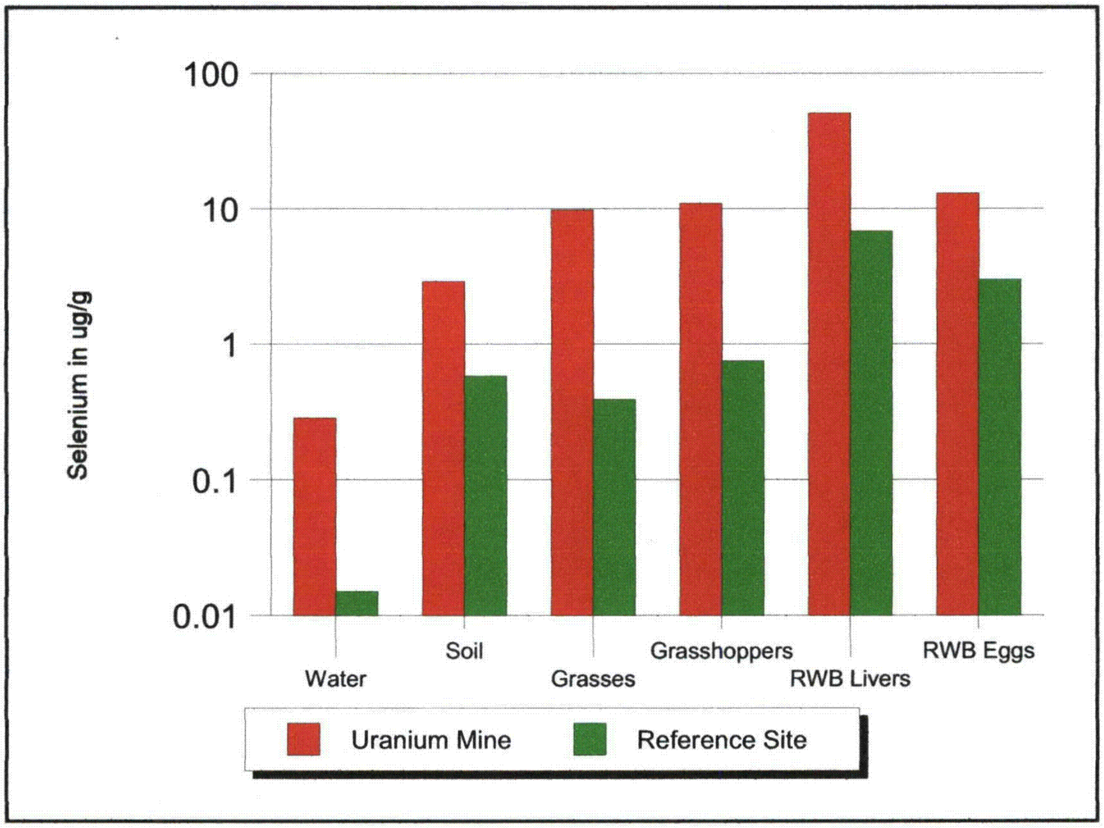

Figure 5. Mean selenium concentrations in water, soil and biota from the PRI in-situ uranium mine and Reference Site irrigated areas, Converse County, Wyoming. Concentrations are in µg/g dry weight except for water which are reported in µg/L. [RWB = Red-winged blackbird]

trations in water, soil and biota from the in-situ uranium mine grassland receiving wastewater via irrigation and ncentrations are in µg/g dry weight except for water which are reported in µg/L.

	Study Area				Reference Site				Comparisons	
	n	Mean	Range	Variance	n	Mean	Range	Variance	Δ	Kruskal - Wallis p Value*
	8	285	32 - 450	0.026	2	15	1 - 28	NC	270	NC
	5	3.1	2.6 - 4.2	0.40	5	0.63	0.55 - 0.81	0.01	2.51	**0.009**
	5	12.7	6.8 - 24	50.05	4	0.43	0.3 - 0.62	0.02	12.26	**0.014**
	6	12.8	11 - 20	12.97	5	0.73	0.6 - 0.87	0.01	12.10	**0.005**
zzard	2	47.5	12 - 83	NC	2	0.7	0.6 - 0.8	NC	46.8	NC
ers	4	46.8	33 - 53	85.8	3	6.8	3.7 - 10	9.92	39.94	**0.034**
gs	6	17.4	13.2 - 22	11.93	13	3	2.4 - 3.6	0.10	14.34	**0.001**

rea and Reference Site means

<0.05)

Selenium concentrations in water samples collected from the pooled water in the irrigated field at the Study Area were lower than those from samples collected from the center pivot irrigator, and the purge storage reservoir (Table 3).

Table 3. Selenium concentrations (µg/L) in water collected from the the in-situ uranium mine grassland (Study Site) receiving wastewater via irrigation.

Site	n	Mean	Range
Center Pivot Irrigator	2	395	340 - 450
Purge Storage Reservoir	3	307	260 - 350
Pooled Water at Irrigated Area	3	46	32 - 69

n = number of samples

Five sediment samples collected from the purge storage reservoir at the uranium mine had selenium concentrations ranging from 7.8 to 38.8 µg/g with a mean of 18.5 µg/g. Selenium concentrations >4 µg/g in sediments are considered a high hazard for the aquatic bird food chain (Lemly 1995). Pondweed samples collected from the purge storage reservoir at the uranium mine had selenium concentrations ranging from 434 to 508 µg/g with a mean of 459 µg/g. These concentrations are 144 to 169 times higher than the 3 µg/g dietary threshold for potential toxic effects in aquatic migratory birds.

A dandelion sample collected at the Study Area had a selenium concentration of 28 µg/g. An alfalfa sample from the Reference Site had 0.41 µg/g of selenium. Selenium concentrations in livers from three lark bunting nestlings collected from one nest in the Study Area had selenium concentrations ranging from 7.8 to 8.8 µg/g. A composite sample of the gizzard contents from these three nestlings had 1.6 µg/g of selenium. Selenium concentrations in two composite samples of gizzard contents from red-winged blackbirds were 12 and 83 µg/g at the Study Area and 0.6 and 0.8 at the Reference Site. Three starling eggs (one from each of three nests) collected from the nest boxes placed at the Study Area had a mean selenium concentration of 7 µg/g with a range of 6.2 to 7.9 µg/g. One starling egg collected from a nest box at the Reference Site had a selenium concentration of 2.7 µg/g. Two western meadowlark eggs (each from two nests) from the Study Area had selenium concentrations of 18 and 28 µg/g.

DISCUSSION

Elevated selenium concentrations in water, soil, grasshoppers, and red-winged blackbird eggs and livers collected from the Study Area demonstrate that selenium is being mobilized and bioaccumulated in the food chain. Mean selenium concentrations in soil and water were 5 and 19 times higher, respectively, in the Study Area than at the Reference Site. Mean selenium concentrations in biota were 5.8 to 30 times higher in the Study Area than at the Reference Site.

It is unclear why selenium concentrations in pooled water at the Study Area were significantly lower than waterborne concentrations in the purge storage reservoir and the irrigator. Selenium could be removed from solution and bound to the wet soil/sediments in the pools (Lemly and Smith 1987). Additionally, cattails growing in the pooled water could be removing the selenium from the water as cattails are strong selenium accumulators (Schuler et al. 1990).

Sediment collected from the purge storage reservoir at the uranium mine had selenium concentrations ranging from 7.8 to 38.8 µg/g with a mean of 18.5 µg/g. Selenium concentrations >4 µg/g in sediments are considered a high hazard for the aquatic bird food chain (Lemly 1995). The selenium concentrations in the sediment were of the same magnitude as sediment from Goose Lake, a closed basin, at the Kendrick irrigation project near Casper, Wyoming (See et al. 1992) where reproduction in aquatic migratory birds was adversely affected.

Selenium concentrations in pondweed collected from the purge storage reservoir were extremely elevated (434 to 508 µg/g). These concentrations were four to five times higher than the maximum concentration of 104 µg/g reported for pondweed from several irrigation projects in the western United States by the Department of Interior's National Irrigation Water Quality Program (NIWQP). The NIWQP investigated irrigation-induced selenium contamination in the western United States. Selenium concentrations in pondweed were also almost twice as high as those reported by Schuler et al. (1990) for widgeon grass (*Ruppia maritima*) at Kesterson Reservoir in California. Heinz et al. (1987 and 1989) found that selenomethionine concentrations of 15 to 20 µg/g in the diet of mallards resulted in mortality. It is unknown if waterfowl have a taste aversion to the pondweed at the purge storage reservoir due to the extremely high concentrations of selenium or if they are consuming enough of this pondweed to suffer mortality or other chronic effects. The limited amount of bird use observed at this reservoir suggests that a low number of waterfowl would be exposed if they feed on the pondweed. Observations on bird use at the purge storage reservoir by mine personnel also show that waterfowl do not use the pond for any substantial amount of time (Bill Kearney, Environmental Superintendent, Power Resources, Glenrock, WY, personal communications, February 28, 2000).

Arsenic concentrations in pondweed also were at the level of concern of 2 to 5 µg/g (U.S. Dept. Interior 1998). Boron concentrations in pondweed ranged from 26 to 236 µg/g dry weight with a mean concentration of 134 µg/g. Dietary levels as low as 30 µg/g and fed to adult mallards adversely affected the growth rate of their ducklings (Smith and Anders 1989). Hoffman et al. (1990) reported reduced growth in female mallard ducklings fed diets containing 100 µg/g of boron.

The mean total soil selenium at both the Study Area and the Reference Site (3.1 and 0.63 µg/g, respectively) exceeded the mean concentration for soils in the western United States (0.23 µg/g);

however, the selenium concentration ranges were within those reported for western U.S. soils (Shacklette and Boerngen 1984). The mean total soil selenium at the Study Area was comparable to soil from several sites from the Kendrick irrigation project near Casper, Wyoming that had total selenium > 2 µg/g (See et al. 1992). Soils with total selenium concentrations > 2 µg/g are usually associated with selenosis in livestock (Thorton 1981).

Mean selenium concentrations in grasses from the Study Area were 30 times higher than at the Reference Site and were four times higher than the concentrations in the soil. Grasses are selenium nonaccumulators and generally contain < 25 µg/g of selenium (Wu 1998). Selenium in the soil is usually available as selenate and selenite, both of which are absorbed by grasses and transformed into organic selenium compounds such as selenomethionine (Wu 1998) which is highly available and toxic to birds (Heinz 1996, Heinz et al. 1989).

Selenium concentrations in grasshoppers from the Study Area were 18 times higher than the Reference Site and were equivalent to the concentrations found in the grasses. Mean selenium concentrations in grasshoppers from the Study Area were twice as high as the concentrations reported by Santolo and Yamamoto (1999) from grasshoppers at selenium-contaminated grasslands at Kesterson Reservoir in California; however, the maximum selenium concentration at the mine did not exceed that reported at Kesterson. Grasshoppers bioaccumulate the selenium from the vegetation at the Study Area. The grasshoppers in turn are consumed by birds inhabiting the Study Area. Two composite samples of gizzard contents from several red-winged blackbirds collected from the Study Area had selenium concentrations of 12 and 83 µg/g which shows that the birds are ingesting elevated selenium. Excess selenium consumed by female birds is usually incorporated into their eggs (O'Toole and Raisbeck 1998). Elevated selenium substitutes sulfur in proteins formed in the cells which disrupts the normal development of the embryo and leads to terata and mortality (Ohlendorf and Hothem 1995, O'Toole and Raisbeck 1998).

The range of selenium concentrations in red-winged blackbird eggs from the uranium mine (13.2 to 22 µg/g) was similar to or slightly higher than those reported for the same species and matrix from several irrigation projects in the western United States by the NIWQP. Selenium concentrations in red-winged blackbird eggs reported by the NIWQP ranged from 2 to 18 µg/g. Red-winged blackbird eggs collected from the Uncompahgre Irrigation Project in western Colorado, an area with elevated selenium, had selenium concentrations ranging from 4 to 18 µg/g. Selenium at these irrigation projects was mobilized by irrigation of seleniferous soils with resultant bioaccumulation by fish and wildlife (Seiler 1996). The mean selenium concentration in red-winged blackbird eggs from the uranium mine (17.4 µg/g) was also higher than the 11.1 µg/g mean value reported for red-winged blackbird eggs reported at Martin Reservoir in Texas (King 1988 and Skorupa 1998). Reduced egg hatchability was reported in the red-winged blackbird eggs at Martin Reservoir; however, it is unclear if it was associated with the elevated selenium concentrations (J. Skorupa, U.S. Fish and Wildlife Service, Sacramento, CA. Personal Communications, February 23, 2000). We were unable to determine egg hatchability in red-winged blackbird eggs at the uranium mine site due to the low number of nests, the confounding effects of a late-season snow storm and possibly nest predation and/or observer disturbance. Nests at the reference site were successful as this site did not receive as much snow during the June 4th storm. Additionally, the blackbird nests were located on a small marsh and received greater protection from predators. All red-winged blackbird eggs collected from the uranium mine contained concentrations of selenium (13.2 to 22 µg/g) well above the threshold (>8 µg/g) known to be reproductively toxic to sensitive bird species (Lemly 1993, Ohlendorf et al. 1993, Heinz 1996). It should be noted that the reproductive toxicity threshold for red-winged blackbirds is unknown.

Selenium concentrations in western meadowlark eggs were slightly higher than the range reported for the same species at selenium-contaminated grasslands at Kesterson Reservoir in California by G.M. Santolo (G.M. Santolo, CH2M Hill, Sacramento, CA. Personal Communications, August 1999) (3.9 to 17 µg/g) and by Ohlendorf and Hothem (1995) (9.7 to 24 µg/g). Selenium concentrations in western meadowlark eggs (18 and 28 µg/g, n=2) also exceeded the toxic threshold of 8 µg/g for sensitive species of birds; however, the sensitivity of meadowlarks to selenium is unknown.

Selenium concentrations in livers from red-winged blackbirds collected from the Study Area were nearly seven times higher than the Reference Site and higher than those reported by the NIWQP for livers from blackbirds collected from the Los Pinos River in southwestern Colorado (4.2 to 6.8 µg/g) and from red-winged blackbirds collected from the lower Gila River in Arizona in 1994 and 1995 (8 to 14 µg/g) (Kirke King, U.S. Fish and Wildlife Service, personal communications, Nov. 1999).

MANAGEMENT IMPLICATIONS

Mobilization and bioaccumulation of selenium and its potential adverse effects on fish and migratory birds have been intensively documented in irrigation projects throughout the western United States as well as in reservoirs and wetlands receiving selenium contaminated water from industrial sites such as coal-fired power plants and oil refineries (Skorupa 1998). Our study shows that application of in situ uranium mine wastewater containing elevated selenium on a grassland can lead to bioaccumulation of this element in the food chain. Although we were unable to determine if the elevated selenium concentrations were causing impaired reproduction or other effects on the resident songbirds inhabiting the irrigated grassland at the mine, we did document elevated selenium concentrations in red-winged blackbird eggs comparable to those associated with reduced hatchability in the same species at Martin Reservoir, Texas (Skorupa 1998). A controlled egg hatchability study using an incubator and eggs from birds nesting at the Study Area and the Reference Site may help in determining if the elevated selenium concentrations are causing impaired reproduction in red-winged blackbirds as well as lark buntings and meadowlarks.

Based on the results of this study efforts should be made to discourage red-winged blackbirds from nesting at the area irrigated with in situ uranium mine wastewater. Selenium concentrations in red-winged blackbird eggs were at levels suspected of causing reduced hatchability in this species. Red-winged blackbirds can be discouraged from nesting at the irrigated area by preventing the ponding of water and the growth of cattails. Although bioaccumulation of selenium was documented in lark buntings and western meadowlarks, the effects of this trace element on these grassland bird species are unknown. Additional study is needed to determine the sensitivity of these species to selenium.

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Appendix
Analytical Results

ug/L) in water collected from the Highland Uranium In Situ Mine irrigated area and a reference site, Smith Ranch,

Study Area						Reference Site	
PRIWATR3	PRIWATR4	PRIWATR5	PRIWATR6	PRIWATR7	PRIWATR8	REFWATR2	REFWATR6
30 June 1998	30 June 1998	30 June 1998	30 June 1998	11 August 1998	11 August 1998	14 July 1998	11 August 1998
PRI-Pond	PRI-Pivot	PRI-Pool	PRI-Pond	PRI-Pond	PRI-Pool	Smith Ranch	Smith Ranch
730	3150	330	720	90	330	<30.0	<50
2.2	5.5	1.9	2.2	2.6	3.5	0.9	1
140	160	190	150	130	170	81	75
131	320	136	128	223	149	20	21
<0.2	0.3	0.3	0.3	<0.2	<0.2	<0.2	0.4
0.2	1.1	0.4	0.38	<0.1	<0.1	<0.1	<0.1
<0.9	3.1	<0.9	0.001	1	1	<0.9	<0.9
3	268	8.3	3	<2	3	<2.00	<2
350	8960	483	410	53	910	110	130
<0.2	<0.2	<0.2	<0.2	<0.4	<0.4	<0.2	<0.4
84300	85000	111000	83200	87700	162000	18400	17800
160	757	385	175	107	1430	47	43
<5	<5	<5	<5	<5	<5	<5	<5
21	55	27	21	22	28	1.1	<3
0.2	157	1.1	0.52	0.41	0.79	3.5	<0.09
350	450	69	310	260	37	28	1
3690	3890	4780	3640	3950	6230	690	679
5	22	8.8	6.2	8	13	<2	6
8.7	7410	15	23	<6	<6	12	<6

ug/g dry weight) in soil collected from the Highland Uranium In Situ Mine irrigated area and a reference site, Smith
ng.

Study Area				Reference Site				
2	PRISOIL3	PRISOIL4	PRISOIL5	REFSOIL1	REFSOIL2	REFSOIL3	REFSOIL4	REFSOIL5
0	18800	19500	17000	11600	11500	11400	10100	11900
.4	5.1	3.8	3.6	3.7	3.4	3.3	3.7	3.8
.2	2.4	3	2	3.9	5.5	3	3.8	4.2
.1	113	128	148	75.8	76.1	84.5	73.5	75.5
1	1.4	1.1	1.3	0.99	0.92	1.2	0.93	1.2
0	<.200	<.200	<.200	0.3	0.3	0.3	0.3	<.100
9	32	28	35	19	17	20	17	16
2	17	16	18	15	14	17	15	16
0	21600	18400	21200	13100	12500	14800	12600	13200
2	0.019	0.018	0.022	0.012	0.012	0.016	0.015	0.015
0	4570	4350	5190	4000	4040	4670	3910	4110
7	222	216	191	229	223	228	227	224
0	<.500	<.500	<.500	<.500	<.500	<.500	<.500	<.500
1	18	16	18	13	13	17	14	15
2	13	13	12	13	12	15	13	13
.8	4.2	3.2	2.9	0.81	0.55	0.58	0.64	0.56
.2	68.1	55.9	75.7	49.1	46.3	45.8	45.1	49
3	38.9	32	39.6	23	21	21	19	20
8	58.6	55.6	56.5	58	57.3	66.8	55.1	60

g/g dry weight) in vegetation collected from the Highland Uranium In Situ Mine irrigated area and a reference site, Smith ng.

	Study Area				Reference Site				
	PRIVEG03	PRIVEG04	PRIVEG05	PRIVEG06	REFVEG01	REFVEG02	REFVEG03	REFVEG04	REFVEG05
	Brome	Foxtail Barley	Foxtail Barley	Brome	Brome Grass	Brome Grass	Kentucky Bluegrass	Brome Grass	Alfalfa
0	20	20	8	10	20	40	44	42	200
2	<.0900	<.0900	<.0900	<.0900	<.0900	0.1	<.0900	0.1	0.1
0	9.9	5.1	5.2	11	11	13	12	19	53
4	13	6.7	5.9	9	33.4	34.4	14	25	8.5
0	<.0200	<.0200	<.0200	<.0200	<.0200	<.0200	<.0200	<.0200	<.0200
4	0.082	<.0200	<.0200	0.04	0.081	0.15	0.09	0.11	0.18
0	2.3	3.8	4	3.2	3.2	3.2	1.6	1.9	0.5
3	6.2	7.4	8.9	4.7	12	34	9.9	20	7.3
1	53	48	42	51	64	72	82	73	165
2	0.01	0.01	<.00900	0.01	0.01	<.00900	0.01	0.01	<.00900
0	2220	747	986	1580	1740	2040	2630	2170	4180
5	100	78.3	65.4	155	60.6	48.7	69.9	48	39.4
0	<.500	0.5	<.500	<.500	2	1	1	3	4.5
1	0.8	1.4	1.4	1.2	1.4	1	1.4	0.6	2.3
0	<.0700	<.0700	<.0700	<.0700	<.0700	<.0700	<.0700	0.09	<.0700
5	9.8	15	24	6.8	0.31	0.62	0.3	0.47	0.41
1	79.8	32.6	25.4	53.8	36.3	40.6	31	33.4	93.4
0	<.800	<.800	<.800	<.800	<.800	<.800	<.800	<.800	<.800
7	22	18	19	17	25	28	28	25	32.8

g/g dry weight) in grasshoppers collected from the Highland Uranium In Situ Mine irrigated area and a reference site, Vyoming.

Study Area				Reference Site				
NV02	PRIINV03	PRIINV04	PRIINV05	REFINV01	REFINV02	REFINV05	REFINV06	REFINV07
36	38	31	40	230	98	440	520	250
0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.3	0.1
14	16	13	13	19	17	13	14	15
1.9	2	1.9	2.2	3.1	2.1	3.5	4.1	2.9
<.0200	<.0200	<.0200	<.0200	<.0200	<.0200	0.02	0.03	<.0200
0.44	0.45	0.36	0.31	0.28	0.32	0.28	0.24	0.31
0.2	<.200	<.200	<.200	0.3	<.200	0.5	0.5	0.5
36	39	36	39	28	31	30	28	28
63	69	63	71	169	110	261	360	174
<.0500	<.0500	<.0500	<.0400	<.0400	<.0500	<.0400	<.0500	<.0400
1250	1180	1220	1230	1240	1150	1140	1230	1130
31	29.8	31.8	33.6	12	10	12	13	10
0.5	<.500	0.7	<.500	1.6	2	1	1	1.8
<.400	1	0.9	1	1	0.9	1	1	1
<.0700	<.0700	<.0700	<.0700	<.0700	<.0700	0.1	0.1	<.0700
20	13	11	11	0.78	0.75	0.6	0.87	0.65
31.6	25.9	24.9	25.8	17.7	18.1	16.6	16.1	16.8
<.800	<.800	<.800	<.800	<.800	<.800	1	1	<.800
134	146	140	144	140	131	142	147	143

g/g dry weight) in livers from Red-winged Blackbird collected from the Highland Uranium In Situ Mine irrigated area
, Converse County, Wyoming.

Study Area			Reference Site		
RIRWB04	PRIRWB05	PRIRWB06	REFRWB01	REFRWB05	REFRWB06
<10.0	<9.00	<10.0	<10.0	<9.00	<20.0
<.100	<.100	<.100	<.100	<.100	<.200
18	20	25	24	21	48
<.200	<.200	<.200	<.200	<.200	<.300
<.0300	<.0200	<.0200	<.0300	<.0200	<.0400
1.2	0.98	0.943	0.04	0.82	0.45
<.200	<.200	0.1	0.4	<.200	0.4
25	18	18	27	18	18
989	1150	571	1320	784	1200
0.27	0.093	0.07	0.04	0.36	0.34
745	765	807	815	826	741
4.9	5.1	6.8	2.8	3.4	4.1
4.3	3.5	3.8	2.7	3.8	3
<.500	<.400	<.100	<.500	<.400	<.200
<.0900	<.0800	<.0900	<.100	<.0800	<.200
33	53	50.1	3.7	6.8	10
0.42	0.3	0.2	0.78	0.2	<.200
<1.00	<.900	<.900	<1.00	<.900	<2.00
85.1	81	88.4	72.8	76.5	73.3

Appendix F. Trace elements (in µg/g dry weight) in livers from Lark Buntings collected from the Highland Uranium In Situ Mine irrigated area, Converse County, Wyoming.

Element	PRILBNL1	PRILBNL2	PRILBNL3
Al	<20.0	<20.0	<20.0
As	<.200	<.200	<.200
B	72	33	34
Ba	<.400	<.400	<.400
Be	<.0500	<.0600	<.0600
Cd	<.0400	<.0500	<.0500
Cr	<.400	<.500	<.500
Cu	24	25	33
Fe	1250	929	809
Hg	0.06	<.0400	<.0400
Mg	813	821	805
Mn	4.8	4.2	5.3
Mo	4	3	3
Ni	<.900	<1.00	<1.00
Pb	<.200	<.200	<.200
Se	7.8	8.8	8.6
Sr	1.5	0.4	0.4
V	<2.00	<2.00	<2.00
Zn	94.5	77.6	96

Appendix G. Trace elements (in µg/g dry weight) in the gizzard contents from Lark Buntings and Red-winged Blackbirds collected from the Highland Uranium In Situ Mine irrigated area, and a reference site, Smith Ranch, Converse County, Wyoming.

	PRILBNC1	PRIRWBC1	PRIRWBC2	REFRWBC1	REFRWBC2
Element	Lark Bunting	Red-Winged Blackbird	Red-Winged Blackbird	Red-Winged Blackbird	Red-Winged Blackbird
Al	4150	130	430	890	670
As	1.5	0.3	0.57	0.3	3.2
B	9.4	61.3	42	62	62
Ba	176	5.4	29.1	12	7.1
Be	0.99	<.0200	0.04	<.0300	0.21
Cd	0.31	0.77	0.61	0.51	0.84
Cr	4	0.63	1	1.9	1.3
Cu	18	37	26	23	25
Fe	2510	181	354	618	4560
Hg	<.0500	<.0500	<.0700	<.0700	<.0800
Mg	1700	1850	2640	1690	1390
Mn	181	82.5	179	54	85.8
Mo	0.8	1	2	2	2
Ni	1.5	<.400	6.6	<.600	<.600
Pb	2.6	0.1	0.2	1	1.4
Se	1.6	12	83	0.6	0.8
Sr	235	42.1	146	46.3	145
V	5.9	<.800	<1.00	3	4.3
Zn	120	265	176	210	178

ts (in µg/g dry weight) in Red-winged Blackbird eggs collected from the Highland Uranium In Situ Mine irrigated area, Ranch, Converse County, Wyoming.

Study Area					Reference Site								
	PRIRBE03	PRIRBE04	PRIRBE05	PRIRBE06	REFRBE01	REFRBE02	REFRBE03	REFRBE04	REFRBE05	REFRBE06	REFRBE07	REFRBE08	REFRBE09
0	<4.00	<5.00	<10.0	<10.0	35	<3.00	<3.00	<3.00	<3.00	<3.00	<10.0	<9.00	<9.00
0	<.100	<.100	<.100	<.100	<.100	<.100	<.100	<.100	<.100	<.100	<.100	<.100	<.0900
0	<2.00	<2.00	0.9	<.700	2	2	2	2	<1.00	2	1	1	1
.6	3.6	3.5	6.9	3.4	1.3	0.58	0.96	2.1	0.44	1.9	2.1	3.3	2.5
0	<.0300	<.0300	<.0200	0.02	<.0200	<.0200	<.0200	<.0200	<.0200	<.0200	<.0300	<.0200	<.0200
0	<.0200	<.0200	0.13	<.00900	<.0200	<.0200	<.0200	<.0200	<.0200	0.02	<.0200	0.04	<.0200
.4	0.4	0.5	0.4	0.3	0.5	0.5	0.63	0.69	0.59	0.7	<.200	0.5	0.4
.9	3.3	2.6	3.1	1	3.2	3.4	3.7	3.3	2.7	3.5	3.3	3.8	3.3
21	119	188	180	211	142	132	135	122	104	111	167	146	168
6	0.063	<.0200	0.08	0.07	0.081	0.13	0.04	0.05	0.18	0.16	0.06	0.062	0.06
5	423	426	344	443	505	591	360	511	405	519	399	478	498
.5	3.1	5.2	4	6.1	4.2	3.4	5.4	3.8	3.2	4.6	3.9	2.4	3
0	<1.00	<1.00	<.800	<.700	<1.00	<.900	<.900	<.900	1	<.900	<.600	<.600	<.600
0	<.400	<.500	<.100	<.100	<.400	<.400	<.400	<.400	<.400	<.400	<.500	<.400	<.400
0	<.0800	<.100	<.100	<.100	<.0700	<.0700	<.0700	<.0700	<.0700	<.0700	<.0900	<.0800	<.0800
20	15	13.2	22	19	3	3.1	2.7	2.7	3.3	3	2.4	3.1	3.2
.7	10.2	9.84	9.2	17.4	16.3	24.5	8.07	16.9	5.84	19.6	8	10.6	16.8
.7	<.500	<.600	<1.00	<1.00	<.400	<.400	<.400	<.400	<.400	0.5	<.900	<.800	<.800
.6	61.2	81	62.3	75.2	76.3	70.8	71.7	72.9	60.4	77.1	66.3	73.3	66.1

Appendix I. Trace elements (in µg/g dry weight) in Western meadowlark and European starling eggs collected from the Highland Uranium In Situ Mine irrigated area, and a reference site, Smith Ranch, Converse County, Wyoming.

	Study Area					Reference Site
	PRIWME01	PRIMLE02	PRISTE01	PRISTE03	PRISTE05	REFSTE01
Element	Western Meadowlark	Western Meadowlark	European Starling	European Starling	European Starling	European Starling
Al	<3.00	<3.00	<4.00	<3.00	14	<9.00
As	<.100	<.100	<.100	<.100	<.100	<.100
B	<1.00	2	<2.00	<1.00	2	<.600
Ba	9.89	8.45	16.7	14.1	8.9	2
Be	<.0200	<.0200	<.0200	<.0200	<.0200	<.0200
Cd	<.0200	<.0200	<.0200	<.0200	<.0200	<.0200
Cr	<.200	0.66	0.3	0.4	<.200	0.5
Cu	4.2	5.1	3.1	3.1	2.5	3.1
Fe	83.1	77	114	106	152	110
Hg	0.03	0.05	0.1	0.11	0.087	0.072
Mg	505	858	411	456	458	429
Mn	2.7	2.3	4.7	4.1	3.9	4
Mo	<.900	<.900	<1.00	<.900	<.900	0.8
Ni	<.400	<.400	<.400	<.400	<.400	<.400
Pb	<.0700	<.0700	<.0800	<.0700	<.0700	0.2
Se	18	28	6.2	7.1	7.9	2.7
Sr	14.7	49.6	14.8	16.6	13.3	9.1
V	<.400	0.5	<.500	0.6	<.400	<.800
Zn	65.9	54.1	59.8	50.7	53	48

nts (in µg/g dry weight) in pondweed collected from the Purge Storage Reservoir # 1, Highland Uranium In Situ Mine, ng.

VEG2	PRIAVEG3	PRIAVEG4	PRIAVEG5	PRIAVEG6	PRIAVEG7	PRIAVEG8
3490	5630	3310	8280	2370	4080	4140
2	2.4	1.7	3.4	3.7	3	3
133	96	218	26	191	236	99.4
198	203	215	293	206	194	236
0.1	0.1	0.09	0.27	0.09	0.1	0.1
0.1	0.1	0.1	0.1	0.1	0.1	0.1
3.4	4.7	3	10.1	2.9	6.3	6.2
3.2	4	5	5.5	3.3	5	4.4
1630	2510	2020	4280	1120	1620	2270
0.014	0.014	0.016	0.027	0.008	0.016	0.018
5170	5330	5500	4580	6110	5650	5100
1930	2300	1820	2020	2560	3060	2470
2	1	4	<.900	2	2	1
14.7	18.9	17.3	19.9	25.6	30.6	20.1
1.91	2.18	1.87	4.82	1.14	1.68	2.29
434	438	452	450	508	473	503
784	805	680	783	939	819	919
8.2	11	10	11	9.3	12	9.6
29.3	32	34.8	28.6	32.8	41.9	29.6

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Sturgeon Chub Current Range

(Missouri River Basin)

Map 4

Map compiled by USFWS, Ecological Services, Bismarck, ND, March 2001. All ranges shown are for representative purposes only and may not accurately represent the true range.