DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

50 CFR Part 17

[FF09E22000 FXES11130900000 201]

RIN 1018–BC98

Endangered and Threatened Wildlife and Plants; Removal of 23 Extinct Species From the Lists of Endangered and Threatened Wildlife and Plants

AGENCY: Fish and Wildlife Service, Interior.

ACTION: Proposed rule.

SUMMARY: We, the U.S. Fish and Wildlife Service (Service), propose to remove 23 species from the Federal Lists of Endangered and Threatened Wildlife and Plants due to extinction. This proposal is based on a review of the best available scientific and commercial information, which indicates that these species are no longer extant and, as such, no longer meet the definition of an endangered species or a threatened species under the Endangered Species Act of 1973, as amended (Act). We are seeking information and comments from the public regarding this proposed rule.

DATES: We will accept comments received or postmarked on or before November 29, 2021. Comments submitted electronically using the Federal eRulemaking Portal (see **ADDRESSES**, 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 **FOR FURTHER INFORMATION CONTACT** by November 15, 2021.

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

(1) *Electronically:* Go to the Federal eRulemaking Portal: *http://www.regulations.gov.* In the Search box, enter the appropriate docket number (see table under *Public Comments* in **SUPPLEMENTARY INFORMATION**). Then, click on the Search button. On the resulting page, in the Search panel on the left side of the screen, under the Document Type heading, check the Proposed Rule box to locate this document. You may submit a comment by clicking on ‘‘Comment Now!’’

(2) *By hard copy:* Submit by U.S. mail to: Public Comments Processing, Attn: [Insert appropriate docket number; see table under *Public Comments* in **SUPPLEMENTARY INFORMATION**], U.S. Fish and Wildlife Service, MS: PRB/3W, 5275 Leesburg Pike, Falls Church, VA 22041–3803.

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

FOR FURTHER INFORMATION CONTACT:

Species	Contact information
Bridled white-eye, Kauai akialoa, Kauai nukupuu, Kauai ‘o‘o (honeyeater), large Kauai thrush (kama), little Mariana fruit bat, Maui akepa, Maui nukupuu, Molokai creeper (kakawahie), *Phyllostegia glabra* var. *lanaiensis* (no common name), and po‘ouli (honeycreeper).	Earl Campbell, Field Supervisor, Pacific Islands Fish and Wildlife Office, 808–792–9400, 300 Ala Moana Boulevard, Suite 3–122, Honolulu, HI 96850.
Bachman's warbler	Thomas McCoy, Field Supervisor, South Carolina Field Office, 843–300–0431, 176 Croghan Spur, Charleston, SC 29407.
Flat pigtoe, southern acornshell, stirrupshell, and upland combshell	Stephen Ricks, Field Supervisor, Mississippi Field Office, 601–321–1122, 6578 Dogwood View Parkway, Suite A, Jackson, MS 39213.
Green blossom (pearly mussel), tubercled blossom (pearly mussel), turgid blossom (pearly mussel), and yellow blossom (pearly mussel).	Daniel Elbert, Field Supervisor, Tennessee Field Office, 931–528–6481, Interior Region 2—South Atlantic-Gulf (Tennessee), 446 Neal Street, Cookeville, TN 38506.
Ivory-billed woodpecker	Joe Ranson, Field Supervisor, Louisiana Field Office, 337–291–3113, 200 Dulles Dr., Lafayette, LA 70506.
San Marcos gambusia	Adam Zerrenner, Field Supervisor, Austin Ecological Services Field Office, 512–490–0057 (ext. 248), 10711 Burnet Rd., Suite 200, Austin, Texas 78758.
Scioto madtom	Patrice Ashfield, Field Supervisor, Ohio Ecological Services Field Office, 614–416–8993, 4625 Morse Road, Suite 104, Columbus, OH 43230.

Persons who use a telecommunications device for the deaf (TDD) may call the Federal Relay Service at 800–877–8339.

SUPPLEMENTARY INFORMATION:

Executive Summary

Why we need to publish a rule. Section 4 of the Act (16 U.S.C. 1533) and its implementing regulations in title 50 of the Code of Federal Regulations (50 CFR part 424) set forth the procedures for adding species to, removing species from, or reclassifying species on the Federal Lists of Endangered and Threatened Wildlife and Plants (List or Lists) in 50 CFR part 17. Under our regulations at 50 CFR 424.11(e)(1), a species shall be delisted if, after conducting a status review based on the best scientific and commercial data available, we determine that the species is extinct. The 23 species within this proposed rule are currently listed as endangered or threatened; we are proposing to delist them due to extinction. We can only delist a species by issuing a rule to do so.

What this document does. We propose to remove 23 species from the Lists due to extinction.

The basis for our action. We may determine that a species should be removed from the List because it no longer meets the definition of an endangered species or a threatened species, including whether the best available information indicates that a species is extinct.

Information Requested

Public Comments

We intend that any final rule resulting from this proposal will be based on the best available scientific and commercial data and will be as accurate and effective as possible. Therefore, we request comments or information from other concerned governmental agencies, Native American Tribes, the scientific community, industry, or any other interested parties concerning this proposed rule. Comments should be as specific as possible. We are specifically requesting comments on any additional information on whether these species are extant or extinct. This information can include:

(1) Any information that indicates whether the best available information supports a determination that one of the species is or is not extinct, including:

(a) Biological or ecological requirements as it relates to the detectability of the species, including but not limited to: Lifespan, life stage, maturation period, physical description and ease of identification, vocalization, and habitat requirements for feeding, breeding, and sheltering;

(b) Survey efforts past and current including information on how extensive the surveys were, the methodology used in the survey, and how effective were the methods used to detect the species (*i.e.,* were the surveys designed to effectively detect the species if it is present in the area?); or

(c) Last sighting of the species including a description of location of the sighting, the type of sighting (*e.g.,* visual or auditory), length of time since last detection, and the frequency of last sightings.

(2) Factors that may have resulted in the extinction 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.

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 a 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 **ADDRESSES**.

You may submit your comments or materials electronically, or view a detailed description of the basis for a species determination, on the internet at *http://www.regulations.gov* under the following docket numbers:

Species	Docket No.
Kauai akialoa	FWS–R1–ES–2020–0104
Kauai nukupuu	FWS–R1–ES–2020–0104
Kauai ʻoʻo (honeyeater)	FWS–R1–ES–2020–0104
Large Kauai thrush (kam'a)	FWS–R1–ES–2020–0104
Maui akepa	FWS–R1–ES–2020–0104
Maui nukupuu	FWS–R1–ES–2020–0104
Molokai creeper (kakawahie)	FWS–R1–ES–2020–0104
Poʻouli (honeycreeper)	FWS–R1–ES–2020–0104
Bridled white-eye	FWS–R1–ES–2020–0104
Little Mariana fruit bat	FWS–R1–ES–2020–0104
Phyllostegia glabra var. *lanaiensis* (no common name)	FWS–R1–ES–2020–0104
San Marcos gambusia	FWS–R2–ES–2020–0105
Scioto madtom	FWS–R3–ES–2020–0106
Flat pigtoe	FWS–R4–ES–2020–0107
Southern acornshell	FWS–R4–ES–2020–0107
Stirrupshell	FWS–R4–ES–2020–0107
Upland combshell	FWS–R4–ES–2020–0107
Green blossom (pearly mussel)	FWS–R4–ES–2020–0108
Tubercled blossom (pearly mussel)	FWS–R4–ES–2020–0108
Turgid blossom (pearly mussel)	FWS–R4–ES–2020–0108
Yellow blossom (pearly mussel)	FWS–R4–ES–2020–0108
Ivory-billed woodpecker	FWS–R4–ES–2020–0109
Bachman's warbler	FWS–R4–ES–2020–0110

Supporting information used to prepare the determinations, as well as comments and materials we receive, will be available for public inspection on *http://www.regulations.gov,* or by contacting the appropriate person, as specified under **FOR FURTHER INFORMATION CONTACT.**

If you submit information via *http://www.regulations.gov,* your entire submission—including any personal identifying information—will be posted on the website. 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.*

Because we will consider all comments and information we receive during the comment period, our final determinations may differ from this proposal. Based on the new information we receive (and any comments on that new information), we may conclude that the species should remain listed as endangered or threatened, or reclassify from threatened to endangered, instead of being delisted because new evidence indicates that it is not extinct.

Public Hearing

Section 4(b)(5) of the Act provides for a public hearing on this proposal, if requested. Requests must be received by the applicable date specified in **DATES**. Such requests must be sent to the address shown in **FOR FURTHER INFORMATION CONTACT**. We will schedule a public hearing on this proposal, if requested, and announce the date, time, and place of the hearing, as well as how to obtain reasonable accommodations, in the **Federal Register** and local newspapers at least 15 days before the hearing. For the immediate future, we will provide these public hearings using webinars that will be announced on the Service's website, in addition to the **Federal Register**. The use of these virtual public hearings is consistent with our regulations at 50 CFR 424.16(c)(3).

Peer Review

In accordance with our policy, ''Notice of Interagency Cooperative Policy for Peer Review in Endangered Species Act Activities,'' which was published on July 1, 1994 (59 FR 34270) and our August 22, 2016, Director's Memorandum ''Peer Review Process,'' we will seek, or have sought, the expert opinion of at least three appropriate and independent specialists regarding scientific data and interpretations contained in this proposed rule for each species or group of species. In certain cases, species will be grouped together for peer review based on similarities in biology or geographic occurrences. We will send copies of the five-year species status reviews to the peer reviewers immediately following publication in the **Federal Register**. We will ensure that the opinions of peer reviewers are objective and unbiased by following the guidelines set forth in the Director's Memo, which updates and clarifies Service policy on peer review (U.S. Fish and Wildlife Service 2016). The purpose of such review is to ensure that our decisions are based on scientifically sound data, assumptions, and analysis. Accordingly, our final decisions may differ from this proposal.

Background

Section 4(c) of the Act requires the Service to maintain and publish Lists of Endangered and Threatened Species. This includes delisting species that are extinct or presumed extinct based on the best scientific and commercial data available. The Service can decide to delist a species presumed extinct on its own initiative, as a result of a 5-year review under section 4(c)(2) of the Act, or because we are petitioned to delist due to extinction. Congress made clear that an integral part of the statutory framework is for the Service to make delisting decisions when appropriate and revise the Lists accordingly. For example, section 4(c)(1) of the Act requires the Service to revise the Lists to reflect recent determinations, designations, and revisions. Similarly, section 4(c)(2) requires the Service to review the lists at least every 5 years; determine, based on those reviews, whether any species should be delisted or reclassified; and, if so, apply the same standards and procedures as for listings under sections 4(a) and 4(b). Finally, to make a finding that a particular action is warranted but precluded, the Service must make two determinations: (1) That the immediate proposal and timely promulgation of a final regulation is precluded by pending proposals to determine whether any species is endangered or threatened; and (2) that expeditious progress is being made to add qualified species to either of the Lists and to remove species from the Lists (16 U.S.C. 1533(b)(3)(B)(iii)). Delisting species that will not benefit from the Act's protections because they are extinct allows us to allocate resources responsibly for on-the-ground conservation efforts, recovery planning, 5-year reviews, and other protections for species that are extant and will therefore benefit from those actions.

Regulatory and Analytical Framework

Section 4 of the Act (16 U.S.C. 1533) and its implementing regulations (50 CFR part 424) set forth the procedures for adding species to, removing species from, or reclassifying species on the Lists. Our regulations (50 CFR 424.11(e)) state that the Secretary shall delist a species if the Secretary finds that, after conducting a status review based on the best scientific and commercial data available:

(1) The species is extinct;
(2) The species does not meet the definition of an endangered species or a threatened species. In making such a determination, we consider the same five factors and apply the same standards set forth as for listing and reclassification; or
(3) The listed entity does not meet the statutory definition of a species.

In this proposed rule, we use the commonly understood biological definition of ''extinction'' as meaning that no living individuals of the species remain in existence. A determination of extinction will be informed by the best available information to indicate that no individuals of the species remain alive, either in the wild or captivity. This is in contrast to ''functional extinction,'' where individuals of the species remain alive but the species is no longer viable and/or no reproduction will occur (*e.g.,* any remaining females cannot reproduce, only males remain, etc.).

In our analyses, we attempted to minimize the possibility of either (1) prematurely determining that a species is extinct where individuals exist but remain undetected, or (2) assuming the species is extant when extinction has already occurred. Our determinations of whether the best available information indicates that a species is extinct included an analysis of the following criteria: Detectability of the species, adequacy of survey efforts, and time since last detection. All three criteria require taking into account applicable aspects of species' life history. Other lines of evidence may also support the determination and be included in our analysis.

In conducting our analyses of whether these species are extinct, we considered and thoroughly evaluated the best scientific and commercial data available. We reviewed the information available in our files, and other available published and unpublished information. These evaluations may include information from recognized experts; Federal, State, and Tribal governments; academic institutions; foreign governments; private entities; and other members of the public.

The 5-year reviews of these species contain more detailed biological information on each species. This supporting information can be found on the internet at *http://www.regulations.gov* under the appropriate docket number (see table under *Public Comments,* above). The following information summarizes the analyses for each of the species proposed for delisting by this rule.

Summary of Biological Status and Threats

Mammals

Little Mariana Fruit Bat (Pteropus tokudae)

I. Background

The little Mariana fruit bat (*Pteropus tokudae*) was listed as endangered on August 27, 1984 (49 FR 33881), and was included in the Recovery Plan for Mariana Fruit Bat (*Pteropus mariannus,* or fanihi in the Chamorro language) and the Little Mariana Fruit Bat (USFWS 1990). Last observed in 1968, the little Mariana fruit bat was ''among the most critically endangered species of wildlife under U.S. jurisdiction,'' as noted in the 1984 final listing rule (49 FR 33881, August 27, 1984, p. 49 FR 33882), which cited hunting and loss of habitat as the primary factors contributing to its rarity. Three 5-year status reviews have been completed; the 2009 (initiated on March 8, 2007; see 72 FR 10547) and 2015 (initiated on February 5, 2013; see 78 FR 8185) reviews did not recommend a change in status (USFWS 2009b, 2015). The 5-year status review completed in 2019 (initiated on May 7, 2018; see 83 FR 20088) recommended delisting due to extinction likely resulting from habitat loss, poaching, and predation by the brown tree snake (*Boiga irregularis*). This recommendation was based on a reassessment of all available information for the species, coupled with an evaluation of population trends and threats affecting the larger, extant Mariana fruit bat, which likely shares similar behavioral and biological traits and provides important context for the historical decline of the little Mariana fruit bat. (USFWS 2019).

The little Mariana fruit bat was first described from a male type specimen collected in August 1931 (Tate 1934, p. 1). Its original scientific name, *Pteropus tokudae,* remains current. Only three confirmed observations of the little Mariana fruit bat existed in the literature based on collections of three specimens: Two males in 1931 (Tate 1934, p. 3), and a female in 1968 (Perez 1972, p. 146), all on the island of Guam where it was presumably endemic. Despite the dearth of confirmed collections and observations, two relatively recent studies have confirmed the taxonomic validity of the little Mariana fruit bat, via morphology (Buden *et al.* 2013, entire) and genetics (Almeida *et al.* 2014, entire). A study of the physical morphology of several Micronesia *Pteropus* spp., including all three known little Mariana fruit bat specimens, concluded that the species was a distinct taxon (Buden *et al.* 2013, entire). Subsequently, genetic analysis of skin samples from 50 of the 63 described *Pteropus* species supported the Mariana little fruit bat's taxonomic distinctness (Almeida *et al.* 2014, entire).

The little Mariana fruit bat belonged to a primarily tropical group of bats in the Megachiroptera suborder characterized by relatively large size, frugivorous diet (fruit-eating), and lack of echolocation. Its genus, *Pteropus,* comprises 63 species, including many coastal species endemic to Pacific islands (Almeida *et al.* 2014, pp. 83–84). Given the homogeneity of life-history traits within the *Pteropus* genus, we expect that the little Mariana fruit bat exhibited similar behavior and life history to other members of the genus, including group roosting and foraging within forest habitat, lengthy care of few offspring, and slow population growth (USFWS 1990, p. 7; Wiles 1987, p. 154). Lifespan for the little Mariana fruit bat is unknown, but the Mariana fruit bat may survive for 30 years in captivity (USFWS 2020, unpaginated) and other bats within the genus live between 14 and 40 years. In the most recent 5-year review completed in 2019, we drew upon our knowledge of the larger and still extant Mariana fruit bat's biology to extrapolate a likely timeline and explanation for the little Mariana fruit bat's rarity, decline, and eventual extinction.

The earliest available scientific literature indicates that the little Mariana fruit bat was always likely rare, as suggested by written accounts of the species first recorded in the early 1900s (Baker 1948, p. 54; Perez 1972, pp. 145–146; Wiles 1987, p 154). In addition to possibly having been inherently rare, as suggested by the literature, a concurrent decline in the little Mariana fruit bat population likely occurred during the well-documented decrease in Mariana fruit bat abundance on Guam in the 1900s. In 1920, it was ''not an uncommon sight'' to see fruit bats flying over the forest during the daytime in Guam (Wiles 1987, p. 150). Just 10 years later (when the first two little Mariana fruit bat specimens were collected), fruit bats were uncommon on the island (Wiles 1987, p. 150), and were found mostly in northern Guam; introduced firearms may have been a contributing factor in their decline because they increased the efficiency of hunting (Wiles 1987, p. 150).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The little Mariana fruit bat was much smaller than the related Mariana fruit bat (Tate 1934, p. 2; Perez 1972, p. 146; Buden *et al.* 2013, pp. 109–110). Adult bats measured approximately 5.5 to 5.9 inches (in) (14 to 15.1 centimeters (cm)) in head-body length, with a wingspan of approximately 25.6 to 27.9 in (650 to 709 mm). The adults weighed approximately 5.36 ounces (152 grams). Although primarily dark brown in color, the little Mariana fruit bat showed some variation on the neck and head which could appear pale gold and grayish or yellowish-brown in color. Because of their small size (O'Shea and Bogan 2003, pp. 49, 254; USFWS 2009, p. 55), it is possible that adult little Mariana fruit bats were historically confused with juvenile fruit bats. Therefore, historical accounts of the species may have been underrepresented (Perez 1972, p. 143; Wiles 1987, p. 15).

The challenges of surveying for the Mariana fruit bat and most *Pteropus* spp. (including in theory, the little Mariana fruit bat) are numerous. Mariana fruit bats sleep during the day in canopy emergent trees, either solitarily or within colonial aggregations that may occur across several acres (O'Shea and Bogan 2003, p. 254; Utzurrum *et al.* 2003, p. 49; USFWS 2009, p. 269). The tropical islands where many tropical fruit bats (*Pteropus* spp.) are located have widely diverse and steeply topographical habitat, making surveys difficult. Additionally, most *Pteropus* spp. choose roost sites (both colonial and individual) that occur in locations difficult for people to reach, such as adjacent to steep cliffsides in remote forest areas (Wilson and Graham 1992, p. 65). The selection of roost sites in these areas is likely both a result of their evolved biology (for example to take advantage of updrafts for flight (Wilson and Graham 1992, p. 4)) and learned behavior to avoid poachers (USFWS 2009, pp. 24–25; Mildenstein and Johnson 2017, p. 36). To avoid triggering this avoidance behavior, surveyors must generally keep a distance of 164 feet (50 meters) and survey only downwind of roost sites (Mildenstein and Boland 2010, pp. 12–13; Mildenstein and Johnson 2017, pp. 55, 86). Additionally, *Pteropus* spp. typically sleep during the day and do not vocalize, and flying individuals may be easily counted twice due to their foraging patterns (Utuzurrum *et al.* 2003, p. 54).

Survey Effort

Historically, surveys to estimate colonial fruit bat numbers have generally involved two relatively simple and inexpensive methods, direct counts and station counts (or departure, or exit counts) (Utuzurrum *et al.* 2003, pp. 53–54). With direct counts, surveyors attempt to determine the number of bats in a roosting colony (or individual bats) at a single site during the day. Direct counts usually involve use of binoculars or a spotting scope, depending on the observation distance from the colony or individuals (Kunz *et al.* 1996; Eby *et al.* 1999; Garnett *et al.* 1999; Worthington *et al.* 2001 as cited in Mildenstein and Boland 2010, pp. 2–3). Conversely, surveyors conduct exit counts in the late afternoon to early evening when bats begin to depart from the roost site for evening foraging. Exit counts are typically conducted at locations with wide and unimpeded views of either areas known to contain colonies, or forested areas that would likely serve as roost sites for bats. Occasionally, surveyors may conduct both exit and direct counts by boat or by air with a helicopter. More recently, direct and exit count surveys involve use of computers and digital photography to aid the process (Mildenstein and Boland 2010, pp. 2–3).

By 1945, fruit bats were difficult to locate even in the northern half of Guam, where they were largely confined to forested cliff lines along the coasts (Baker 1948, p. 54). During surveys conducted between 1963 and 1968, the Guam Division of Aquatic and Wildlife Resources (DAWR) confirmed that bats were declining across much of Guam and were absent in the south. It was also during these same field studies that the third and last little Mariana fruit bat was collected in northern Guam in 1968 (Baker 1948, p. 146).

Increased survey efforts during the late 1970s and early 1980s reported no confirmed sightings of the little Mariana fruit bat (Wheeler and Aguon 1978, entire; Wheeler 1979, entire; Wiles 1987, entire; Wiles 1987, pp. 153–154). When the little Mariana fruit bat was listed as endangered (49 FR 33881; August 27, 1984), we noted that the species was on the verge of extinction and had not been verifiably observed after 1968. When we published a joint recovery plan for the little Mariana fruit bat and the Mariana fruit bat in 1990, we considered the little Mariana fruit bat already extinct based upon the available literature (USFWS 1990, p. 7).

During the 1990s, researchers recorded decreasing Mariana fruit bat numbers on Guam and increasing fatalities of immature bats. They hypothesized the decline was due to predation by the brown tree snake (Wiles *et al.* 1995, pp. 33–34, 39–42). With bat abundance continuing to decline in the 2000s, researchers now estimate the island's Mariana fruit bat population currently fluctuates between 15 and 45 individuals (Mildenstein and Johnson 2017, p. 24; USFWS 2017, p. 54). Even if the little Mariana fruit bat persisted at undetectable numbers for some time after its last confirmed collection in 1968, it is highly likely the little Mariana fruit bat experienced the same pattern of decline that we are now seeing in the Mariana fruit bat.

Time Since Last Detection

As stated above, the little Mariana fruit bat was last collected in northern Guam in 1968 (Baker 1948, p. 146). Intensive survey efforts conducted by Guam DAWR and other researchers in subsequent decades have failed to locate the species. Decades of monthly (and, later, annual) surveys for the related Mariana fruit bat by qualified personnel in northern Guam have failed to detect the little Mariana fruit bat (Wheeler and Aguon 1978, entire; Wheeler 1979, entire; Wiles 1987, entire; Wiles 1987, pp. 153–154; USFWS 1990, p. 7).

III. Analysis

Like the majority of bat species in the genus *Pteropus,* specific biological traits likely exacerbated the little Mariana fruit bat's susceptibility to human activities and natural events (Wilson and Graham 1992, pp. 1–8). For example, low fecundity in the genus due to late reproductive age and small broods (1 to 2 young annually) inhibits population rebound from catastrophic events such as typhoons, and from slow progression of habitat loss and hunting pressure that we know occurred over time. The tendency of *Pteropus* bats to roost together in sizeable groups or colonies in large trees rising above the surrounding canopy makes them easily detected by hunters (Wilson and Graham 1992, p. 4). Additionally, *Pteropus* bats show a strong tendency for roost site fidelity, often returning to the same roost tree year after year to raise their young (Wilson and Graham 1992, p. 4; Mildenstein and Johnson 2017, pp. 54, 68). This behavior likely allowed hunters and (later) poachers to easily locate and kill the little Mariana fruit bat and, with the introduction of firearms, kill them more efficiently (Wiles 1987, pp. 151, 154; USFWS 2009, pp. 24–25; Mildenstein and Johnston 2017, pp. 41–42). The vulnerability of the entire genus *Pteropus* is evidenced by the fact that 6 of the 62 species in this genus have become extinct in the last 150 years (including the little Mariana fruit bat). The International Union for Conservation of Nature (IUCN) categorizes an additional 37 species in this genus at risk of extinction (Almeida *et al.* 2014, p. 84).

In discussing survey results for the Mariana fruit bat in the late 1980s, experts wrote that the level of illegal poaching of bats on Guam remained extremely high, despite the establishment of several legal measures to protect the species beginning in 1966 (Wiles 1987, p. 154). They also wrote about the effects of brown tree snake predation on various fruit bats species (Savidge, 1987, entire; Wiles 1987, pp. 155–156). To date, there is only one documented instance of brown tree snake actually preying on the Mariana fruit bat; in that case, three young bats were found within the stomach of a snake (Wiles 1987, p. 155). However, immature *Pteropus* pups are particularly vulnerable to predators between approximately 3 weeks and 3 months of age. During this timeframe, the mother bats stop taking their young with them while they forage in the evenings, leaving them alone to wait at their roost tree (Wiles 1987, p. 155).

Only three specimens of little Mariana fruit bat have ever been collected, all on the island of Guam, and no other confirmed captures or observations of this species exist. Based on the earliest records, the species was already rare in the early 1900s. Therefore, since its discovery, the little Mariana fruit bat likely experienced greater susceptibility to a variety of factors because of its small population size. Predation by the brown tree snake, alteration and loss of habitat, increased hunting pressure, and possibly competition with the related Mariana fruit bat for the same resources under the increasingly challenging conditions contributed to the species' decreased ability to persist.

It is highly likely the brown tree snake, the primary threat thought to be the driver of multiple bird and reptile species extirpations and extinctions on Guam, has been present throughout the little Mariana fruit bat's range for at least the last half-century, and within the last northern refuge in northern Guam since at least the 1980s. Because of its life history and the challenges presented by its small population size, we conclude that the little Mariana fruit bat was extremely susceptible to predation by the brown tree snake.

IV. Conclusion

At the time of listing in 1984, hunting and loss of habitat were considered the primary threats to the little Mariana fruit bat. The best available information now indicates that the little Mariana fruit bat is extinct. The species appears to have been vulnerable to pervasive, rangewide threats including habitat loss, poaching, and predation by the brown tree snake. Since its last detection in 1968, qualified observers have conducted surveys and searches throughout the range of the little Mariana fruit bat but have not detected the species. Available information indicates that the species was not able to persist in the face of anthropogenic and environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

Birds

Bachman's Warbler (Vermivora bachmanii)

I. Background

The Bachman's warbler (*Vermivora bachmanii*) was listed on March 11, 1967 (32 FR 4001), as endangered under the Endangered Species Preservation Act of 1966, as a result of the loss of breeding and wintering habitat. Two 5-year reviews were completed for the species on February 9, 2007 (initiated on July 26, 2005; see 70 FR 43171), and May 6, 2015 (initiated on September 23, 2014; see 79 FR 56821). Both 5-year reviews recommended that if the species was not detected within the following 5 years, it would be appropriate to delist due to extinction.

The Bachman's warbler was first named in 1833 as *Sylvia bachmanii* based on a bird observed in a swamp near Charleston, South Carolina (AOU 1983, pp. 601–602). The Bachman's warbler was among the smallest warblers with a total length of 11.0 to 11.5 centimeters (cm) (4.3 to 4.5 inches (in)). The species was found in the southeastern portions of the United States from the south Atlantic and Gulf Coastal Plains, extending inland in floodplains of major rivers (eastern Texas, Louisiana, Arkansas, bootheel of Missouri, Alabama, Georgia, North and South Carolinas, Virginia, and flyovers in Florida). However, breeding was documented only in northeast Arkansas, southeast Missouri, southwest Kentucky, central Alabama, and southeast South Carolina. Bachman's warbler was a neotropical migrant; historically, the bulk of the species' population left the North American mainland each fall for Cuba and Isle of Pines (Dingle 1953, pp. 67–68, 72–73).

Available information indicates that migratory habitat preferences differed from winter and breeding habitat preferences in that the bird used or tolerated a wider range of conditions and vegetative associations during migration. Historical records indicate the Bachman's warbler typically nested in low, wet, forested areas containing variable amounts of water, but usually with some permanent water. While it is not definitively known, it is thought that they preferred small edges created by fire or storms with a dense understory of the cane species *Arundinaria gigantea* and palmettos. Nests were typically found in shrubs low to the ground from late March through June, and average known clutch size was 4.2 +/−0.7 (with a range of 3 to 5) (Hamel 2018, pp. 14–15). During the winter in Cuba, it was found in a wider variety of habitats across the island including forests, ranging from dry, semi-deciduous forests to wetlands, and even in forested urban spaces (Hamel 1995, p. 5). Life expectancy is unknown, but other warbler species live for 3 to 11 years (Klimkiewicz *et al.* 1983, pp. 292–293).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The Bachman's warbler was one of the smallest warblers with a total length of 11.0 to 11.5 cm. The bill was slender with a slight downward curve in both sexes and was a unique feature within the genus. The male was olive-green above with yellow forehead, lores, eye-ring, chin, and underparts; a black throat and crown; and dusky wings and tail. Males also had a yellow shoulder patch and bright rump. Generally, while similar, plumage of females was paler. Females lacked any black coloration and had olive green upperparts with yellow forehead and underparts. The eye-ring was whiter than in the males, and the crown was grayish. The dark patch on the throat was usually missing and the eye-ring was pale. Females had a buffy or bright yellowish forehead and a gray crown with no black; a whitish or white crissum; and less pronounced white spots on the tail (Hamel and Gauthreaux 1982, pp. 235–239; Hamel 1995, p. 2). Immature males resembled females. Males were easy to distinguish from other warblers. However, the drab coloration of the females and immature birds made positive identification difficult (Hamel and Gauthreaux 1982, p. 235). Additionally, females were much more difficult to identify because variability in plumage was greater. Immature females were also most likely to be confused with other similarly drab warblers. The song of the Bachman's warbler was a zeep or buzzy zip given by both sexes (Hamel 2020, Sounds and Vocal Behavior). This species may have been difficult to differentiate on call alone, as its call was somewhat reminiscent of the pulsating trill of the northern parula (*Parula americana*) (Curson *et al.* 1994, p. 95), and only two recordings exist from the 1950s (Hamel 2018, p. 32) to guide ornithologists on distinguishing it this way. Despite the fact that it could be mistaken for the northern parula, Bachman's warbler was of high interest to birders, and guides have been published specifically to aid in field identification (Hamel and Gauthreaux 1982, entire). As a result, substantial informal and formal effort has been expended searching for the bird and verifying potential sightings as outlined below (see ''Survey Effort'').

Survey Effort

Although Bachman's warbler was first described in 1833, it remained relatively unnoticed for roughly the next 50 years. Population estimates are qualitative in nature and range from rare to abundant (Service 1999, pp. 4–448). Populations were probably never large and were found in ''some numbers'' between 1890 and 1920, but afterwards populations appeared to be very low (Hamel 2018, pp. 16–18). For instance, several singing males were reported in Missouri and Arkansas in 1897 (Widmann 1897, p. 39), and Bachman's warbler was seen as a migrant along the lower Suwannee River in flocks of several species (Brewster and Chapman 1891, p. 127). The last confirmed nest was documented in 1937 (Curson *et al.* 1994, p. 96). A dramatic decline occurred sometime between the early 1900s and 1940 or 1950. Recognition of this decline resulted in the 1967 listing of the species (32 FR 4001; March 11, 1967) under the Endangered Species Preservation Act of 1966.

Between 1975 and 1979, an exhaustive search was conducted in South Carolina, Missouri, and Arkansas. No Bachman's warblers were located (Hamel 1995, p. 10). The last (though unconfirmed) sighting in Florida was from a single bird observed near Melbourne in 1977. In 1989, an extensive breeding season search was conducted on Tensas National Wildlife Refuge in Louisiana. Six possible Bachman's warbler observations occurred, but could not be documented sufficiently to meet acceptability criteria established for the study (Hamilton 1989, as cited in Service 2015, p. 4).

An experienced birder reported multiple, possible sightings of Bachman's warbler at Congaree National Park, South Carolina, in 2000 and 2001. These included hearing a male and seeing a female. In 2002, the National Park Service partnered with the Service and the Atlantic Coast Joint Venture to investigate these reports. Researchers searched over 3,900 acres of forest during 166 hours of observation in March and April; however, no Bachman's warbler sightings or vocalizations were confirmed. As noted previously, females and immature birds are difficult to positively identify. Males (when seen) are more easily distinguishable from other species. Researchers trying to verify the sightings traced several promising calls back to northern parulas and finally noted that they were confident the species would have been detected had it been present (Congaree National Park 2020, p. 3).

In several parts of the Bachman's warbler's range, relatively recent searches (since 2006) for ivory-billed woodpecker also prompted more activity in appropriate habitat for Bachman's warbler. Although much of the search period for ivory-billed woodpecker is during the winter, the searches usually continue until the end of April, when Bachman's warbler would be expected in the breeding range. Therefore, because Bachman's warbler habitat overlaps ivory-billed woodpecker habitat, the probability that Bachman's warbler would be detected, if present, has recently increased (Service 2015, pp. 5–6). Further, in general, substantial informal effort has been expended searching for Bachman's warbler because of its high interest among birders (Service 2015, p. 5). In spite of these efforts, Bachman's warbler has not been observed in the United States in more than three decades.

In Cuba, the species' historical wintering range, the last ornithologist to see the species noted that the species was observed twice in the 1960s in the Zapata Swamp: One sighting in the area of a modern-day hotel in Laguna del Tesoro and the other one in the Santo Tomas, Zanja de la Cocodrila area. Some later potential observations (*i.e.,* 1988) in the same areas were thought to be a female common yellowthroat (Navarro 2020, pers. comm.). A single bird was reported in Cuba in 1981 at Zapata Swamp (Garrido 1985, p. 997; Hamel 2018, p. 20). However, additional surveys in Cuba by Hamel and Garrido in 1987 through 1989 did not confirm additional birds (Navarro 2020, pers. comm.). There have been no sightings or bird surveys in recent years in Cuba, and all claimed sightings of Bachman's warbler from 1988 onwards have been rejected by the ornithological community (Navarro 2020, pers. comm.). Curson *et al.* (1994, p. 96) considers all sightings from 1978 through 1988 in Cuba as unconfirmed.

Time Since Last Detection

After 1962, reports of the Bachman's warbler in the United States have not been officially accepted, documented observations (Chamberlain 2003, p. 5). Researchers have been thorough and cautious in verification of potential sightings, and many of the more recent ones could not be definitively verified. Bachman's warbler records from 1877–2001 in North America are characterized as either relying on physical evidence or on independent expert opinion, or as controversial sightings (Elphick *et al.* 2010, pp. 8, 10). In Cuba, no records have been verified since the 1980s (Navarro 2020, pers. comm.).

Other Considerations Applicable to the Species' Status

At breeding grounds, the loss of habitat from clearing of large tracts of palustrine (*i.e.,* having trees, shrubs, or emergent vegetation) wetland beginning in the 1800s was a major factor in the decline of the Bachman's warbler. Most of the palustrine habitat in the Mississippi Valley (and large proportions in Florida) was historically converted to agriculture or affected by other human activities (Fretwell *et al.* 1996, pp. 8, 10, 124, 246). Often the higher, drier portions of land that the Bachman's warbler required for breeding were the first to be cleared because they were more accessible and least prone to flooding (Hamel 1995, pp. 5, 11; Service 2015, p. 4). During World Wars I and II, many of the remaining large tracts of old growth bottomland forest were cut, and the timber was used to support the war effort (Jackson 2020, Conservation and Management, p. 2). At the wintering grounds of Cuba, extensive loss of primary forest wintering habitat occurred due to the clearing of large areas of the lowlands for sugarcane production (Hamel 2018, p. 24). Hurricanes also may have caused extensive damage to habitat and direct loss of overwintering Bachman's warblers. Five hurricanes occurred between November 1932 and October 1935. Two storms struck western Cuba in October 1933, and the November 1932 hurricane is considered one of the most destructive ever recorded. These hurricanes, occurring when Bachman's warblers would have been present at their wintering grounds in Cuba, may have resulted in large losses of the birds (Hamel 2018, p. 19).

III. Analysis

As early as 1953, Bachman's warbler was reported as one of the rarest songbirds in North America (Dingle 1953, p. 67). The species may have gone extinct in North America by 1967 (Elphick *et al.* 2010, p. 619). Despite extensive efforts to document presence of the species, no new observations of the species have been verified in the United States or Cuba in several decades (Elphick *et al.* 2010, supplement; Navarro 2020, pers. comm.). Given the likely lifespan of the species, it has not been observed in several generations.

IV. Conclusion

As far back as 1977, Bachman's warbler has been described as being on the verge of extinction (Hooper and Hamel 1977, p. 373) and the rarest songbird native to the United States (Service 1999, pp. 4–445). The species has not been seen in the United States or Cuba since the 1980s, despite extensive efforts to locate it and verify potential sightings. Therefore, we conclude that the best available scientific and commercial information indicates that the species is extinct.

Bridled White-eye (Zosterops conspicillatus conspicillatus)

I. Background

The bridled white-eye (*Zosterops conspicillatus conspicillatus,* or Nossa in the Chamorro language), was listed as endangered in 1984 (49 FR 33881; August 27, 1984), and was included in the Recovery Plan for the Native Forest Birds of Guam and Rota of the Commonwealth of the Northern Mariana Islands (USFWS 1990, entire). The species was last observed in 1983, and the 1984 final listing rule for the bridled white-eye noted that the species ''may be the most critically endangered bird under U.S. jurisdiction'' (49 FR 33881, August 27, 1984, p. 49 FR 33883) and cited disease and predation by nonnative predators, including the brown tree snake (*Boiga irregularis*), as the likely factors contributing to its rarity (49 FR 33881, August 27, 1984, p. 49 FR 33884). Three 5-year status reviews were completed for the bridled white-eye; the 2009 (initiated on March 8, 2007; see 72 FR 10547) and 2015 (initiated on March 6, 2012; see 77 FR 13248) reviews did not recommend a change in status (USFWS 2009a, 2015). After reevaluation of all available information, the 5-year status review completed in 2019 (initiated on May 7, 2018; see 83 FR 20088) recommended delisting due to extinction, based on continued lack of detections and the pervasive rangewide threat posed by the brown tree snake (USFWS 2019, p. 10).

At the time of listing, the bridled white-eye on Guam was classified as one subspecies within a complex of bridled white-eye (*Zosterops conspiculatus*) populations found in the Mariana Islands. The most recent taxonomic work (Slikas *et al.* 2000, p. 360) continued to classify the Guam subspecies within the same species as the bridled white-eye populations currently found on Saipan, Tinian, and Aguiguan in the Commonwealth of the Northern Mariana Islands (*Z. c. saypani*) but considered the Rota population (*Z. rotensis;* now separately listed as endangered under the Act) to be a distinct species.

Endemic only to Guam, within the Mariana Islands, the bridled white-eye was a small (0.33 ounce or 9.3 grams), green and yellow, warbler-like forest bird with a characteristic white orbital ring around each eye (Jenkins 1983, p. 48). The available information about the life history of the species is sparse, based on a few early accounts in the literature (Seale 1901, pp. 58–59; Stophet 1946, p. 540; Marshall 1949, p. 219; Baker 1951, pp. 317–318; Jenkins 1983, pp. 48–49). Nonterritorial and often observed in small flocks, the species was a canopy-feeding insectivore that gleaned small insects from the twigs and branches of trees and shrubs (Jenkins 1983, p. 49). Although only minimal information exists about the bridled white-eye's nesting habits and young, observations of nests during several different months suggests the species bred year-round (Marshall 1949, p. 219; Jenkins 1983, p. 49). No information is available regarding longevity of the bridled white-eye, but lifespans in the wild for other white-eyes in the same genus range between 5 and 13 years (Animal Diversity Web 2020; The Animal Aging and Longevity Database 2020; *WorldLifeExpectancy.com 2020*).

The bridled white-eye was reported to be one of the more common Guam bird species between the early 1900s and the 1930s (Jenkins 1983, p. 5). However, reports from the mid- to late-1940s indicated the species had perhaps become restricted to certain areas on Guam (Baker 1951, p. 319; Jenkins 1983, p. 50). By the early- to mid-1970s, the bridled white-eye was found only in the forests in the very northern portion of Guam (Wiles *et al.* 2003, p. 1353). It was considered rare by 1979, causing experts to conclude that the species was nearing extinction (Jenkins 1983, p. 50).

By 1981, the bridled white-eye was known to inhabit only a single 395-acre (160-hectare) limestone bench known as Pajon Basin in a limestone forest at Ritidian Point, an area that later became the Guam National Wildlife Refuge. Nestled at the base of towering limestone cliffs of about 426 feet (130 meters), the site was bordered by adjoining tracts of forest on three sides, and ocean on the northern side (Wiles *et al.* 2003, p. 1353). Pajon Basin was also the final refuge for many of Guam's native forest bird species and was the last place where 10 of Guam's forest bird species were still observed together in one locality at historical densities (Savidge 1987, p. 661; Wiles *et al.* 2003, p. 1353).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The bridled white-eye has been described as active and occurred in small flocks of 3 to 12 individuals (Jenkins 1983, p. 48). Although apparently not as vocal as its related subspecies on the other Mariana Islands, the bridled white-eye was observed singing and typically vocalized with ''chipping calls'' while flocking, less so during foraging (Jenkins 1983, p. 48). Although perhaps not correctly identified as a ''secretive'' or ''cryptic'' species (Amidon in litt. 2000, pp. 14–15), the detectability of the related Rota bridled white-eye (*Zosterops rotensis*) is greatest during surveys when it is close to the observer, relative to other species of birds that are detected at further distances. While we are unaware of surveys for the bridled white-eye using alternative methodologies specific for rare or secretive bird species, we conclude there is still sufficient evidence of extinction based upon the large body of literature confirming the impacts of the brown tree snake on Guam (see discussion below under ''III. Analysis'').

Survey Effort

Variable circular plot (VCP) studies are surveys conducted at pre-established stations along transects. Surveyor counts all birds seen and heard during an 8-minute count period and estimates the distance from the count station to each bird seen or heard. From this information, an estimate of the number of birds in a surveyed area is determined and the confidence interval for the estimate is derived. During a multi-year VCP study at Pajon Basin consisting of annual surveys between 1981 and 1987, observations of the bridled white-eye drastically declined in just the first 3 years of the study. In 1981, 54 birds were observed, and in 1982, 49 birds were documented, including the last observation of a family group (with a fledging) of the species. One year later, during the 1983 survey, only a single individual bridled white-eye was sighted. Between 1984 and 1987, researchers failed to detect the species within this same 300-acre (121-hectare) site (Beck 1984, pp. 148–149).

Between the mid- and late-1980s, experts had already begun to hypothesize that the bridled white-eye had become extinct (Jenkins 1983, p. 50; Savidge 1987, p. 661). Although human access has become more restricted within portions of Andersen Air Force Base since 1983, the Guam DAWR has, to date, continued annual roadside counts across the island as well as formal transect surveys in northern Guam in areas previously inhabited by the bridled white-eye. The species remains undetected since the last observation in Pajon Basin in 1983 (Wiles 2018, pers. comm.; Quitugua 2018, pers. comm.; Aguon 2018, pers. comm.).

Time Since Last Detection

Researchers failed to observe the species at the Pajon Basin during the annual surveys between 1984 and 1987, and during subsequent intermittent avian surveys in northern Guam in areas where this species would likely occur (Savidge 1987, p. 661; Wiles *et al.* 1995, p. 38; Wiles *et al.* 2003, entire).

III. Analysis

The brown tree snake is estimated to be responsible for the extinction, extirpation, or decline of 2 bat species, 4 reptiles, and 13 of Guam's 22 (59 percent) native bird species, including all of the native forest bird species with the exception of the Micronesian starling (*Aplonis opaca*) (Wiles *et al.* 2003, p. 1358; Rodda and Savidge 2007, p. 307). The most comprehensive study of the decline (Wiles *et al.* 2003, entire) indicated that 22 bird species were severely impacted by the brown tree snake.

The study also found that in areas newly invaded by the snake, observed declines of avian species were greater than or equal to 90 percent and occurred rapidly, with the average duration just 8.9 years. The study also examined traits of the birds that made them more or less susceptible to predation by the brown tree snake, and determined that the ability and tendency to nest and roost in locations where snakes were less common (*e.g.,* cave walls) correlated with greater likelihood of coexistence with the snake. Large clutch size and large body size correlated with a species' greater persistence, although large body size appeared to only delay, but not prevent, extirpation. Measuring a mere 0.33 ounces (9.3 grams), the bridled white-eye was relatively small in size, and its nests were located in areas accessible to brown tree snakes (Baker 1951, pp. 316–317; Jenkins 1983, pp. 49–50).

We used a recent analytical tool that assesses information on threats to infer species extinction based on an evaluation of whether identified threats are sufficiently severe and prolonged to cause local extinction, as well as sufficiently extensive in geographic scope to eliminate all occurrences (Keith *et al.* 2017, p. 320). Applying this analytical approach to the bridled white-eye, we examined years of research and dozens of scientific publications and reports that indicate that the effects of predation by the brown tree snake have been sufficiently severe, prolonged, and extensive in geographic scope to cause widespread range contraction, extirpation, and extinction for several birds and other species. Based on this analysis, we conclude that the bridled white-eye is extinct and brown tree snake predation was the primary causal agent.

IV. Conclusion

At the time of its listing in 1984, disease and predation by nonnative predators, including the brown tree snake, were considered the primary threats to the bridled white-eye. The best available information now indicates that the bridled white-eye is extinct. The species appears to have been vulnerable to the pervasive, rangewide threat of predation from the brown tree snake. Since its last detection in 1983, qualified observers have conducted surveys and searches throughout the range of the bridled white-eye and have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

Ivory-Billed Woodpecker (Campephilus principalis)

I. Background

The ivory-billed woodpecker (*Campephilus principalis*) was first described by Mark Catesby in 1731 (Tanner 1942, p. xv), under a different taxonomic nomenclature. It was the largest woodpecker in the United States and the second largest in North America with an overall length of approximately 48–51 centimeters (cm) (18–20 inches), an estimated wingspan of 76–80 cm (29–31 inches), and a weight of 454–567 grams (g) (16–20 ounces); however, data from live birds are lacking, so these estimates were based on observations by ornithologists from the late 19th century who collected specimens (Service 2010, pp. 1–2).

The ivory-billed woodpecker was listed as endangered throughout its range on March 11, 1967 (32 FR 4001) under the Endangered Species Preservation Act of 1966. Although no threats were identified at the time of listing, land clearing and timber harvesting were known at the time as threats acting on the species. A status review was announced on April 10, 1985 (50 FR 14123) to determine if the species was extinct and should therefore be proposed for delisting. We did not receive any confirmed reports of live birds as a result of that review. In 1986, we funded a large-scale survey that included coverage of potential sites throughout the species' historical range (Jackson 1989, p. 74; Jackson 2006, p. 1–2, USFWS 2010, p. 69). The study also included soliciting requests for new sightings and investigating those reports for validity, as well as researching historical sources (Jackson 1989, p. 74). No conclusive evidence of ivory-billed woodpeckers was obtained during that study.

Another status review was announced on November 6, 1991 (56 FR 56882) for all species (foreign and domestic listings) listed before 1991. In this review, the status of many species was simultaneously evaluated with no in-depth assessment of the five factors or threats as they pertain to the individual species. The document stated that the Service was seeking any new or additional information reflecting the necessity of a change in the status of the species under review. The document indicated that if significant data were available warranting a change in a species' classification, the Service would propose a rule to modify the species' status. No change in the bird's listing classification was found to be warranted. Each year, the Service reviews and updates listed species information for inclusion in the required Recovery Report to Congress. While considerable effort was placed on confirming reported sightings after 2004 (details provided below), no further sightings occurred. By 2013, the ornithological community determined that these sightings could not be confirmed. Since 2013, our annual recovery data call included status recommendations such as ''presumed extinct'' for the ivory-billed woodpecker.

A 5-year review was most recently announced on May 7, 2018 (83 FR 20092), with a 60-day public comment period ending July 6, 2018. During the public comment period, the Service received and considered four public comments describing reported, but not verifiable, encounters as well as indications that the inability to conclusively document existence does not mean that the species is extinct (Trahan 2020, pers. comm.). The Service also reviewed a variety of additional resources, including published and unpublished scientific information provided by other Service offices, State wildlife agencies, stakeholders, and other partners. Specific sources included the final rule listing this species under the Act (32 FR 4001; March 11, 1967); the recovery plan (Service 2010, entire); peer-reviewed scientific publications; unpublished field observations by Federal, State, and other experienced biologists; unpublished studies and survey reports; and notes and communications from other qualified individuals. The 5-year review was also sent to four independent peer reviewers; one responded with comments. This 5-year review was finalized on June 3, 2019, and recommended that the ivory-billed woodpecker be delisted due to extinction (USFWS 2019, entire).

Much of what we know about the ivory-billed woodpecker comes from research in Louisiana during the late 1930s (Service 2010, pp. xv, vii, 10–22, 67). Suitable habitat for the ivory-billed woodpecker is thought to be extensive forested areas with old-growth characteristics and a naturally high volume of dead and dying wood, particularly in virgin bottomland hardwoods that may sustain the species between disturbance events (*e.g.,* fires, storms, or other events expected to kill or stress trees) (Tanner 1942, pp. 46–47, 52). The home range for the ivory-billed woodpecker is thought to have been fairly large due to their ability to fly long distances, up to at least several kilometers a day between favored roost sites and feeding areas. The estimated ivory-billed woodpecker density historically ranged from one breeding pair per 6.25 square miles to one breeding pair per 17 square miles (Tanner 1942, p. 32).

Breeding was thought to occur between January and April (Tanner 1942, pp. 95–96). Clutch size reportedly ranged from 1 to 5 eggs with an estimated incubation period of approximately 20 days (Service 2010, p. 11). Both sexes of ivory-billed woodpecker incubated the eggs as well as fed the young for a period of about 5 weeks until the young fledged (Tanner 1942, pp. 101, 104). The young may have been fed by the parents for an additional 2 months and roosted near and foraged with the parents into the next breeding season. Dead or dying portions of live trees, and sometimes dead trees, may have been excavated for nest cavities. These cavities ranged from 4.6 meters (m) (15.1 feet (ft)) to over 21 m (69 ft) up a nest tree, although rarely below 9 m (29.5 ft) from a tree's base (Service 2010, p. 11). Ivory-billed woodpeckers not only used nest cavities but excavated roost cavities as well, which are similar in appearance to nest cavities. Pairs or group members were found to roost in trees near each other, and they also were reported to leave the roost after sunrise (Tanner 1942, pp. 57–59). The roosting area is known to have been the center of activity for ivory-billed woodpeckers; however, insect abundance (*i.e.,* food availability) was thought to be important to distribution as well (Tanner 1942, pp. 33–36, 46, 52). Although it is not known for certain, lifespan for the species was estimated to be in excess of 10 years (USFWS 2020, p. 24).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The ivory-billed woodpecker had a black and white plumage with a white chisel-tipped beak, yellow eyes, and a pointed crest. It was sexually dimorphic, with the sexes exhibiting different characteristics (*i.e.,* sizes, coloring, etc.). Females had a solid black crest, and males were red from the nape to the top of the crest with an outline of black on the front of the crest (Service 2010, p. 1). This large woodpecker produced distinctive sounds and had distinctive markings (*e.g.,* large white patch on the wing that can be seen from long distances (Tanner 1942, p. 1)), indicating a certain degree of detectability during surveys, if present.

Survey Effort

The last commonly agreed-upon sighting of the species was on the Singer Tract in the Tensas River region of northeast Louisiana in April of 1944 (Service 2019, p. 9). Since this sighting, the most compelling evidence of the existence of the ivory-billed woodpecker was in 2004 in Arkansas (Fitzpatrick *et al.* 2005, pp. 1460–1462). From 2004 to 2005, within the same area of Bayou DeView, located in the

Cache River National Wildlife Refuge (NWR) in Arkansas, observers reported sightings, audio recordings, and a video interpreted to be an ivory-billed woodpecker (Service 2010, p. 13). The original 2004 encounter as well as the other reports and video from Arkansas spurred an extensive search effort in the area that was led by the Cornell Laboratory of Ornithology and the Arkansas Nature Conservancy beginning in 2005. Multiple approaches were used, including visual methods, aural methods, and playback methods (alone and in combination), as well as helicopter surveys. However, after completing analysis of detection probabilities associated with all of the methods, researchers noted few, if any, ivory-billed woodpeckers could have remained undetected in the Big Woods of Arkansas during the period from 2005 to 2009 (Rohrbaugh and Lammertink 2016, p. 40). Further, although the bird in the video was first interpreted as an ivory-billed woodpecker, there is dispute among the ornithological community as to whether it was an actual ivory-billed woodpecker or instead a pileated woodpecker (*Dryocopus pileatus*). No conclusive videos gathered since then that confirm the persistence of the ivory-billed woodpecker. After additional extensive analysis of the recordings, it was determined that these recordings do not constitute evidence of the presence of ivory-billed woodpeckers (Charif *et al.* 2005, p. 1489; Fitzpatrick *et al.* 2005, p. 1462; Jackson 2006, p. 3).

Since the reported ivory-billed woodpecker in 2004/2005 at the Cache River NWR, a survey design was developed and implemented during search efforts throughout the species' historical range. Many State, Federal, and private partners (*e.g.,* State wildlife agencies, the Service, and the Cornell Laboratory of Ornithology) collaborated over a 5-year period to conduct extensive searches for evidence of the species' presence within the historical range; however, no individuals were reliably located, and no conclusive evidence confirmed the species' persistence (Service 2010, pp. V, VII, 2–9, 75–89). Since the 5-year survey effort was completed, other survey efforts based on sightings and vocalizations reported by wildlife professionals and other individuals have continued throughout the range through present day. These efforts include:

- *2005–2013:* Pearl River swamp, Louisiana and Choctawhatchee River swamp, Florida—Approximately 1,500 hours were spent surveying these two swamps with a kayak and video cameras. Three video clips were produced from both areas; however, the blurred images are inconclusive as to whether they are ivory-billed woodpeckers or not (Collins 2017, entire; Donahue 2017, p. 2).
- *2007–2011:* 30 additional areas in the southeastern United States (Pascagoula Basin of Mississippi, Mobile Basin of Alabama, Congaree and Coastal Basins of South Carolina, Apalachicola Basin of north Florida, and Everglades/Big Cypress Complex of south Florida) were surveyed with no presence of ivory-billed woodpeckers found (Lammertink and Rohrbaugh 2016, p. 7).
- *2011:* White River NWR, Arkansas—Searches were completed a year and a half after a tornado; no evidence of ivory-billed woodpecker presence was observed, further adding to negative outcome of the 2005–2009 search efforts in this NWR (Lammertink and Rohrbaugh 2016, p. 7).
- *2011:* Avoyelles Parish, Louisiana—Survey on private property and Pomme de Terre Wildlife Management Area (WMA). No observations of ivory-billed woodpeckers were made (Lammertink and Rohrbaugh 2016, p. 7).
- *2011:* Lee River State Natural Area, South Carolina—No evidence of ivory-billed woodpecker presence was found during surveys (Lammertink and Rohrbaugh 2016, p. 7).
- *2009–present:* Louisiana—A search group, Project Coyote, was founded to search for ivory-billed woodpeckers in Louisiana; no evidence has been offered that constitutes undeniable confirmation that the species persists (Michaels 2018, p. 79).
- *2016:* Cuba—An expedition to Cuba was initiated in search of the ivory-billed woodpecker; no presence found (McClelland 2016, pp. 13–15).

Although there have been many sightings reported over the years since the last unrefuted sighting in 1944, there is much debate over the validity of these reports. Furthermore, there is no objective evidence (*e.g.,* clear photographs, feathers of demonstrated recent origin, specimens, etc.) of the continued existence of the species.

Additionally, researchers analyzed the temporal pattern of the collection dates of museum specimens from 1853 to 1932 throughout the historical range to estimate the probability of the persistence of the species into the 21st century, as well as the probability that the species would be found at survey sites with continued efforts. The probability of persistence in a 2011 analysis was less than 0.000064, and this analysis estimated the probable extinction date to be between 1960 and 1980 (Gotelli *et al.* 2011, entire). While differing in assumptions, treatment of data, and statistical methods used, other analyses had qualitatively similar conclusions (*e.g.,* Roberts *et al.* 2009, entire; Solow *et al.* 2011, entire).

Time Since Last Detection

The last unrefuted sighting of the ivory-billed woodpecker occurred in April 1944 on the Singer Tract in the Tensas River region of northeast Louisiana (Service 2015, p. 9).

III. Analysis

The decline of mature forested habitat with a high percentage of recently dead or dying trees and widespread collection of the species likely led to the extirpation of the population sometime after the 1940s. Although there have been potential sightings reported over the years since the last agreed-upon sighting in 1944, there is much debate over the validity of these reports. Furthermore, there is no objective evidence (*e.g.,* clear photographs, feathers of demonstrated recent origin, specimens, etc.) of the continued existence of the species despite extensive searches. Given the likely lifespan of the species, this means it has not been indisputably observed in more than seven generations.

IV. Conclusion

The ivory-billed woodpecker has not been definitively sighted since 1944, despite decades of extensive survey effort. The loss of mature forest habitat and widespread collection of the species likely led to its extirpation in the 1940s or soon thereafter. Therefore, we conclude that the best available scientific and commercial information indicates that the species is extinct.

Kauai akialoa (Akialoa stejnegeri)

I. Background

Kauai akialoa (*Akialoa stejnegeri;* listed as *Hemignathus stejnegeri*), a Hawaiian honeycreeper, was listed as endangered on March 11, 1967 (32 FR 4001). It was included in the Kauai Forest Birds Recovery Plan (USFWS 1983), and the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006, p. 2–86). At the time of listing, we considered Kauai akialoa to have very low population numbers and to be threatened by habitat loss, avian disease, and predation by rats (*Rattus* spp.). The last confirmed observation of the species was in 1965, although there was an unconfirmed sighting in 1969 (Reynolds and Snetsinger 2001, p. 142). Two 5-year status reviews have been completed, in 2009 (initiated on July 6, 2005; see 70 FR 38972) and 2018 (initiated on February 13, 2015; see 80

FR 8100). The 2009 review did not recommend a change in status, though there was some information indicating the species was already extinct. The 5-year status review completed in 2019 recommended delisting due to extinction based on consideration of additional information about the biological status of the species, included in the discussion below (USFWS 2019, pp. 5, 10).

The life history of Kauai akialoa is poorly known and based mainly on observations from the end of the 19th century (USFWS 2006, p. 2–86). There is no information on the lifespan of the Kauai akialoa nor its threats when it was extant. The species was widespread on Kauai and occupied all forest types above 656 feet (200 meters) elevation (Perkins 1903, pp. 369, 422, 426). Its historical range included nearly all Kauai forests visited by naturalists at the end of the 19th century. After a gap of many decades, the species was seen again in the 1960s, when one specimen was collected (Richardson and Bowles 1964, p. 30). It has not been seen since, despite efforts by ornithologists (Conant *et al.* 1998, p. 15) and birders, and intensive survey efforts by wildlife biologists spanning 1968 to 2018 (USFWS 1983, p. 2; Hawaii Department of Land and Natural Resources unpubl. data; Reynolds and Snetsinger 2001, entire; Crampton *et al.* 2017 entire; Crampton 2018, pers. comm.).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The Kauai akialoa was a large (6.7 to 7.5 inches, or 17 to 19 centimeters, total length), short-tailed Hawaiian honeycreeper with a very long, thin, curved bill, the longest bill of any historically known Hawaiian passerine. The plumage of both sexes was olive-green; males were more brightly colored, were slightly larger, and had a somewhat longer bill (USFWS 2006, p. 2–86). The Kauai akialoa's relatively large size and distinctive bill suggest that if it were extant, it would be detectable by sight and recognized.

Survey Effort

A comprehensive survey of Hawaiian forest birds was initiated in the 1970s using the VCP method (Scott *et al.* 1986, entire). VCP surveys in Hawaii are conducted at pre-established stations along transects. The surveyor counts all birds seen and heard during an 8-minute count period and estimates the distance from the count station to each bird seen or heard. From this information, an estimate of the number of birds in area surveyed is determined and the confidence interval for this estimate derived. VCP surveys have been the primary method used to count birds in Hawaii; however, it is not appropriate for all species and provides poor estimates for extremely rare birds (Camp *et al.* 2009, p. 92). In recognition of this problem, the Rare Bird Search (RBS) was undertaken from 1994 to 1996, to update the status and distribution of 13 ''missing'' Hawaiian forest birds (Reynolds and Snetsinger 2001, pp. 134–137). The RBS was designed to improve efficiency in the search for extremely rare species, using the method of continuous observation during 20- to 30-minute timed searches in areas where target species were known to have occurred historically, in conjunction with audio playback of species vocalizations (when available). Several recent surveys and searches, including the RBS, have been unsuccessful in detecting Kauai akialoa despite intensive survey efforts by wildlife biologists from 1968 to 1973, and in 1981, 1989, 1993, 1994, 2000, 2005, and 2011 to 2018 (Hawaii Department of Land and Natural Resources unpubl. data; Reynolds and Snetsinger 2001, entire; Crampton *et al.* 2017, entire; Crampton 2018 pers. comm.). An unconfirmed 1969 report may have been the last sighting of Kauai akialoa (Conant *et al.* 1998, p. 15). Kauai akialoa has been presumed likely extinct for some time (Reynolds and Snetsinger 2001, p. 142).

In addition, extensive time has been spent by qualified observers in the historical range of the Kauai akialoa searching for the small Kauai thrush (*Myadestes palmeri*), akekee (*Loxops caeruleirostris*), and Kauai creeper (*Oreomystis bairdi*). Hawaii Forest Bird Surveys (HFBS) were conducted in 1981, 1989, 1994, 2000, 2005, 2007, 2008, 2012, and 2018 (Paxton *et al.* 2016, entire). The Kauai Forest Bird Recovery Project (KFBRP) conducted occupancy surveys for the small Kauai thrush in Kokee State Park, Hono O NaPali Natural Area Reserve, Na Pali Kona Forest Reserve, and Alakai Wilderness Preserve, from 2011 to 2013 (Crampton *et al.* 2017, entire), and spent over 1,500 person-hours per year from 2015 to 2018 searching for Kauai creeper and akekee nests. During the HFBS in 2012 and 2018, occupancy surveys and nest searches did not yield any new detections of Kauai akialoa. The KFBRP conducted mist-netting in various locations within the historical range for Kauai akialoa from 2006 through 2009, and from 2011 through 2018, and no Kauai akialoa were caught or encountered (Crampton 2018, pers. comm.).

Time Since Last Detection

Another approach used to determine whether extremely rare species are likely extinct or potentially still extant is to calculate the probability of a species' extinction based on time (years) since the species was last observed (Elphick *et al.* 2010, p. 620). This approach, when applied to extremely rare species, has the drawback that an incorrect assignment of species extinction may occur due to inadequate survey effort and/or insufficient time by qualified observers spent in the area where the species could still potentially exist. Using 1969 as the last credible sighting of Kauai akialoa, the authors' estimated date for the species' extinction is 1973, with 95 percent confidence that the species was extinct by 1984.

III. Analysis

The various bird species in the subfamily Drepanidinae (also known as the Hawaiian honeycreepers), which includes Kauai akialoa, are highly susceptible to introduced avian disease. They are particularly susceptible to avian malaria (*Plasmodium relictum*), which results in high rates of mortality. At elevations below approximately 4,500 feet (1,372 meters) in Hawaii, the key factor driving disease epizootics (outbreaks) of pox virus (*Avipoxvirus*) and avian malaria is the seasonal and altitudinal distribution and density of the primary vector of these diseases, *Culex quinquefasciatus* (Atkinson and Lapointe 2009a, pp. 237–238, 245–246).

A recent analytic tool was consulted using information on threats to infer species extinction based on an evaluation of whether identified threats are sufficiently severe and prolonged to cause local extinction, and sufficiently extensive in geographic scope to eliminate all occurrences (Keith *et al.* 2017, p. 320). The disappearance of many Hawaiian honeycreeper species over the last century from areas below approximately 4,500 feet elevation points to effects of avian disease having been sufficiently severe and prolonged, and extensive in geographic scope, to cause widespread species' range contraction and possible extinction. It is highly likely avian disease is the primary causal factor for the disappearance of many species of Hawaiian honeycreepers from forested areas below 4,500 feet on the islands of Kauai, Oahu, Molokai, and Lanai (Scott *et al.* 1986, p. 148; Banko and Banko 2009, pp. 52–53; Atkinson and Lapointe 2009a, pp. 237–238).

It is widely established that small populations of animals are inherently more vulnerable to extinction because of random demographic fluctuations and stochastic environmental events (Mangel and Tier 1994, p. 607; Gilpin and Soulé 1986, pp. 24–34). Formerly widespread populations that become small and isolated often exhibit reduced levels of genetic variability, which diminishes the species' capacity to adapt and respond to environmental changes, thereby lessening the probability of long-term persistence (*e.g.,* Barrett and Kohn 1991, p. 4; Keller and Waller 2002, p. 240; Newman and Pilson 1997, p. 361). As populations are lost or decrease in size, genetic variability is reduced, resulting in increased vulnerability to disease and restricted potential evolutionary capacity to respond to novel stressors (Spielman *et al.* 2004, p. 15261; Whiteman *et al.* 2006, p. 797). As numbers decreased historically, effects of small population size were very likely to have negatively impacted Kauai akialoa, reducing its potential for long-term persistence.

Several recent surveys and searches (1981 to 2018), including the RBS, have been unsuccessful in detecting Kauai akialoa despite efforts by ornithologists (Conant *et al.* 1998, p. 15) and birders, and intensive survey efforts by wildlife biologists in 1968 to 1973, 1981, 1989, 1994, 2000, 2005, and from 2011 to 2018 (Hawaii Department of Land and Natural Resources unpubl. data; USFWS 1983, p. 2; Reynolds and Snetsinger 2001, entire; Crampton *et al.* 2017, entire; Crampton 2018, pers. comm.). Using 1969 as the last credible sightings, based on independent expert opinion, the estimated date for the species' extinction is 1973, with 95 percent confidence of the species having become extinct by 1984 (Elphick *et al.* 2010, p. 620).

IV. Conclusion

At the time of listing in 1967, the Kauai akialoa faced threats from habitat loss, avian disease, and predation by introduced mammals. The best available information now indicates that the Kauai akialoa is extinct. The species appears to have been vulnerable to introduced avian disease. In addition, the effects of small population size likely limited the species' genetic variation and adaptive capacity, thereby increasing the vulnerability of the species to environmental stressors including habitat loss and degradation. Since its last detection in 1969, qualified observers have conducted extensive surveys and searches but have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

Kauai nukupuu (Hemignathus hanapepe)

I. Background

The Kauai nukupuu (*Hemignathus hanapepe*) was listed as endangered on March 11, 1967 (32 FR 4001), and was included in the Kauai Forest Birds Recovery Plan (USFWS 1983), as well as the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006). At the time of listing, observations of only two individuals had been reported during that century (USFWS 1983, p. 3). The last confirmed observation (based on independent expert opinion and physical evidence) of the species was in 1899 (Eliphick *et al.* 2010, p. 620). Two 5-year status reviews have been completed, in 2010 (initiated on April 11, 2006; see 71 FR 18345) and 2019 (initiated on February 13, 2015; see 80 FR 8100). The 2010 review did not recommend a change in status, though there was some information indicating the species was already extinct. The 5-year status review completed in 2019 recommended delisting due to extinction based on consideration of additional information about the biological status of the species, included in the discussion below (USFWS 2019, pp. 4–5, 10).

The historical record provides little information on the life history of Kauai nukupuu (USFWS 2006, p. 2–89). There is no specific information on the lifespan or breeding biology of Kauai nukupuu, although it is presumed to be similar to its closest relative, akiapolaau (*Hemignathus munroi,* listed as *Hemignathus wilsoni*), a honeycreeper from the island of Hawaii. Similar to the akiapolaau, the Kauai nukupuu uses its bill to extract invertebrates from epiphytes, bark, and wood. The last confirmed observation (based on independent expert opinion and physical evidence) of Kauai nukupuu was in 1899 (Eliphick *et al.* 2010, p. 620); however, there was an unconfirmed observation in 1995 (Conant *et al.* 1998, p. 14).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Kauai nukupuu was a medium-sized, approximately 23-gram (0.78-ounce), Hawaiian honeycreeper (family Fringillidae, subfamily Drepanidinae) with an extraordinarily thin, curved bill, slightly longer than the bird's head. The lower mandible was half the length of the upper mandible. Adult male plumage was olive-green with a yellow head, throat, and breast, whereas adult female and immature plumage consisted of an olive-green head and yellow or yellowish gray under-parts (USFWS 2006, p. 2–89). The long, curved, and extremely thin bill of Kauai nukupuu, in combination with its brightly colored plumage, would have made this bird highly detectable to ornithologists and birders had it persisted (USFWS 2006, p. 2–89). No subsequent sightings or vocalizations have been documented since the unconfirmed sighting in 1995, despite extensive survey efforts.

Survey Effort

In the absence of early historical surveys, the extent of the geographical range of the Kauai nukupuu is unknown. A comprehensive survey of Hawaiian forest birds was initiated in the 1970s using the VCP method (Scott *et al.* 1986, entire) (see *Survey Effort* section for the Kauai akialoa, above, for the description of the VCP surveys). Several recent surveys and searches, including the RBS, have been unsuccessful in detecting Kauai nukupuu despite intensive survey efforts by wildlife biologists from 1968 to 1973, and in 1981, 1989 1993, 1994, 2000, 2005, and 2011 to 2018 (Hawaii Department of Land and Natural Resources unpubl. data; Reynolds and Snetsinger 2001, entire; Crampton *et al.* 2017, entire; Crampton 2018 pers. comm.). During the RBS, Kauai nukupuu were not detected. The lack of detections combined with analysis of detection probability ($P \geq 0.95$) suggested that the possible population count was fewer than 10 birds in 1996 (Reynolds and Snetsinger 2001, p. 142).

Extensive time has been spent by qualified observers in the historical range of the Kauai nukupuu searching for the small Kauai thrush (*Myadestes palmeri*), akekee (*Loxops caeruleirostris*), and Kauai creeper (*Oreomystis bairdi*). Hawaii Forest Bird Surveys (HFBS) were conducted in 1981, 1989, 1994, 2000, 2005, 2007, 2008, 2012, and 2018 (Paxton *et al.* 2016, entire). During the HFBS in 2012 and 2018, occupancy surveys and nest searches did not yield any new detections of the Kauai nukupuu. The KFBRP conducted mist-netting in various locations within the historical range for the Kauai nukupuu from 2006 through 2009, and from 2011 through 2018, and no Kauai nukupuu were caught or encountered (Crampton 2018, pers. comm.). Despite contemporary search efforts, the last credible sighting of Kauai nukupuu occurred in 1899.

Time Since Last Detection

Using 1899 as the last credible sighting of Kauai nukupuu based on independent expert opinion and physical evidence, the estimated date for the species' extinction was 1901, with 95 percent confidence that the species was extinct by 1906 (Elphick *et al.* 2010, p. 620).

III. Analysis

Some of the reported descriptions of this species better match the Kauai amakihi (*Chlorodrepanis stejnegeri*) (USFWS 2006, p. 2–90). Although skilled observers reported three unconfirmed sightings of Kauai nukupuu in 1995 (Reynolds and Snetsinger 2001, p. 142), extensive hours of searching within the historical range failed to detect any individuals. The last credible sightings of Kauai nukupuu was in 1899, based on independent expert opinion and physical evidence (Elphick *et al.* 2010, p. 620). It was estimated that 1901 was the year of extinction, with 95 percent confidence that the species was extinct by 1906. The species was likely vulnerable to the persistent threats of avian disease combined with habitat loss and degradation, which remain drivers of extinction for Hawaiian forest birds.

V. Conclusion

At the time of listing in 1967, the Kauai nukupuu had not been detected for almost 70 years. Since its last detection in 1899, qualified observers have conducted extensive surveys and searches throughout the range of the Kauai nukupuu and have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

Kauai ʻoʻo (Moho braccatus)

I. Background

The Kauai ʻoʻo (*Moho braccatus*) was listed as endangered on March 11, 1967 (32 FR 4001), and was included in the Kauai Forest Birds Recovery Plan (USFWS 1983), as well as the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006). At the time of listing, the population size was estimated at 36 individuals (USFWS 1983, p. 3). Threats to the species included the effects of low population numbers, habitat loss, avian disease, and predation by introduced mammals. The last plausible record of a Kauai ʻoʻo was a vocal response to a recorded vocalization played by a field biologist on April 28, 1987, in the locality of Halepaakai Stream. Two 5-year status reviews have been completed, in 2009 (initiated on July 6, 2005; see 70 FR 38972) and 2018 (initiated on February 13, 2015; see 80 FR 8100). The 2009 review did not recommend a change in status, though there was some information indicating the species was already extinct. The 5-year status review completed in 2018 recommended delisting due to extinction based on consideration of new information about the biological status of the species, included in the discussion below (USFWS 2019, pp. 5, 10).

The Kauai ʻoʻo measured 7.7 inches (19.5 centimeters) and was somewhat smaller than the *Moho* species on the other islands. It was glossy black on the head, wings, and tail; smoky brown on the lower back, rump, and abdomen; and rufous-brown on the upper tail coverts. It had a prominent white patch at the bend of the wing. The thigh feathers were golden yellow in adults and black in immature birds (Berger 1972, p. 107). The Kauai ʻoʻo is one of four known Hawaiian species of the genus *Moho* and one of five known Hawaiian bird species within the family Mohoidae (Fleischer *et al.* 2008, entire). Its last known habitat was the dense ohia forest in the valleys of Alakai Wilderness Preserve. It reportedly fed on various invertebrates and the fruits and nectar from ohia, lobelia, and other flowering plants. There is no information on the lifespan of the Kauai ʻoʻo.

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The vocalizations of this species were loud, distinctive, and unlikely to be overlooked. The song consisted of loud whistles that have been described as flute-like, echoing, and haunting, suggesting that detectability would be high in remaining suitable habitat if the Kauai ʻoʻo still existed (USFWS 2006 p. 2–47).

Survey Effort

In the absence of early historical surveys, the extent of the geographical range of the Kauai ʻoʻo cannot be reconstructed. The comprehensive surveys of Hawaiian forest birds are described in the Survey Effort section of the Kauai akialoa. Several recent surveys and searches, including the VCP and RBS, have been unsuccessful in detecting Kauai ʻoʻo despite intensive survey efforts by wildlife biologists from 1968 to 1973, and in 1981, 1989 1993, 1994, 2000, 2005, and 2011 to 2018 (Hawaii Department of Land and Natural Resources unpubl. data; Reynolds and Snetsinger 2001, entire; Crampton *et al.* 2017, entire; Crampton 2018 pers. comm.). During the RBS, coverage of the search area was extensive; therefore, there was a high probability of detecting a Kauai ʻoʻo. None were detected, and it was concluded the Kauai ʻoʻo was likely extinct ($P \geq 0.95$) (Reynolds and Snetsinger 2001, p. 142).

Extensive time has been spent by qualified observers in the historical range of the Kauai ʻoʻo searching for the small Kauai thrush (*Myadestes palmeri*), akekee (*Loxops caeruleirostris*), and Kauai creeper (*Oreomystis bairdi*). Hawaii Forest Bird Surveys (HFBS) were conducted in 1981, 1989, 1994, 2000, 2005, 2007, 2008, 2012, and 2018 (Paxton *et al.* 2016, entire). During the HFBS in 2012 and 2018, occupancy surveys and nest searches did not yield any new detections of Kauai ʻoʻo. The KFBRP conducted mist-netting in various locations within the historical range for Kauai ʻoʻo from 2006 through 2009 and 2011 through 2018, and no Kauai ʻoʻo were caught or encountered (Crampton 2018, pers. comm.). The last credible sighting was in 1987.

Time Since Last Detection

Using 1987 as the last credible sighting of the Kauai ʻoʻo based on independent expert opinion, the estimated date for the species' extinction was 1991, with 95 percent confidence that the species was extinct by 2000 (Elphick *et al.* 2010, p. 620).

III. Analysis

The various bird species in the subfamily Drepanidinae (also known as the Hawaiian honeycreepers), which includes Kauai ʻoʻo, are highly susceptible to introduced avian disease, particularly avian malaria (*Plasmodium relictum*). At elevations below approximately 4,500 feet (1,372 meters) in Hawaii, the key factor driving disease epizootics of pox virus (*Avipoxvirus*) and avian malaria is the seasonal and altitudinal distribution and density of the primary vector of these diseases, *Culex quinquefasciatus* (Atkinson and Lapointe 2009a, pp. 237–238, 245–246). Because they occur at similar altitudes and face similar threats, please refer to the Analysis section for the Kauai akialoa, above, for more information.

IV. Conclusion

At the time of listing in 1967, the Kauai ʻoʻo faced threats from effects of low population numbers, habitat loss, avian disease, and predation by introduced mammals. The best available information now indicates that the Kauai ʻoʻo is extinct. The species appears to have been vulnerable to introduced avian disease. In addition, the effects of small population size likely limited the species' genetic variation and adaptive capacity, thereby increasing the vulnerability of the species to environmental stressors including habitat loss and degradation. Since its last detection in 1987, qualified observers have conducted extensive surveys and searches and have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

Large Kauai Thrush (Myadestes myadestinus)

I. Background

The large Kauai thrush (*Myadestes myadestinus,* or kamaʻo in the Hawaiian language) was listed as endangered on October 13, 1970 (35 FR 16047), and was included in the Kauai Forest Birds Recovery Plan (USFWS 1983), as well as the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006). At the time of listing, the population size was estimated at 337 individuals (USFWS 1983, p. 3). Threats to the species included effects of low population numbers, habitat loss, avian disease, and predation by introduced mammals. Two 5-year status reviews were completed in 2009 (initiated on July 6, 2005; see 70 FR 38972) and 2019 (initiated on February 13, 2015; see 80 FR 8100). The 2009 review did not recommend a change in status, though there was some information indicating the species was already extinct. The 5-year status review completed in 2019 recommended delisting due to extinction based on consideration of additional information about the biological status of the species, included in the discussion below (USFWS 2019, pp. 5, 10).

The large Kauai thrush was a medium-sized (7.9 inches, or 20 centimeters, total length) solitaire. Its plumage was gray-brown above, tinged with olive especially on the back, and light gray below with a whitish belly and undertail coverts. The large Kauai thrush lacked the white eye-ring and pinkish legs of the smaller puaiohi (small Kauai thrush, *Myadestes palmeri*) (USFWS 2006, p. 2–19). There is no specific information on the life history of the large Kauai thrush; however, it is presumed that it is similar to the more common and closely related Hawaii thrush (*Myadestes obscurus*). Nests of the large Kauai thrush have not been described but may be a cavity or low platform, similar to those of the Hawaii thrush. Nesting likely occurred in the spring. The diet of the large Kauai thrush was reported to include fruits and berries, as well as insects and snails. The last (unconfirmed) observation of the large Kauai thrush was made during the February 1989 Kauai forest bird survey (Hawaii Department of Land and Natural Resources unpubl. data). However, the last credible sighting of the large Kauai thrush occurred in 1987.

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The large Kauai thrush was often described for its habit of rising into the air, singing a few vigorous notes and then suddenly dropping down into the underbrush. The vocalizations of this species varied between sweet and melodic to lavish and flute-like, often given just before dawn and after dusk (USFWS 2006 p. 2–19). These behaviors suggest that detectability would be high in remaining suitable habitat if the large Kauai thrush still existed. No subsequent sightings or vocalizations have been documented despite extensive survey efforts by biologists and birders.

Survey Effort

Several recent surveys and searches, including the VCP and RBS, have been unsuccessful in detecting the large Kauai thrush despite intensive survey efforts by wildlife biologists from 1968 to 1973, and in 1981, 1989, 1993, 1994, 2000, 2005, and 2011 to 2018 (Hawaii Department of Land and Natural Resources unpubl. data; Reynolds and Snetsinger 2001, entire; Crampton *et al.* 2017, entire; Crampton 2018, pers. comm.). During the RBS in 2001, coverage of the search area was extensive; therefore, they had a high probability of detecting the large Kauai thrush. None were detected, and it was concluded that the large Kauai thrush was likely extinct ($P \geq 0.95$) (Reynolds and Snetsinger 2001, p. 142).

Extensive time has been spent by qualified observers in the historical range of the large Kauai thrush searching for the small Kauai thrush (*Myadestes palmeri*), akekee (*Loxops caeruleirostris*), and Kauai creeper (*Oreomystis bairdi*). Hawaii Forest Bird Surveys (HFBS) were conducted in 1981, 1989, 1994, 2000, 2005, 2007, 2008, 2012, and 2018 (Paxton *et al.* 2016, entire). During the HFBS in 2012 and 2018, occupancy surveys and nest searches did not yield any new detections of the large Kauai thrush. The KFBRP conducted mist-netting in various locations within the historical range for the large Kauai thrush from 2006 through 2009, and from 2011 through 2018, and no large Kauai thrush were caught or encountered (Crampton 2018, pers. comm.). The last credible sighting of the large Kauai thrush occurred in 1987.

Time Since Last Detection

Using 1987 as the last credible sighting of the large Kauai thrush based on independent expert opinion, the estimated date for the species' extinction was 1991, with 95 percent confidence that the species was extinct by 1999 (Elphick *et al.* 2010, p. 620).

III. Analysis

Several recent surveys and searches, including the RBS and HFBS, have been unsuccessful in detecting the large Kauai thrush despite intensive survey efforts by wildlife biologists in 1993, 1994, 2000, 2005, and 2011 to 2018 (Hawaii Department of Land and Natural Resources unpubl. data; Reynolds and Snetsinger 2001, entire; Crampton *et al.* 2017, entire; Crampton 2018, pers. comm.). Using 1987 as the last credible sighting based on independent expert opinion and the species' observational record, the estimated date for the species' extinction was 1991, with 95 percent confidence the species was extinct by 1999 (Elphick *et al.* 2010, p. 620). Another analysis determined that the large Kauai thrush was probably extinct at the time of the RBS in 1994 ($P \geq 0.95$) (Reynolds and Snetsinger 2001, p. 142).

IV. Conclusion

At the time of listing in 1970, the large Kauai thrush faced threats from low population numbers, habitat loss, avian disease, and predation by introduced mammals. The best available information now indicates that the large Kauai thrush is extinct. The species appears to have been vulnerable to the effects of small population size, which likely limited its genetic variation, disease resistance, and adaptive capacity, thereby increasing the vulnerability of the species to the environmental stressors of habitat degradation and predation by nonnative mammals. Since its last credible detection in 1987, qualified observers have conducted extensive surveys and searches throughout the range of the species but have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

Maui Akepa (Loxops coccineus ochraceus)

I. Background

The Maui akepa (*Loxops coccineus ochraceus,* listed as *Loxops ochraceus*) was listed as endangered on October 13, 1970 (35 FR 16047), and was included in the Maui-Molokai Forest Birds Recovery Plan (USFWS 1984, pp. 12–13), and the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006, pp. 2–94, 2–134–2–137). At the time of listing, we considered Maui akepa to have very low population numbers, and to face threats from habitat loss, avian disease, and predation by introduced mammals. Three 5-year status reviews have been completed; the 2010 (initiated on April 11, 2006; see 71 FR 18345) and 2015 (initiated on March 6, 2012; see 77 FR 13248) reviews did not recommend a change in status, though there was some information indicating the species was already extinct (USFWS 2010, p. 12; USFWS 2015, p. 10). The 5-year status review completed in 2018 (initiated on February 12, 2016; see 81 FR 7571) recommended delisting due to extinction, based in part on continued lack of detections and consideration of extinction probability (USFWS 2018, pp. 5, 10).

The Maui akepa was known only from the island of Maui in the Hawaiian Islands. Maui akepa were found in small groups with young in the month of June when the birds were molting (Henshaw 1902, p. 62). The species was observed preying on various insects including small beetles, caterpillars, and small spiders, as well as drinking the nectar of ohia (*Metrosideros polymorpha*) flowers (Rothschild 1893 to 1900, pp. 173–176; Henshaw 1902, p. 62; Perkins 1903, pp. 417–420). The species appeared to also use the ohia tree for nesting as a pair of Maui akepa was observed building a nest in the terminal foliage of a tall ohia tree (Perkins 1903, p. 420).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Maui akepa adult males varied from dull brownish orange to ochraceus (light brownish yellow), while females were duller and less yellowish (USFWS 2006, p. 2–134). Although the species was easily identifiable by sight, its small body size (less than 5 inches (13 centimeters) long) and habitat type (dense rain forest) made visual detection difficult. Songs and calls of Maui akepa could be confused with those of other Maui forest bird species; therefore, detection of the species requires visual confirmation of the individual producing the songs and calls (USFWS 2006, p. 2–135).

Survey Effort

In the absence of early historical surveys, the extent of the geographical range of the Maui akepa is unknown. Because the species occupied Maui Island, one might expect that it also inhabited Molokai and Lanai Islands like other forest birds in the Maui Nui group, but there are no fossil records of Maui akepa from either of these islands (USFWS 2006, p. 2–135). All historical records of the Maui akepa in the late 19th and early 20th century were from high-elevation forests most accessible to naturalists, near Olinda and Ukulele Camp on the northwest rift of Haleakala, and from mid-elevation forests in Kipahulu Valley (USFWS 2006, p. 2–134). This range suggests that the birds were missing from forests at lower elevations, perhaps due to the introduction of disease-transmitting mosquitoes to Lahaina in 1826 (USFWS 2006, p. 2–135). From 1970 to 1995, there were few credible sightings of Maui akepa (USFWS 2006, p. 2–136).

The population of Maui akepa was estimated at 230 individuals, with a 95 percent confidence interval of plus or minus 290 individuals (Scott *et al.* 1986, pp. 37, 154) during VCP surveys in 1980. In other words, the estimate projects a maximum population of 520 individuals and a minimum population of zero. However, confidence intervals were large, and this estimate was based on potentially confusing auditory detections, and not on visual observation (USFWS 2006, p. 2–136). On Maui, VCP surveys are conducted at survey stations spaced 328 to 820 feet (100 to 250 meters) apart, on transect lines spaced 1 to 2 miles (1.6 to 3.2 kilometers) apart (Scott *et al.* 1986, pp. 34–40). It is estimated that 5,865 8-minute point counts would be needed to determine with 95 percent confidence the absence of Maui akepa on Maui (Scott *et al.* 2008, p. 7). In 2008, only 84 VCP counts had been conducted on Maui in areas where this species was known to have occurred historically. Although the results of the 1980 VCP surveys find Maui akepa extant at that time, tremendous effort is required using the VCP method to confirm this species' extinction (Scott *et al.* 2008). For Maui akepa, nearly 70 times more VCP counts than conducted up to 2008 would be needed to confirm the species' extinction with 95 percent confidence.

Songs identified as Maui akepa were heard on October 25, 1994, during the RBS in Hanawi Natural Area Reserve (Hanawi NAR) and on November 28, 1995, from Kipahulu Valley at 6,142 feet (1,872 meters) elevation, but the species was not confirmed visually. Auditory detections of Maui akepa require visual confirmation because of possible confusion or mimicry with similar songs of Maui parrotbill (*Pseudonestor xanthophrys*) (Reynolds and Snetsinger 2001, p. 140). The last confirmed record, as defined above, of Maui akepa was from Hanawi NAR in 1988 (Engilis 1990, p. 69).

Qualified observers spent extensive time searching for Maui akepa, po'ouli (*Melamprosops phaeosoma*), and Maui nukupuu (*Hemignathus lucidus affinis,* listed as *Hemignathus affinis*) in the 1990s. Between September 1995 and October 1996, 1,730 acres (700 hectares) in Hanawi NAR were searched during 318 person-days (Baker 2001, p. 147), including the area with the most recent confirmed sightings of Maui akepa. During favorable weather conditions (good visibility and no wind or rain) teams would stop when ''chewee'' calls given by Maui parrotbill, or when po'ouli and Maui nukupuu were heard, and would play either Maui parrotbill or akiapolaau (*Hemignathus munroi,* listed as *Hemignathus wilsoni*) calls and songs to attract the bird for identification. Six po'ouli were found, but no Maui akepa were detected (Baker 2001, p. 147). The Maui Forest Bird Recovery Project (MFBRP) conducted searches from 1997 through 1999 from Hanawi NAR to Koolau Gap (west of Hanawi NAR), for a total of 355 hours at three sites with no detections of Maui akepa (Vetter 2018, pers. comm.). The MFBRP also searched Kipahulu Valley on northern Haleakala from 1997 to 1999, for a total of 320 hours with no detections of Maui akepa. However, the Kipahulu searches were hampered by bad weather, and playback was not used (Vetter 2018, pers. comm.). Despite over 10,000 person-hours of searches in the Hanawi NAR and nearby areas from October 1995 through June 1999, searches failed to confirm earlier detections of Maui akepa (Pratt and Pyle 2000, p. 37). While working on Maui parrotbill recovery from 2006 to 2011, the MFBRP spent extensive time in the area of the last Maui akepa sighting. The MFBRP project coordinator concluded that if Maui akepa were present, they would have been detected (Mounce 2018, pers. comm.).

Time Since Last Detection

The last confirmed sighting (as defined for the RBS) of the Maui akepa was in 1988 (Engilis 1990, p. 69). Surveys conducted during the late 1980s to the 2000s failed to locate the species (Pratt and Pyle 2000, p. 37; Baker 2001, p. 147). Using 1980 as the last documented observation record for Maui akepa (the 1988 sighting did not meet the author's criteria for a ''documented'' sighting), 1987 was estimated to be the year of extinction of Maui akepa, with 2004 as the upper 95 percent confidence bound on that estimate (Elphick *et al.* 2010, p. 620).

III. Analysis

Reasons for decline presumably are similar to threats faced by other endangered forest birds on Maui, including small populations, habitat degradation by feral ungulates and introduced invasive plants, and predation by introduced mammalian predators, including rats (*Rattus* spp.), cats (*Felis catus*), and mongoose (*Herpestes auropunctatus*) (USFWS 2006, p. 2–136). Rats may have played an especially important role as nest predators of Maui akepa. While the only nest of Maui akepa ever reported was built in tree foliage, the birds may also have selected tree cavities as does the very similar Hawaii akepa (*Loxops coccineus coccineus*). In Maui forests, nest trees are of shorter stature than where akepa survive on Hawaii Island. Suitable cavity sites on Maui are low in the vegetation, some near or at ground level, and thus more accessible to rats. High densities of both black and Polynesian rats (*Rattus rattus* and *R. exulans*) are present in akepa habitat on Maui (USFWS 2006, p. 2–136).

The population of Maui akepa was estimated at 230 birds in 1980 (Scott *et al.* 1986, p. 154); however, confidence intervals on this estimate were large. In addition, this may have been an overestimate because it was based on audio detections that can be confused with similar songs of Maui parrotbill. The last confirmed sighting of Maui akepa was in 1988, from Hanawi NAR (Engilis 1990, p. 69). Over 10,000 search hours in Hanawi NAR and nearby areas including Kipahulu Valley from October 1995 through June 1999 failed to confirm presence of Maui akepa (Pratt and Pyle 2000, p. 37). Field presence by qualified observers from 2006 to 2011 in the area Maui akepa was last known failed to detect this species, and the MFBRP project coordinator concluded that if Maui akepa were present they would have been detected (Mounce 2018, pers. comm.). Further, using the method to determine probability of species extinction based on time (years) since the species was last observed (using 1980 as the last documented observation record, as described above), the estimated year the Maui akepa became extinct is 1987, with 2004 as the upper 95 percent confidence bound on that estimate (Elphick *et al.* 2010, p. 620).

IV. Conclusion

At the time of listing in 1970, we considered the Maui akepa to be facing threats from habitat loss, avian disease, and predation by introduced mammals. The best available information now indicates that the Maui akepa is extinct. The species appears to have been vulnerable to the effects of small population size, which likely limited its genetic variation, disease resistance, and adaptive capacity, thereby increasing the vulnerability of the species to the environmental stressors of habitat degradation and predation by nonnative mammals. Since the last detection in 1988, qualified observers have conducted extensive surveys in that same area with no additional detections of the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that best available scientific and commercial information indicates that the species is extinct.

Maui Nukupuu (Hemignathus lucidus affinis)

I. Background

The Maui nukupuu (*Hemignathus lucidus affinis,* listed as *Hemignathus affinis*) was listed as endangered on October 13, 1970 (35 FR 16047), and was included in the Maui-Molokai Forest Birds Recovery Plan (USFWS 1984, pp. 8, 10–12), and the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006, pp. 2–92–2–96). At the time of listing, we considered Maui nukupuu to have very low population numbers and to be threatened by habitat loss, avian disease, and predation by introduced mammals. The 5-year status review completed in 2018 (initiated on February 12, 2016; see 81 FR 7571) recommended delisting due to extinction (USFWS 2018, p. 11).

The Maui nukupuu was known only from the island of Maui in the Hawaiian Islands. The historical record provides little information on the life history of the Maui nukupuu (Rothschild 1893 to 1900, pp. 103–104; Perkins 1903, pp. 426–430). Nothing is known of its breeding biology, which likely was similar to its closest relative, the akiapolaau (*Hemignathus munroi*) on Hawaii Island. The Maui nukupuu was insectivorous and probed bark, lichen, and branches to extract insects, foraging behaviors that resembled those of akiapolaau. Diet of the Maui nukupuu was reported to be small weevils and larvae of orders Coleoptera and Lepidoptera (Perkins 1903, p. 429). There is scant evidence that Maui nukupuu took nectar from flowers. Maui nukupuu often joined mixed-species foraging flocks (Perkins 1903, p. 429).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The Maui nukupuu was a medium-sized (approximately 0.78 ounce, or 23 gram) Hawaiian honeycreeper with an extraordinarily thin, curved bill that was slightly longer than the bird's head. The lower mandible was half the length of the upper mandible and followed its curvature rather than being straight (as in the related akiapolaau) (USFWS 2006, p. 2–92). Adult males were olive green with a yellow head, throat, and breast, whereas adult females and juveniles had an olive-green head and yellow or yellowish gray under-parts. The species' coloration and bill shape were quite distinctive, making visual identification of Maui nukupuu relatively easy. The Maui nukupuu's song resembled the warble of a house finch (*Carpodacus mexicanus*), but was lower in pitch. Both the song and the ''kee-wit'' call resembled those of Maui parrotbill (*Pseudonestor xanthophrys*), and audio detection required visual confirmation (USFWS 2006, p. 2–92).

Survey Effort

Historically, the Maui nukupuu was known only from Maui, but subfossil bones of a probable Maui nukupuu from Molokai show that the species likely formerly inhabited that island (USFWS 2006, p. 2–92). All records from late 19th and early 20th centuries were from locations most accessible to naturalists, above Olinda on the northwest rift of Haleakala, and from mid-elevation forests in Kipahulu Valley (USFWS 2006, pp. 2–134). Observers at the time noted the restricted distribution and low population density of Maui nukupuu. As on Kauai, introduced mosquitoes and avian diseases may have already limited these birds to forests at higher elevations, and we can presume that the Maui nukupuu once had a much wider geographic range (USFWS 2006, pp. 2–92). In 1967, Maui nukupuu were rediscovered in the upper reaches of Kipahulu Valley on the eastern slope of Haleakala, east Maui (Banko 1968, pp.

65–66; USFWS 2006, pp. 2–95). Since then, isolated sightings have been reported on the northern and eastern slopes of Haleakala, but these reports are uncorroborated by behavioral information or follow-up sightings (USFWS 2006, pp. 2–95).

Based on a single sighting of an immature bird during VCP surveys in 1980, the population of Maui nukupuu was estimated to be 28 individuals, with a 95 percent confidence interval of plus or minus 56 individuals (Scott *et al.* 1986, pp. 37, 131). On Maui, VCP surveys are conducted at survey stations spaced 328 to 820 feet (100 to 250 meters) apart, on transect lines spaced 1 to 2 miles (1.6 to 3.2 kilometers) apart (Scott *et al.* 1986, pp. 34–40). It was estimated that 1,357 8-minute point counts would be needed to determine with 95 percent confidence the absence of Maui nukupuu on Maui (Scott *et al.* 2008, p. 7). In 2008, only 35 VCP counts had been conducted on Maui in areas where Maui nukupuu could still potentially exist. Although the results of VCP surveys in 1980 find Maui nukupuu extant at that time, a tremendous effort is required to confirm this species' extinction using VCP method (Scott *et al.* 2008). For Maui nukupuu, nearly 39 times more VCP counts than conducted up to 2008 would be needed to confirm this species' extinction with 95 percent confidence. The RBS reported an adult male Maui nukupuu with bright yellow plumage at 6,021 feet (1,890 meters) elevation in 1996 from Hanawi Natural Area Reserve (Hanawi NAR) (Reynolds and Snetsinger 2001, p. 140). Surveys and searches have been unsuccessful in finding Maui nukupuu since the last confirmed sighting by RBS. Based on these results, the last reliable record of Maui nukupuu was from Hanawi NAR in 1996 (24 years ago).

Qualified observers spent extensive time searching for Maui nukupuu, po'ouli (*Melamprosops phaeosoma*), and Maui akepa (*Loxops coccineus ochraceus,* listed as *Loxops ochraceus*) in the 1990s. Between September 1995 and October 1996, 1,730 acres (700 hectares) of Hanawi NAR were searched during 318 person-days (Baker 2001, p. 147). Please refer to the Maui akepa Survey Effort section above for the method used in this survey. The Maui Forest Bird Recovery Project (MFBRP) conducted searches from 1997 to 1999, from Hanawi NAR to Koolau Gap (west of the last sighting of Maui nukupuu) for a total of 355 hours of searches at three sites with no detections of Maui nukupuu (Vetter 2018, pers. comm.). The MFBRP also searched Kipahulu Valley on northern Haleakala from 1997 to 1999, for a total of 320 hours, with no detections of Maui nukupuu. The Kipahulu searches were hampered, however, by bad weather, and playback was not used (Vetter 2018, pers. comm.). Despite over 10,000 person-hours of searching in the Hanawi NAR and nearby areas from October 1995 through June 1999, searches failed to confirm detection in 1996 of Maui nukupuu, or produce other sightings (Pratt and Pyle 2000, p. 37). While working on Maui parrotbill recovery from 2006 to 2011, the MFBRP spent extensive time in the area of the last Maui nukupuu sighting. The MFBRP project coordinator concluded that if Maui nukupuu were still present they would have been detected (Mounce 2018, pers. comm.).

Time Since Last Detection

The Maui nukupuu was last sighted in the Hanawi NAR in 1996 (Reynolds and Snetsinger 2001, p. 140). Surveys conducted during the late 1990s and early 2000s were unable to locate the species (Pratt and Pyle 2000, p. 37; Baker 2001, p. 147).

Elphick et al 2010 (p. 630) attempted to apply their method to predict the probability of species extinction for the Maui nukupuu based on time (years) since the species was last observed (see *Time Since Last Detection* section for Kauai akialoa, above). Basing extinction probability solely on the sighting record without physical evidence has the drawback that an incorrect assignment of species extinction may occur due to inadequate survey effort and/or insufficient time spent by qualified observers in areas where the species could still potentially exist. Therefore, observations in 1967, 1980, and 1996 were not considered for this analysis because they did not meet the researchers' criteria for a confirmed sighting. Therefore, using 1896 as the last observation of Maui nukupuu, under their stringent criteria, the authors were unable to determine an estimated date for species extinction.

III. Analysis

The Maui nukupuu is also affected by small population sizes and other threats, as discussed above under the Analysis section for the Maui akepa. The population of Maui nukupuu was estimated to be 28 birds in 1980 (Scott *et al.* 1986, pp. 37, 131); however, confidence intervals on this estimate were large. This population was vulnerable to negative effects of small population size, including stochastic effects and genetic drift that can accelerate the decline of small populations. However, even rare species can persist despite having low numbers. The last confirmed sighting of Maui nukupuu was in 1996, from Hanawi NAR (Reynolds and Snetsinger 2001, p. 140). Over 10,000 person-search hours in Hanawi NAR and nearby areas, including Kipahulu Valley, from October 1995 through June 1999 failed to confirm this sighting or to detect other individuals (Pratt and Pyle 2000, p. 37). While working on Maui parrotbill recovery from 2006 to 2011, the MFBRP spent extensive time in the area of the last Maui nukupuu sighting; however, no Maui nukupuu were observed, and the MFBRP project coordinator concluded that if Maui nukupuu were still present they would have been detected (Mounce 2018, pers. comm.).

IV. Conclusion

At the time of listing in 1970, Maui nukupuu had very low population numbers and faced threats from habitat loss, avian disease, and predation by introduced mammals. The species appears to have been vulnerable to avian disease and the effects of small population size. The latter likely limited the species' genetic variation and adaptive capacity, thereby increasing the vulnerability of the species to the environmental stressors of habitat degradation and predation by nonnative mammals. Since its last detection in 1996, qualified observers have conducted extensive searches in the area where the species was last sighted and other native forest habitat where the species occurred historically, but have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial data indicate that the species is extinct.

Molokai Creeper (Paroeomyza flammea)

I. Background

The Molokai creeper (*Paroreomyza flammea,* or kākāwahie in the Hawaiian language) was listed as endangered on October 13, 1970 (35 FR 16047), and was included in the Maui-Molokai Forest Birds Recovery Plan (USFWS 1984, pp. 18–20) and the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006, pp. 2–121– 2–123). At the time of listing, the Molokai creeper was considered extremely rare and faced threats from habitat loss, avian disease, and predation by introduced mammals. Three 5-year status reviews have been completed; the 2009 (initiated on July 6, 2005; see 70 FR 38972) and 2015 (initiated on March 6, 2012; see 77 FR 13248) reviews did not recommend a change in status, though there was some information indicating the species was already extinct (USFWS 2009, p. 11; USFWS 2015, p. 8). The 5-year status review completed in 2018 (initiated on February 12, 2016; see 81 FR 7571) recommended delisting due to extinction based in part on continued lack of detections and consideration of extinction probability (USFWS 2018, p. 9).

The Molokai creeper was known only from Molokai in the Hawaiian Islands. Only fragmentary information is available about the life history of the species from the writings of early naturalists (Perkins 1903, pp. 413–417; Pekelo 1963, p. 64; USFWS 2006, p. 2–122). This species was an insectivore that gleaned vegetation and bark in wet ohia (*Metrosideros polymorpha*) forests and was known almost solely from boggy areas of Molokai (Pekelo 1963, p. 64), although there is one record in 1907 of the species from lower elevation forest of leeward east Molokai (USFWS 2006, pp. 2–121).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Adult males were mostly scarlet in various shades, while adult females were brown with scarlet washes and markings, and juvenile males ranged from brown to scarlet with many gradations. The bill was short and straight. Its calls were described as chip or chirping notes similar to other creeper calls (USFWS 2006, pp. 2–122). Its closest relatives are the Maui creeper (*Paroreomyza montana*) and the Oahu creeper (*P. maculata*). The species' coloration and bill shape were distinctive, and Molokai creeper was identified visually with confidence.

Survey Effort

Molokai creeper was common in 1907, but by the 1930s, they were considered in danger of extinction (Scott *et al.* 1986, p. 148). The species was last detected in 1963, on the west rim of Pelekunu Valley (Pekelo 1963, p. 64). Surveys and searches have been unsuccessful in finding the Molokai creeper since the last sighting, including VCP surveys on the Olokui Plateau in 1980 and 1988, and the RBS of the Kamakou-Pelekunu Plateau in 1995 (Reynolds and Snetsinger 2001, p. 141). Following up on a purported sighting in 2005 of a Molokai thrush (*Myadestes lanaiensis rutha*), a survey was conducted over 2 to 3 days in Puu Alii Natural Area Reserve (Puu Alii NAR), the last place the Molokai creeper was sighted in the 1960s (Pekelo 1963, p. 64; USFWS 2006, pp. 2–29). Using playback recordings for Molokai thrush, searchers covered the reserve area fairly well, but no Molokai creepers or Molokai thrush were detected (Vetter 2018, pers. comm.).

No Molokai creepers were detected during VCP surveys beginning in the late 1970s to the most recent Hawaiian forest bird survey on Molokai in 2010 (Scott *et al.* 1986, p. 37; Camp 2015, pers. comm.). On Molokai, VCP surveys are 8-minute point counts conducted at stations separated by a distance of 492 to 656 feet (150 to 200 meters) along transect lines 1 to 2 miles (1.6 to 3.2 kilometers) apart (Scott *et al.* 1986, pp. 34–40). It was estimated that 215,427 8-minute point counts would be needed to determine with 95 percent confidence the absence of Molokai creeper on Maui (Scott *et al.* 2008, p. 7). In 2008, only 131 VCP counts had been conducted on Molokai in areas where Molokai creeper could still potentially exist. For the Molokai creeper, nearly 1,650 times more VCP counts than conducted up to 2008 would be needed to confirm the species' extinction with 95 percent confidence. Based on species detection probability, the RBS determined the likelihood of the Molokai creeper being extirpated from the Kamakou-Pelekunu plateau was greater than 95 percent. The RBS estimated the Molokai creeper to be extinct over the entirety of its range, but, because not all potential suitable habitat was searched, extinction probability was not determined (Reynolds and Snetsinger 2001, p. 141).

Time Since Last Detection

The last reliable record (based on independent expert opinion and physical evidence) of Molokai creeper was from Pelekunu Valley in 1963 (Pekelo 1963, p. 64). Using 1963 as the last reliable observation record for Molokai creeper, 1969 is estimated to be year of extinction of the Molokai creeper with 1985 as the upper 95 percent confidence bound (Elphick *et al.* 2010, p. 620).

III. Analysis

The Molokai creeper faces similar threats to the other Maui bird species (see *Analysis* section for the Maui akepa, above). The last confirmed detection of the Molokai creeper was in 1963 (Pekelo 1963, p. 64). Forest bird surveys in 1980, 1988, and 2010, and the RBS in 1994–1996 (although not including the Olokui Plateau), failed to detect this species. A 2- to 3-day search by qualified personnel for the Molokai thrush in Puu Alii NAR in 2005, the last location where Molokai creeper was sighted, also failed to detect the Molokai creeper. The estimated year of extinction is 1969, with 1985 as the 95 percent confidence upper bound (Elphick *et al.* 2010, p. 620). It is highly likely that avian disease, thought to be the driver of range contraction and disappearance of many Hawaiian honeycreeper species, was present periodically throughout nearly all of the Molokai creeper's range over the last half-century.

IV. Conclusion

At the time of listing in 1970, the Molokai creeper was considered to be facing threats from habitat loss, avian disease, and predation by introduced mammals. The best information now indicates that the Molokai creeper is extinct. The species appears to have been vulnerable to avian disease, as well as the effects of small population size. The latter likely limited the species' genetic variation and adaptive capacity, thereby increasing the vulnerability of the species to the environmental stressors of habitat degradation and predation by nonnative mammals. Since its last detection in 1963, qualified observers have conducted extensive searches for the Molokai creeper but have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

Po'ouli (Melamprosops phaeosoma)

I. Background

We listed the po'ouli (*Melamprosops phaeosoma*) as endangered on September 25, 1975 (40 FR 44149), and the species was included in the Maui-Molokai Forest Birds Recovery Plan (USFWS 1984, pp. 16–17), and the Revised Recovery Plan for Hawaiian Forest Birds (USFWS 2006, pp. 2–144–2–154). At the time of listing, we considered the po'ouli to have very low abundance and to likely be threatened by habitat loss, avian disease, and predation by introduced mammals. Three 5-year status reviews have been completed; the 2010 (initiated on April 11, 2006; see 71 FR 18346) and 2015 (initiated on March 6, 2012; see 77 FR 13248) reviews did not recommend a change in status, though there was some information indicating the species was already extinct (USFWS 2010, p. 13; USFWS 2105, p. 8). The 5-year status review completed in 2018 (initiated on February 12, 2016; see 81 FR 7571) recommended delisting due to extinction, based in part on continued lack of detections and consideration of extinction probability (USFWS 2018, pp. 4–5, 10).

The poʻouli was known only from the island of Maui in the Hawaiian Islands and was first discovered in 1973, in high-elevation rainforest on the east slope of Haleakala (USFWS 2006, p. 2–146). Fossil evidence shows that the poʻouli once inhabited drier forests at lower elevation on the leeward slope of Haleakala, indicating it once had a much broader geographic and habitat range (USFWS 2006, p. 2–147). Poʻouli were observed singly, in pairs, and in family groups consisting of both parents and a single offspring (Pratt *et al.* 1997, p. 1). Poʻouli foraged primarily on tree branches, making extensive use of the subcanopy and understory. They seemed to have preferred the native hydrangea (kanawao (*Broussaisia arguta*)), the native holly (kawau (*Ilex anomala*)), and ohia (*Metrosideros polymorpha*) (Pratt *et al.* 1997, p. 4). Poʻouli gleaned from, probed, and excavated moss mats, lichen, and bark for small invertebrate prey. Egg-laying took place in March and April for two nests observed, and clutch size was probably two eggs (Kepler *et al.* 1996, pp. 620–638). The female alone incubated eggs and brooded chicks, but both parents fed the chicks. Throughout nesting, the male fed the female at or away from the nest. Poʻouli often associated with mixed species foraging flocks of other insectivorous honeycreepers. Poʻouli were unusually quiet. Males rarely sang and did so mostly as part of courtship prior to egg-laying. The maximum lifespan of this species is estimated to be 9 years (The Animal Aging and Longevity Database 2020, unpaginated).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The poʻouli was a medium-sized, 0.9 ounce (26 gram), stocky Hawaiian honeycreeper, easily recognized by its brown plumage and characteristic black mask framed by a gray crown and white cheek patch. However, poʻouli were unusually quiet. Although distinctive visually, because the species rarely vocalized, it was difficult to survey by audio detections.

Survey Effort

The poʻouli was first discovered in 1973 (USFWS 2006, p. 2–146). Total population was estimated at 140 individuals, with a 95 percent confidence interval of plus or minus 280 individuals, during VCP surveys in 1980 (Scott *et al.* 1986, pp. 37, 183), but estimates of population size and density were likely inaccurate and considered imprecise due to the species' low density and cryptic behavior (USFWS 2006, p. 2–147). In 1994, after nearly 2 years without a sighting, the continued existence and successful breeding of five to six poʻouli in the Kuhiwa drainage of Hanawi Natural Area Reserve (Hanawi NAR) was confirmed (Reynolds and Snetsinger 2001, p. 141). Thorough surveys of the historical range between 1997 and 2000, the Maui Forest Bird Recovery Program (MFBRP) located only three birds, all in separate territories in Hanawi NAR. These three poʻouli were color-banded in 1996 and 1997, and subsequently observed (see below), but no other individuals have been observed since then (Baker 2001, p. 144; USFWS 2006, pp. 2–147–2–148). The MFBRP searched Kipahulu Valley on northern Haleakala from 1997 to 2000, for a total of 320 hours, but failed to detect poʻouli. These searches were hampered by bad weather, however, and playback was not used (Vetter 2018, pers. comm.).

Time Since Last Detection

In 2002, what was thought to be the only female poʻouli of the three in Hanawi NAR was captured and released into one of the male's territories, but she returned to her home range the following day (USFWS 2006, p. 2–151). In 2004, an effort was initiated to capture the three remaining poʻouli to breed them in captivity. One individual was captured and successfully maintained in captivity for 78 days, but died on November 26, 2004, before a potential mate could be obtained. The remaining two birds were last seen in December 2003 and January 2004 (USFWS 2006, pp. 2–153–2–154). While working on Maui parrotbill (*Pseudonestor xanthophrys*) recovery from 2006 to 2011, the MFBRP spent extensive time in the area of the last poʻouli sightings. No poʻouli were seen or heard. The MFBRP project coordinator concluded that if poʻouli were present, they would have been detected (Mounce 2018, pers. comm.).

Using 2004 as the last reliable observation record for poʻouli, 2005 is estimated to be the year of extinction, with 2008 as the upper 95 percent confidence bound on that estimate (Elphick *et al.* 2010, p. 620).

III. Analysis

The Poʻouli faced similar threats to other Maui occurring bird species (see the Analysis section for the Maui akepa, above). The last confirmed sighting of poʻouli was in 2004 from Hanawi NAR (USFWS 2006, p. 2–154). Extensive field presence by qualified individuals from 2006 to 2011 in Hanawi NAR, where poʻouli was last observed, failed to detect this species, as did searches of Kipahulu Valley near Hanawi NAR from 1997 to 1999 (USFWS 2006, p. 2–94). Using 2004 as the last reliable observation record for poʻouli, the estimated year the species went extinct is 2005, with 2008 the upper 95 percent confidence bound on that estimate (Elphick *et al.* 2010, p. 620).

IV. Conclusion

At the time of its listing in 1975, we considered poʻouli to have very low population abundance, and to face threats from habitat loss, avian disease, and predation by introduced mammals. The best available information now indicates that the poʻouli is extinct. Although the poʻouli was last detected as recently as early 2004, the species appears to have been vulnerable to the effects of small population size since it was first discovered in 1973. The small population size likely limited its genetic variation, disease resistance, and adaptive capacity over time, thereby increasing the vulnerability of the species to the environmental stressors of habitat degradation and predation by nonnative mammals. Experienced staff with MFBRP conducted extensive recovery work in poʻouli habitat between 2006 and 2011 and had no detections of the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the species is extinct.

Fishes

San Marcos Gambusia (Gambusia georgei)

I. Background

We listed the San Marcos gambusia (*Gambusia georgei*), a small fish, as endangered throughout all of its range on July 14, 1980 (45 FR 47355). We concurrently designated approximately 0.5 miles of the San Marcos River as critical habitat for the species (45 FR 47355, July 14, 1980, p. 47364). The San Marcos gambusia was endemic to the San Marcos River in San Marcos, Texas. The San Marcos gambusia has historically only been found in a section of the upper San Marcos River approximately from Rio Vista Dam to a point near the U.S. Geological Survey gaging station immediately downstream from Thompson's Island. Only a limited number of species of *Gambusia* are native to the United States; of this subset, the San Marcos gambusia had one of the most restricted ranges.

We listed the species as endangered due to decline in population size, low population numbers, and possibility of lowered water tables, pollution, bottom plowing (a farming method that brings subsoil to the top and buries the previous top layer), and cutting of vegetation (43 FR 30316, July 14, 1978, p. 30317). We identified groundwater depletion, reduced spring flows, contamination, habitat impacts resulting from severe drought conditions, and cumulative effects of human activities as threats to the species (45 FR 47355, July 14, 1980, p. 47361). At the time of listing, this species was extremely rare.

There has also been evidence of hybridization between *G. georgei* and *G. affinis* (western mosquitofish) in the wild. Hybridization between *G. georgei* and *G. affinis* continued for many years without documented transfer of genes between the species that would have resulted in the establishment of a new species (Hubbs and Peden 1969, p. 357). Based on collections in the 1920s, a study in the late 1960s, surmised that limited hybridization with *G. affinis* did not seem to have reduced the specific integrity of either species. However, as fewer *G. georgei* individuals existed in the wild and therefore encountered each other, the chances of hybridization with the much more common *G. affinis* increased.

All currently available scientific data and field survey data indicate that this species has been extinct in the wild for over 35 years. The last known sighting in the wild was in 1983, and past hybridization in the wild between *G. georgei* and *G. affinis* failed to result in establishment of a hybridized species that would facilitate the transfer of genes from one species to the other. Also, captive breeding attempts of *G. georgei* failed. In 1985, the last captive female San Marcos gambusia died. Because no males remained, we concluded captive breeding efforts, and no individuals remain alive in captivity today.

On March 20, 2008, we published a notice in the **Federal Register** (73 FR 14995) that we were initiating a 5-year review of the species. We did not receive any comments or new information, and the 5-year review was not completed at that time. On May 31, 2018, we published a notice in the **Federal Register** (83 FR 25034) initiating another 5-year review of the species. The review relied on available information, including survey results, fish collection records, peer-reviewed literature, various agency records, and correspondences with leading *Gambusia* species experts in Texas. That 5-year review recommended delisting the San Marcos gambusia due to extinction.

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Historically, the San Marcos gambusia had small populations, and the pattern of abundance strongly suggests a decrease beginning prior to the mid-1970s. Historical records indicate that San Marcos gambusia was likely collected from the headwaters of the San Marcos River (Hubbs and Peden 1969, p. 28). The highest number of San Marcos gambusia ever collected was 119 in 1968. Because this species preferred sections of slow-moving waters and had a limited historical range of a small section of the San Marcos River, potential detection was not expected to be difficult.

Survey Effort

In 1976, we contracted a status survey to improve our understanding of the species and its habitat needs. We facilitated bringing individuals into captivity for breeding and study. Many researchers have been involved and have devoted considerable effort to attempts to locate and preserve populations. Intensive collections during 1978 and 1979 yielded only 18 San Marcos gambusia from 20,199 *Gambusia* total, which means San Marcos gambusia amounted to only 0.09 percent of those collections (Edwards *et al.* 1980, p. 20). Captive populations were established at the University of Texas at Austin in 1979, and fish from that captive population were used to establish a captive population at our Dexter National Fish Hatchery in 1980. Both captive populations later became contaminated with another *Gambusia* species. The fish hybridized, and the pure stocks were lost.

Following the failed attempt at maintaining captive populations at Dexter National Fish Hatchery and the subsequent listing of the species in 1980, we contracted for research to examine known localities and collect fish to establish captive refugia. Collections made in 1981 and 1982 within the range of San Marcos gambusia indicated a slight decrease in relative abundance of this species (0.06 percent of all *Gambusia*). From 1981 to 1984, efforts were made to relocate populations and reestablish a culture of individuals for captive refugia. Too few pure San Marcos gambusia and hybrids were found to establish a culture, although attempts were made with the few fish available (Edwards *et al.* 1980, p. 24). In the mid-1980s, staff from the San Marcos National Fish Hatchery and Technology Center also searched unsuccessfully for the species in attempts to locate individuals to bring into captivity.

Intensive searches for San Marcos gambusia were conducted in May, July, and September of 1990, but were unsuccessful in locating any pure San Marcos gambusia. The searches consisted of more than 180 people-hours of effort over the course of 3 separate days and covered the area from the headwaters at Spring Lake to the San Marcos wastewater treatment plant outfall. Over 15,450 *Gambusia* were identified during the searches. One individual collected during the search was visually identified as a possible backcross of *G. georgei* and *G. affinis* (Service 1990 permit report). This individual was an immature fish with plain coloration. Additional sampling near the Interstate Highway 35 type locality has occurred at approximately yearly intervals since 1990, and no San Marcos gambusia have been found. No San Marcos gambusia were found in the 32,811 *Gambusia* collected in the upper San Marcos River by the Service from 1994 to 1996 (Edwards 1999, pp. 6–13).

Time Since Last Detection

Academic researchers, Texas Parks and Wildlife Department scientists, and the Service have continued to search for the San Marcos gambusia during all collection and research with fishes on the San Marcos River. San Marcos gambusia have not been found in the wild since 1983, even with intensive searches, including the ones conducted in May, July, and September of 1990, covering the species' known range and designated critical habitat. Since 1996, all attempts to locate and collect San Marcos gambusia have failed (Edwards 1999, p. 3; Edwards *et al.* 2002, p. 358; Hendrickson and Cohen 2015; Bio-West 2016, p. 43; Bonner 2018, pers. comm.). More recent surveys and analyses of fish species already consider the San Marcos gambusia extinct (Edwards *et al.* 2002; Hubbs *et al.* 2008). Additionally, hybridized individuals have not been documented since 1990.

III. Analysis

Although the population of San Marcos gambusia was historically small, it also had one of the most restricted ranges of *Gambusia* species. San Marcos gambusia have not been found in the wild since 1983, even with intensive searches, including the ones conducted in May, July, and September of 1990, covering the species' known range and designated critical habitat. No San Marcos gambusia were found in the 32,811 *Gambusia* collected in the upper San Marcos River by the Service from 1994 to 1996 (Edwards 1999, pp. 6–13).

Additional sampling near the Interstate Highway 35 type locality has occurred at approximately yearly intervals since 1990. Since 1996, all attempts to survey and collect San Marcos gambusia failed to find them (Edwards 1999, p. 3; Edwards *et al.* 2002, p. 358; Hendrickson and Cohen 2015; Bio-West 2016, p. 43; Bonner 2018, pers. comm.). Additionally, no detections of hybridized San Marcos gambusia with *G. affinis* is further evidence that extinction has occurred.

In addition to the San Marcos gambusia not being found in the wild, all attempts at captive breeding have failed. This is largely due to unsuccessful searches for the species in attempts to locate individuals to bring into captivity.

Due to the narrow habitat preference and limited range of the San Marcos gambusia, and the exhaustive survey and collection efforts that have failed to detect the species, we conclude there is a very low possibility of an individual or population remaining extant but undetected. Therefore, the decrease in San Marcos gambusia abundance, and the lack of hybridized individuals in any recent samples, indicates that the species is extinct.

IV. Conclusion

The San Marcos gambusia was federally listed as endangered in 1980. At the time of listing, this species was rare. The last known collections of San Marcos gambusia from the wild were in 1981 (Edwards 2018, pers. comm.), and the last known sighting in the wild occurred in 1983. In 1985, after unsuccessful breeding attempts with *Gambusia affinis* from the upper San Marcos River, the last captive female San Marcos gambusia died. All available information and field survey data support a determination that the San Marcos gambusia has been extinct in the wild for more than 35 years. We have reviewed the best scientific and commercial data available to conclude that the species is extinct.

Scioto Madtom (Noturus trautmani)

I. Background

The Scioto madtom (*Noturus trautmani*) was listed as endangered on September 25, 1975 (40 FR 44149) due to the pollution and siltation of its habitat and the proposal to construct two impoundments within its range. Scioto madtom was included in 5-year reviews initiated on February 27, 1981 (46 FR 14652), July 22, 1985 (50 FR 29901), and on November 6, 1991 (56 FR 56882). These reviews resulted in no change in the Scioto madtom's listing classification of endangered. Two additional 5-year reviews were initiated in 2009 (74 FR 11600; March 18, 2009) and 2014 (79 FR 38560; July 8, 2014). The recommendations from both of these reviews were to delist the species due to extinction (Service 2009, p. 7; Service 2014, p. 6).

The Scioto madtom was a small, nocturnal species of catfish in the family Ictaluridae. The Scioto madtom has been found only in a small section of Big Darby Creek, a major tributary to the Scioto River, and was believed to be endemic to the Scioto River basin in central Ohio (40 FR 44149, September 25, 1975; Service 1985, p. 10; Service 1988, p. 1).

The species was first collected in 1943 (Trautman 1981, p. 504), and was first described as a species, *Noturus trautmani,* in 1969 (Taylor 1969, pp. 156–160). Only 18 individuals of the Scioto madtom were ever collected. All were found along one stretch of Big Darby Creek, and all but one were found within the same riffle known as Trautman's riffle. The riffle habitat was comprised of glacial cobble, gravel, sand, and silt substrate, with some large boulders (Trautman 1981, p. 505) with moderate current and high-quality water free of suspended sediments.

The Scioto madtom was an omnivorous bottom feeder that ate a wide variety of plant and animal life, which it found with its sensory barbels hanging down in front of its mouth. Little is known of its reproductive habits, although it likely spawned in summer and migrated downstream in the fall (Trautman 1981, p. 505).

The exact cause of the Scioto madtom's decline is unknown, but was likely due to modification of its habitat from siltation, suspended industrial effluents, and agricultural runoff (40 FR 44149, September 25, 1975; Service 1988, p. 2). At the time of listing, two dams were proposed for Big Darby Creek, although ultimately they were never constructed. It should also be noted that the northern madtom (*Noturus stigmosus*) was first observed in Big Darby Creek in 1957, the same year the last Scioto madtom was collected (Service 1982, p. 3; Kibbey 2009, pers. comm.). Both species likely feed on small invertebrates and shelter in openings in and around rocks and boulders. Given the apparent small population size and highly restricted range of the Scioto madtom in the 1940s and 1950s, it is possible that the species was unable to successfully compete with the northern madtom for the same food and shelter resources (Kibbey 2009, pers. comm.).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

The Scioto madtom looked similar to other madtom species but could be distinguished by meristic and morphometric characters, such as the number of pectoral and anal rays. The species, like other madtom species, was relatively cryptic as they hid during the daylight hours under rocks or in vegetation and emerged after dark to forage along the bottom of the stream. Despite these detection challenges, many surveys by experienced biologists have been undertaken to try to locate extant populations of Scioto madtom.

Survey Effort

No Scioto madtoms have been observed since 1957, despite intensive fish surveys throughout Big Darby Creek in 1976–1977 (Service 1977, p. 15), 1981–1985 (Service 1982, p. 1; Service 1985, p. 1), 2014–2015 (OEPA 2018, p. 48), and 2001–2019 (Kibbey 2009, pers. comm.; Zimmerman 2014, 2020, pers. comm.).

The fish surveys conducted in Big Darby Creek in 1976–1977 and 1981–1985 specifically targeted the Scioto madtom. The 1976–1977 survey found 41 madtoms of 3 species and 34 species of fish in riffles at and near the Scioto madtom type locality (Service 1977, pp. 13–15). The 1981–1985 survey occurred throughout Big Darby Creek and found a total of 2,417 madtoms of 5 species (Service 1985, pp. 1, 5, 19–23). Twenty-two percent (542 individuals) of the total madtoms were riffle madtoms of the subgenus Rabida, which also includes the Scioto madtom (Service 1985, p. 1). None of the species identified were the Scioto madtom.

The 2014–2015 fish surveys occurred throughout the Big Darby Creek watershed as part of the Ohio Environmental Protection Agency's (OEPA's) water quality monitoring program. A total of 96,471 fish representing 85 different species and 6 hybrids, were collected at 93 sampling locations throughout the Big Darby Creek study area during the 2014 sampling season. Fish surveys were conducted at numerous sites in Big Darby Creek between 2001 and 2019, using a variety of survey techniques, including seining, boat electrofishing, backpack electrofishing, and dip netting (Zimmerman 2020, pers. comm.). Another survey was also conducted annually in the Big Darby Creek from 1970 to 2005 (Cavender 1999, pers. comm.; Kibbey 2016, pers. comm.).

These surveys also included extensive searches for populations of Scioto madtoms outside of the type locality in Big Darby Creek (Kibbey 2016, pers. comm.). In addition to fish surveys in the Big Darby Creek watershed, the OEPA has conducted a number of fish studies throughout the Upper, Middle, and Lower Scioto River watershed as part of the agency's Statewide Water Quality Monitoring Program (OEPA 1993a, 1993b, 1999, 2002, 2004, 2006, 2008, 2012, 2019, entire). These surveys have never detected a Scioto madtom.

Time Since Last Detection

No collections of the Scioto madtom have been made since 1957. Given that the extensive fish surveys conducted since 1970 within the species' historical location, as well as along the entire length of Big Darby Creek and in the greater Scioto River watershed, have recorded three other species of madtom but not the Scioto madtom, it is highly unlikely that the Scioto madtom has persisted without detection.

Other Considerations Applicable to the Species' Status

The habitat that once supported the Scioto madtom has been drastically altered, primarily via strong episodic flooding. Although periodic flooding has historically been a part of Big Darby Creek's hydrological regime, many of the original riffles where Scioto madtoms were collected from just downstream of the U.S. Route 104 Bridge to approximately one-half mile upstream have been washed out to the point where they are nearly gone (Kibbey 2009, pers. comm.). Furthermore, pollution sources throughout the Scioto River watershed, including row crop agriculture, development, and urban runoff, have reduced the water quality and suitability of habitat for madtoms (OEPA 2012, pp. 1–2).

III. Analysis

There has been no evidence of the continued existence of the Scioto madtom since 1957. Surveys for the species were conducted annually between 1970 and 2005, at the only known location for the species. Additional surveys in the Big Darby Creek watershed have never found other locations of Scioto madtom. After decades of survey work with no individuals being detected, it is extremely unlikely that the species is extant. Further, available habitat for the species in the only location where it has been documented is now much reduced, which supports the conclusion that the species is likely extinct.

IV. Conclusion

We conclude that the Scioto madtom is extinct and, therefore, should be delisted. This conclusion is based on a lack of detections during numerous surveys conducted for the species and significant alteration of habitat at its known historical location.

Mussels

Flat Pigtoe (Pleurobema marshalli)

I. Background

The flat pigtoe (formerly known as Marshall's pearly mussel), *Pleurobema marshalli,* was listed as endangered on April 7, 1987 (52 FR 11162) primarily due to habitat alteration from a free-flowing riverine system to an impounded system. The recovery plan (''Recovery Plan for Five Tombigbee River Mussels'') was completed on November 14, 1989. A supplemental recovery plan (''Mobile River Basin Aquatic Ecosystem Recovery Plan'') was issued on November 17, 2000. This plan did not replace the existing recovery plan; rather, it was intended to provide additional habitat protection and species husbandry recovery tasks. The species' recovery priority number (RPN) is 5, indicating a high degree of threat and low recovery potential. A 5-year review was announced on November 6, 1991 (56 FR 56882); no changes were proposed for the status of this mussel in that review. Two additional 5-year reviews were completed in 2009 (initiated on September 8, 2006; see 71 FR 53127) and 2015 (initiated on March 25, 2014; see 79 FR 16366); both recommended delisting the flat pigtoe due to extinction. The Service solicited peer review from six experts for both 5-year reviews from State, Federal, university, and museum biologists with known expertise and interest in Mobile River Basin mussels (USFWS 2009, pp. 23–24; USFWS 2015, pp. 15–16); we received responses from three of the peer reviewers, and they concurred with the content and conclusion that the species is presumed extinct.

The flat pigtoe was described in 1927, from specimens collected in the Tombigbee River (USFWS 1989, p. 2). The shell of the flat pigtoe had pustules or welts on the postventral surface, and the adults were subovate in shape and approximately 2.4 inches long and 2 inches wide (USFWS 1989, p. 2). Freshwater mussels of the Mobile River Basin, such as the flat pigtoe, are most often found in clean, fast-flowing water in stable sand, gravel, and cobble gravel substrates that are free of silt (USFWS 2000, p. 81). They are typically found buried in the substrate in shoals and runs (USFWS 2000, p. 81). This type of habitat has been nearly eliminated within the historical range of the species because of the construction of the Tennessee-Tombigbee Waterway in 1984, which created a dredged, straightened navigation channel and a series of impoundments that inundated nearly all riverine mussel habitat (USFWS 1989, p. 1).

The flat pigtoe was historically known from the Tombigbee River from just above Tibbee Creek near Columbus, Mississippi, downstream to Epes, Alabama (USFWS 1989, p. 3). Surveys in historical habitat over the past three decades have failed to locate the species, and all historical habitat is impounded or modified by channelization and impoundments (USFWS 2015, p. 5). No live or freshly dead shells have been observed since the species was listed in 1987 (USFWS 2009, p. 4; USFWS 2015, p. 5).

The Tombigbee River freshwater mussel fauna once consisted of more than 40 species (USFWS 1989, p. 1). Construction of the Tennessee-Tombigbee Waterway adversely impacted some of the species (including flat pigtoe), as evidenced by surveys conducted by the Service, the Tennessee Valley Authority (TVA), the Mobile District Corps of Engineers, and others (USFWS 1989, p. 1). The construction of the Tennessee-Tombigbee Waterway was completed in 1984, and drastically modified the upper Tombigbee River from a riverine to a largely impounded ecosystem from Town Creek near Amory, Mississippi, downstream to the Demopolis Lock and Dam (USFWS 1989, p. 1). Construction of the Waterway adversely impacted mussels and eliminated mussel habitat by physical destruction during dredging, increasing sedimentation, reducing water flow, and suffocating juveniles with sediment (USFWS 1989, p. 6). The only remaining habitat after the Waterway was constructed was in several bendways, resulting from channel cuts. These bendways have all experienced reduced flows and increased sediment accumulation, some with several feet of sediment buildup. Thus, no remaining mussel habitat exists (USFWS 1989, p. 6; USFWS 2015, p. 8). The species is presumed extinct by species experts (USFWS 2015, p. 8).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging and can be affected by a variety of factors, including:

- Size of the mussel (smaller mussels, including juvenile mussels, can be more difficult to find in complex substrates than larger mussels, and survey efforts must be thorough enough to try to detect smaller mussels);
- Behavior of the mussel (some are found subsurface, some at the surface, and some above the surface, and position can vary seasonally (some are more visible during the reproductive phase when they need to come into contact with host fish; therefore, surveys likely need to be conducted during different times of the year to improve detection));
- Substrate composition (it can be easier to see/feel mussels in sand and clay than in gravel or cobble; therefore, surveys need to include all substrate types because mussels can fall off host fish into a variety of substrates);
- Size of river (larger rivers usually have more expansive habitat areas to search and are sometimes deep, requiring specialized survey techniques such as self-contained underwater breathing apparatus (SCUBA));
- Flow conditions (visibility can be affected in very fast-flowing, very shallow, or turbid conditions; therefore, surveys need to use tactile or excavation methods, or delay until turbidity conditions improve);
- Surveyor experience (finding mussels requires a well-developed search image, knowledge of instream habitat dynamics, and ability to identify and distinguish species); and
- Survey methodology and effort (excavation and sifting of stream bottom can detect more mussels than visual or tactile surveys).

All of these challenges are taken into account when developing survey protocols for any species of freshwater mussel, including the flat pigtoe. The flat pigtoe was medium-sized (but juveniles were very small) and most often found buried in sand, gravel, or cobble in fast-flowing runs. However, mussels can be found in suboptimal conditions, depending on where they dropped off of the host fish. Therefore, all of the above-mentioned considerations need to be accounted for when trying to detect this mussel species. Despite detection challenges, many well-planned, comprehensive surveys by experienced State and Federal biologists have not been able to locate extant populations of flat pigtoe in the Tombigbee River (USFWS 2000, p. 81; USFWS 2015, p. 5).

Survey Effort

Prior to listing, freshly dead shells of flat pigtoe were collected in 1980, from the Tombigbee River, Lowndes County, Mississippi (USFWS 2009, pp. 4–5), and a 1984 survey of the Gainesville Bendway of Tombigbee River also found shells of the flat pigtoe (USFWS 1989, p. 4). After listing in 1987, surveys in 1988 and 1990 only found weathered, relict shells of the flat pigtoe below Heflin Dam, thus casting doubt on the continued existence of the species in the Gainesville Bendway (USFWS 1989, p. 4; USFWS 2009, p. 5). Over the past three decades, surveys between 1990–2001, and in 2002, 2003, 2009, 2011, and 2015, of potential habitat throughout the historical range, including intensive surveys of the Gainesville Bendway, where adequate habitat and flows may still occur below the Gainesville Dam on the Tombigbee River in Alabama, have failed to find any live or dead flat pigtoes (USFWS 2000, p. 81).

Time Since Last Detection

The flat pigtoe has not been collected alive since completion of the Tennessee-Tombigbee Waterway in 1984 (USFWS 2000, p. 81; USFWS 2015, p. 5). Mussel surveys within the Tombigbee River drainage during 1984–2015 failed to document the presence of the flat pigtoe (USFWS 2015, p. 8).

Other Considerations Applicable to the Species' Status

Habitat modification is the major cause of decline of the flat pigtoe (USFWS 2000, p. 81). Construction of the Tennessee-Tombigbee Waterway for navigation adversely impacted mussels and their habitat by physical destruction during dredging, increasing sedimentation, reducing water flow, and suffocating juveniles with sediment (USFWS 1989, p. 6). Other threats include channel improvements such as clearing and snagging, as well as sand and gravel mining, diversion of flood flows, and water removal for municipal use. These activities impact mussels by altering the river substrate, increasing sedimentation, changing water flows, and killing individuals via dredging and snagging (USFWS 1989, pp. 6–7). Runoff from fertilizers and pesticides results in algal blooms and excessive growth of other aquatic vegetation, resulting in eutrophication and death of mussels due to lack of oxygen (USFWS 1989, p. 7). The cumulative impacts of habitat degradation due to these factors likely led to flat pigtoe populations becoming scattered and isolated over time. Low population levels increased the difficulty of successful reproduction (USFWS 1989, p. 7). When individuals become scattered, the opportunity for egg fertilization is diminished. Coupled with habitat changes that result in reduced host fish interactions, the spiral of failed reproduction leads to local extirpation and eventual extinction of the species (USFWS 1989, p. 7).

III. Analysis

There has been no evidence of the continued existence of the flat pigtoe for more than three decades. Mussel surveys within the Tombigbee River drainage from 1984–2015 have failed to document the presence of the species (USFWS 2015, p. 8). All known historical habitat has been altered or degraded by impoundments, and the species is presumed extinct by most authorities.

IV. Conclusion

We conclude that the flat pigtoe is extinct and, therefore, should be delisted. This conclusion is based on significant alteration of all known historical habitat and lack of detections during numerous surveys conducted throughout the species' range.

Southern Acornshell (Epioblasma othcaloogensis)

I. Background

The southern acornshell (*Epioblasma othcaloogensis*) was listed as endangered on March 17, 1993 (58 FR 14330), primarily due to habitat modification, sedimentation, and water quality degradation. The recovery plan (''Mobile River Basin Aquatic Ecosystem Recovery Plan'') was completed on November 17, 2000. Critical habitat was initially determined to be not prudent (56 FR 58339, November 19, 1991, p. 58346) and later not determinable (58 FR 14330, March 17, 1993, p. 14338), but in 2001, in response to a legal challenge to the ''not determinable'' finding, the U.S. District Court for the Eastern District of Tennessee issued an order requiring the Service to propose and finalize critical habitat for 11 Mobile River Basin-listed mussels, including the southern acornshell. We subsequently published a final critical habitat rule on July 1, 2004 (69 FR 40084). Two 5-year reviews were completed in 2008 (initiated on June 14, 2005; see 70 FR 34492) and 2018 (initiated on September 23, 2014; see 79 FR 56821), both recommending delisting the southern acornshell due to extinction. We solicited peer review from eight experts for both 5-year reviews from State, Federal, university, nongovernmental, and museum biologists with known expertise and interest in Mobile River Basin mussels (Service 2008, pp. 36–37; Service 2018, p. 15); we received responses from five of the peer reviewers, who all concurred with the content and conclusion that the species is presumed extinct.

The southern acornshell was described in 1857 from Othcalooga Creek in Gordon County, Georgia (58 FR 14330, March 17, 1993, p. 14331). Adult southern acornshells were round to oval in shape and approximately 1.2 inches in length (Service 2000, p. 57). *Epioblasma othcaloogensis* was included as a synonymy of *E. penita* and was considered to be an ectomorph of the latter (58 FR 14330, March 17, 1993, p. 14331). Subsequent research classified the southern acornshell as distinct, belonging in a different subgenus; the species is distinguished from the upland combshell (*E. metastriata*) and the southern combshell (*E. penita*) by its smaller size, round outline, a poorly developed sulcus, and its smooth, shiny, yellow periostracum (58 FR 14330, March 17, 1993, p. 14331). The Service recognizes *Unio othcaloogensis* (Lea) and *Unio modicellus* (Lea) as synonyms of *Epioblasma othcaloogensis.*

The southern acornshell was historically found in shoals in small rivers to small streams in the Coosa and Cahaba river systems (Service 2000, p. 57). As with many of the freshwater mussels in the Mobile River Basin, it was found in stable sand, gravel, cobble substrate in moderate to swift currents. The species had a sexual reproduction strategy and require a host fish to complete the life cycle. Historically, the species occurred in upper Coosa River tributaries and the Cahaba River in Alabama, Georgia, and Tennessee (Service 2000, p. 57). In the upper Coosa River system, the southern acornshell occurred in the Conasauga River, Cowan's Creek, and Othcalooga Creek (58 FR 14330, March 17, 1993, p. 14331). At the time of listing in 1993, the species was estimated to persist in low numbers in streams in the upper Coosa River drainage in Alabama and Georgia, and possibly in the Cahaba River (58 FR 14330, March 17, 1993, p. 14331; Service 2018, p. 6). The southern acornshell was last collected in 1973, from the Conasauga River in Georgia and from Little Canoe Creek, near the Etowah and St. Clair County line, Alabama. It has not been collected from the Cahaba River since the 1930s (Service 2018, p. 5).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging, and can be affected by a variety of factors. Please refer to the Species Detectability section for the flat pigtoe above for the descriptions of these factors. The southern acornshell was small-sized (with very small juveniles) and most often found buried in sand, gravel, or cobble in fast flowing runs. However, mussels can be found in sub-optimal conditions, depending on where they dropped off of the host fish. Therefore, all of the detection considerations need to be accounted for when trying to detect this mussel species. Despite detection challenges, many well-planned, comprehensive surveys by experienced State and Federal biologists have not been able to locate extant populations of southern acornshell (Service 2000, p. 57; Service 2008, p. 20; Service 2018, p. 7).

Survey Effort

Prior to listing, southern acornshell was observed during surveys in the upper Coosa River drainage in Alabama and Georgia in 1966–1968 and in 1971–1973, by Hurd (58 FR 14330, March 17, 1993, p. 14331). Records of the species in the Cahaba River are from surveys at Lily Shoals in Bibb County, Alabama, in 1938, and from Buck Creek (Cahaba River tributary), Shelby County, Alabama, in the early 1900s (58 FR 14330, March 17, 1993, p. 14331). Both the 2008 and 2018 5-year reviews reference multiple surveys by experienced Federal, State, and private biologists—17 survey reports from 1993–2006 and 6 survey reports from 2008–2017—and despite these repeated surveys of historical habitat in both the Coosa and Cahaba River drainages, no living animals or fresh or weathered shells of the southern acornshell have been located (Service 2008, p. 19; Service 2018, p. 6).

Time Since Last Detection

The most recent records for the southern acornshell were from tributaries of the Coosa River in 1966–1968 and 1974, and the Cahaba River in 1938 (58 FR 14330, March 17, 1993, p. 14331; Service 2008, p. 19; Service 2018, p. 5). No living populations of the southern acornshell have been located since the 1970s (Service 2000, p. 57; Service 2008, p. 20; Service 2018, p. 7).

Other Considerations Applicable to the Species' Status

Habitat modification was the major cause of decline of the southern acornshell (Service 2000, p. 57). Other threats included channel improvements such as clearing and snagging, as well as sand and gravel mining, diversion of flood flows, and water removal for municipal use; these activities impacted mussels by alteration of the river substrate, increasing sedimentation, alteration of water flows, and direct mortality from dredging and snagging (Service 2000, p. 6–13). Runoff from fertilizers and pesticides results in algal blooms and excessive growth of other aquatic vegetation, resulting in eutrophication and death of mussels due to lack of oxygen (Service 2000, p. 13). The cumulative impacts of habitat degradation likely lead to the southern acornshell populations becoming scattered and isolated over time. Low population levels mean increased difficulty for successful reproduction (Service 2000, p. 14). When individuals become scattered, the opportunity for a female southern acornshell to successfully fertilize eggs is diminished, and the spiral of failed reproduction leads to local extirpation and eventual extinction of the species (Service 2000, p. 14).

III. Analysis

There has been no evidence of the continued existence of the southern acornshell for over five decades; the last known specimens were collected in the early 1970s. When listed in 1993, it was thought that the southern acornshell was likely to persist in low numbers in the upper Coosa River drainage and, possibly, in the Cahaba River. Numerous mussel surveys have been completed within these areas, as well as other areas within the historical range of the species since the listing, with no success. Although other federally listed mussels have been found by mussel experts during these surveys, no live or freshly dead specimens of the southern acornshell have been found (Service 2018, p. 7). The species is presumed extinct.

IV. Conclusion

We conclude that the southern acornshell is extinct and, therefore, should be delisted. This conclusion is based on significant alteration of known historical habitat and lack of detections during numerous surveys conducted throughout the species' range.

Stirrupshell (Quadrula stapes)

I. Background

The stirrupshell (*Quadrula stapes*) was listed as endangered on April 7, 1987 (52 FR 11162), primarily due to habitat alteration from a free-flowing riverine system to an impounded system. The recovery plan (''Recovery Plan for Five Tombigbee River Mussels'') was completed on November 14, 1989. A supplemental recovery plan (''Mobile River Basin Aquatic Ecosystem Recovery Plan'') was completed on November 17, 2000. This plan did not replace the existing recovery plan; rather, it was intended to provide additional habitat protection and species husbandry recovery tasks. A 5-year review was announced on November 6, 1991 (56 FR 56882); no changes were proposed for the status of the stirrupshell in that review. Two additional 5-year reviews were completed in 2009 (initiated on September 8, 2006; see 71 FR 53127) and 2015 (initiated on March 25, 2014; see 79 FR 16366); both recommended delisting the stirrupshell due to extinction. We solicited peer review from six experts for both 5-year reviews from State, Federal, university, and museum biologists with known expertise and interest in Mobile River Basin mussels (Service 2009, pp. 23–24; Service 2015, pp. 15–16); we received responses from three of the peer reviewers, and they concurred with the content and conclusion that the species is presumed extinct.

The stirrupshell was described as *Unio stapes* in 1831, from the Alabama River (Stansbery 1981, entire). Other synonyms are *Margarita* (*Unio*) *stapes* in 1836, *Margaron* (*Unio*) *stapes* in 1852, *Quadrula stapes* in 1900, and *Orthonymus stapes* in 1969 (Service 1989, pp. 2–3). Adult stirrupshells were quadrate in shape and reached a size of approximately 2 inches long and 2 inches wide. The stirrupshell differed from other closely related species by the presence of a sharp posterior ridge and truncated narrow rounded point posteriorly on its shell, and it had a tubercled posterior surface (Service 1989, p. 3; Service 2000, p. 85). Freshwater mussels of the Mobile River Basin, such as the stirrupshell, are most often found in clean, fast-flowing water in stable sand, gravel, and cobble gravel substrates that are free of silt (Service 2000, p. 85). They are typically found buried in the substrate in runs (Service 2000, p. 85). This type of habitat has been nearly eliminated in the Tombigbee River because of the construction of the Tennessee-Tombigbee Waterway, which created a dredged, straightened navigation channel and series of impoundments that inundated much of the riverine mussel habitat (Service 1989, p. 1).

The stirrupshell was historically found in the Tombigbee River from Columbus, Mississippi, downstream to Epes, Alabama; the Sipsey River, a tributary to the Tombigbee River in Alabama; the Black Warrior River in Alabama; and the Alabama River (Service 1989, p. 3). Surveys in historical habitat over the past three decades have failed to locate the species, as all historical habitat is impounded or modified by channelization and impoundments (Tombigbee and Alabama Rivers) or impacted by sediment and nonpoint pollution (Sipsey and Black Warrior Rivers) (Service 1989, p. 6; Service 2000, p. 85; Service 2015, p. 5). No live or freshly dead shells have been observed since the species was listed in 1987 (Service 2009, p. 6; Service 2015, p. 7). A freshly dead shell was last collected from the lower Sipsey River in 1986 (Service 2000, p. 85).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging, and can be affected by a variety of factors. Please refer to the Species Detectability section for the flat pigtoe above for the descriptions of these factors. The stirrupshell was medium-sized (with very small juveniles) and most often found buried in sand, gravel, or cobble in fast flowing runs. However, mussels can be found in sub-optimal conditions, depending on where they dropped off of the host fish. Therefore, all of the detection considerations need to be accounted for when trying to detect this mussel species. Despite detection challenges, many well-planned, comprehensive surveys by experienced State and Federal biologists have not been able to locate extant populations of stirrupshell (Service 1989, pp. 3–4; Service 2000, p. 85; Service 2015, pp. 7–8).

Survey Effort

Prior to listing in 1987, stirrupshell was collected in 1978, from the Sipsey River, and a 1984 and 1986 survey of the Sipsey River found freshly dead shells; a 1984 survey of the Gainesville Bendway of Tombigbee River found freshly dead shells of the stirrupshell (Service 1989, p. 4; Service 2000, p. 85). After listing, surveys in 1988 and 1990 only found weathered, relict shells of the stirrupshell from the Tombigbee River at the Gainesville Bendway and below Heflin Dam, which cast doubt on the continued existence of the species in the mainstem Tombigbee River (Service 1989, p. 4; Service 2009, p. 6). Over the past three decades, repeated surveys (circa 1988, 1998, 2001, 2002, 2003, 2006, 2011) of unimpounded habitat in the Sipsey and Tombigbee Rivers, including intensive surveys of the Gainesville Bendway, have failed to find any evidence of stirrupshell (Service 2009, p. 6; Service 2015, p. 7). The stirrupshell was also known from the Alabama River; however, over 92 hours of dive bottom time were expended searching appropriate habitats for imperiled mussel species between 1997–2007 without encountering the species (Service 2009, p. 6), and a survey of the Alabama River in 2011 also did not find stirrupshell (Service 2015, p. 5). Surveys of the Black Warrior River in 1993 and from 2009–2012 (16 sites) focused on finding federally listed and State conservation concern priority mussel species but did not find any stirrupshells (Miller 1994, pp. 9, 42; McGregor *et al.* 2009, p. 1; McGregor *et al.* 2013, p. 1).

Time Since Last Detection

The stirrupshell has not been collected alive since the Sipsey River was surveyed in 1978 (Service 1989, p. 4); one freshly dead shell was last collected from the Sipsey River in 1986 (Service 2000, p. 85). In the Tombigbee River, the stirrupshell has not been collected alive since completion of the Tennessee-Tombigbee Waterway in 1984 (Service 2015, p. 7). Mussel surveys within the Tombigbee River drainage during 1984–2015 failed to document the presence of the stirrupshell (Service 2015, p. 8). The stirrupshell has not been found alive in the Black Warrior River or the Alabama River since the early 1980s (Service 1989, p. 3).

Other Considerations Applicable to the Species' Status

Because the stirrupshell occurred in similar habitat type and area as the flat pigtoe, it faced similar threats. Please refer to the discussion for the flat pigtoe for more information.

III. Analysis

There has been no evidence of the continued existence of the stirrupshell for nearly four decades; the last live individual was observed in 1978 and the last freshly dead specimen was from 1986. Mussel surveys within the Tombigbee River drainage (including the Sipsey and Black Warrior tributaries) from 1984–2015, and the Alabama River from 1997–2007 and in 2011, have failed to document the presence of the species (Service 2015, pp. 5, 8). All known historical habitat has been altered or degraded by impoundments and nonpoint source pollution, and the species is presumed extinct by most authorities.

IV. Conclusion

We conclude that the stirrupshell is extinct and, therefore, should be delisted. This conclusion is based on significant alteration of all known historical habitat and lack of detections during numerous surveys conducted throughout the species' range.

Upland Combshell (Epioblasma metastriata)

I. Background

The upland combshell, *Epioblasma metastriata,* was listed as endangered on March 17, 1993 (58 FR 14330), primarily due to habitat modification, sedimentation, and water quality degradation. The recovery plan (''Mobile River Basin Aquatic Ecosystem Recovery Plan'') was completed on November 17, 2000. Critical habitat was initially determined to be not prudent (56 FR 58339, November 19, 1991, p. 58346) and later not determinable (58 FR 14330, March 17, 1993, p. 14338), but in 2001, in response to a legal challenge to the ''not determinable'' finding, the U.S. District Court for the Eastern District of Tennessee issued an order requiring the Service to propose and finalize critical habitat for 11 Mobile River Basin-listed mussels, including the upland combshell. We subsequently published a final critical habitat rule on July 1, 2004 (69 FR 40084). Two 5-year reviews were completed in 2008 (initiated on June 14, 2005; see 70 FR 34492) and 2018 (initiated on September 23, 2014; see 79 FR 56821), both recommending delisting the upland combshell due to extinction. We solicited peer review from eight experts for both 5-year reviews from State, Federal, university, nongovernmental, and museum biologists with known expertise and interest in Mobile River Basin mussels (Service 2008, pp. 36–37; Service 2018, p. 15); we received responses from five of the peer reviewers, who concurred with our conclusion that the species is presumed extinct.

The upland combshell was described in 1838, from the Mulberry Fork of the Black Warrior River near Blount Springs, Alabama (58 FR 14330, March 17, 1993, p. 14331). Adult upland combshells were rhomboidal to quadrate in shape and were approximately 2.4 inches in length (58 FR 14330, March 17, 1993, pp. 14330–14331). The upland combshell was considered to be a variation of the southern combshell (= penitent mussel, *Epioblasma penita*), and they were considered synonyms of each other (58 FR 14330, March 17, 1993, p. 14331). However, subsequent research identified morphological differences between the two, and both species were considered to be valid taxa; the upland combshell was distinguished from the southern combshell by the diagonally straight or gently rounded posterior margin of the latter, which terminated at the post-ventral extreme of the shell (58 FR 14330, March 17, 1993, p. 14331). We recognize *Unio metastriatus* and *Unio compactus* as synonyms of *Epioblasma metastriata* (58 FR 14330, March 17, 1993, p. 14331).

The upland combshell was historically found in shoals in rivers and large streams in the Black Warrior, Cahaba, and Coosa River systems above the Fall Line in Alabama, Georgia, and Tennessee (Service 2000, p. 61). As with many of the freshwater mussels in the Mobile River Basin, it was found in stable sand, gravel, and cobble in moderate to swift currents. The historical range included the Black Warrior River and tributaries (Mulberry Fork and Valley Creek); Cahaba River and tributaries (Little Cahaba River and Buck Creek); and the Coosa River and tributaries (Choccolocco Creek and Etowah, Conasauga, and Chatooga Rivers) (58 FR 14330, March 17, 1993, p. 14331). At the time of listing in 1993, the species was estimated to be restricted to the Conasauga River in Georgia, and possibly portions of the upper Black Warrior and Cahaba River drainages (58 FR 14330, March 17, 1993, p. 14331; Service 2008, p. 19). The upland combshell was last collected in the Black Warrior River drainage in the early 1900s; in the Coosa River drainage in 1986, from the Conasauga River near the Georgia/Tennessee State line; and the Cahaba River drainage in the early 1970s (58 FR 14330, March 17, 1993, p. 14331; Service 2000, p. 61; Service 2018, p. 5).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging, and can be affected by a variety of factors. Please refer to the Species Detectability section for the flat pigtoe above for the descriptions of these factors. The Upland combshell was small-sized (with very small juveniles) and most often found buried in sand, gravel, or cobble in fast flowing runs. However, mussels can be found in sub-optimal conditions, depending on where they dropped off of the host fish. Therefore, all of the detection considerations need to be accounted for when trying to detect this mussel species. Despite detection challenges, many well-planned, comprehensive surveys by experienced State and Federal biologists have not been able to locate extant populations of upland combshell (Service 2008, p. 19; Service 2018, p. 5)

Survey Effort

Prior to listing in 1993, upland combshell was observed during surveys in the Black Warrior River drainage in the early 1900s; repeated surveys in this drainage in 1974, 1980–1982, 1985, and 1990 did not encounter the species (58 FR 14330, March 17, 1993, p. 14331). The upland combshell was observed in the Cahaba River drainage in 1938 and in 1973, but a 1990 survey failed to find the species in the Cahaba River drainage (58 FR 14330, March 17, 1993, p. 14331). The species was observed in the upper Coosa River drainage in Alabama and Georgia in 1966–1968, but not during 1971–1973 surveys; a single specimen was collected in 1988 from the Conasauga River (58 FR 14330, March 17, 1993, p. 14331). Both the 2008 and 2018 5-year reviews reference multiple surveys by experienced Federal, State, and private biologists—18 survey reports from 1993–2006 and 10 survey reports from 2008–2017—and despite these repeated surveys of historical habitat in the Black Warrior, Cahaba, and Coosa River drainages, no living animals or fresh or weathered shells of the upland combshell have been located (Service 2008, p. 19; Service 2018, p. 5).

Time Since Last Detection

The most recent records for the upland combshell are many decades old: From tributaries of the Black Warrior in early 1900s, from the Cahaba River drainage in the early 1970s, and from the Coosa River drainage in the mid-1980s (58 FR 14330, March 17, 1993, p. 14331; Service 2008, p. 19; Service 2018, p. 5). No living populations of the upland combshell have been located since the mid-1980s (Service 2000, p. 61; Service 2008, p. 20; Service 2018, p. 7).

Other Considerations Applicable to the Species' Status

Because the upland combshell occurred in similar habitat type and area as the southern acornshell, it faced similar threats. Please refer to the discussion for the southern acornshell for more information on any other overarching consideration.

III. Analysis

There has been no evidence of the continued existence of the upland combshell for over three decades; the last known specimens were collected in the late-1980s. When listed, it was thought that the upland combshell was likely restricted to the Conasauga River in Georgia, and possibly portions of the upper Black Warrior and Cahaba River drainages. Numerous mussel surveys have been completed within these areas, as well as other areas within the historical range of the species since the late-1980s, with no success. Although other federally listed mussels have been found by mussel experts during these surveys, no live or freshly dead specimens of the upland combshell have been found (Service 2018, p. 7). The species is presumed extinct.

IV. Conclusion

We conclude that the upland combshell is extinct and, therefore, should be delisted. This conclusion is based on significant alteration of known historical habitat and lack of detections during numerous surveys conducted throughout the species' range.

Green Blossom (Epioblasma torulosa gubernaculum)

I. Background

The green blossom (pearly mussel), *Epioblasma torulosa gubernaculum,* was listed as endangered on June 14, 1976 (41 FR 24062), and the final recovery plan was issued on July 9, 1984. At the time of listing, the single greatest factor contributing to the species' decline was the alteration and destruction of stream habitat due to impoundments. Two 5-year reviews were completed in 2007 (initiated on September 20, 2005; see 70 FR 55157) and 2017 (initiated on March 25, 2014; see 79 FR 16366); both reviews recommended delisting due to extinction. For the 2017 5-year review, the Service solicited peer review from eight peer reviewers including Federal and State biologists with known expertise and interest in blossom pearly mussels (the green blossom was one of four species assessed in this 5-year review). All eight peer reviewers indicated there was no new information on the species, or that the species was presumed extirpated or extinct from their respective State(s) (USFWS 2017, pp. 8–9).

The green blossom was described in 1865, with no type locality given for the species. However, all historical records indicate the species was restricted to the upper headwater tributary streams of the Tennessee River above Knoxville (USFWS 1983, pp. 1–2). The recovery plan described the green blossom as a medium-sized mussel with a lifespan up to 50 years. The shell outline was irregularly ovate, elliptical, or obovate. The green blossom was a sexually dimorphic, medium-sized species. Females were generally larger than the males and possessed a large, flattened, rounded swelling or expansion that extends from the middle of the base to the upper part of the posterior end. A comprehensive description of shell anatomy is provided in our 5-year review and supporting documents (Parmalee and Bogan 1998, pp. 104–107).

The green blossom was always extremely rare and never had a wide distribution (USFWS 1984, p. 9). Freshwater mussels found within the Cumberland rivers and tributary streams, such as the green blossom, are most often observed in clean, fast-flowing water in substrates that contain relatively firm rubble, gravel, and sand substrates swept free from siltation (USFWS 1984, p. 5). They are typically found buried in substrate in shallow riffle and shoal areas. This type of habitat has been nearly eliminated by impoundment of the Tennessee and Cumberland Rivers and their headwater tributary streams (USFWS 1984, p. 9).

The genus *Epioblasma* as a whole has suffered extensively because members of this genus are riverine, typically found only in streams that are shallow with sandy-gravel substrate and rapid currents (Stansbery 1972, pp. 45–46). Eight species of *Epioblasma* were presumed extinct at the time of the recovery plan, primarily due to impoundments, siltation, and pollution (USFWS 1984, p. 6).

Stream impoundment affects species composition by eliminating those species not capable of adapting to reduced flows and altered temperatures. Tributary dams typically have storage impoundments with cold water discharges and sufficient storage volume to cause the stream below the dam to differ significantly from pre-impoundment conditions. These hypolimnial discharges result in altered temperature regimes, extreme water level fluctuations, reduced turbidity, seasonal oxygen deficits, and high concentrations of certain heavy metals (TVA 1980, entire).

Siltation within the range of the green blossom, resulting from strip mining, coal washing, dredging, farming, and road construction, also likely severely affected the species. Since most freshwater mussels are riverine species that require clean, flowing water over stable, silt-free rubble, gravel, or sand shoals, smothering caused by siltation can be detrimental. The recovery plan indicated that siltation associated with poor agricultural practices and deforestation was probably the most significant factor impacting mussel communities (Fuller 1977, as cited in USFWS 1984, p. 12). The recovery plan also documented numerous coal operations within the range of the green blossom that have caused increased silt runoff, including in the Clinch River, where the last live specimen was collected in 1982 (USFWS 1984, pp. 12–13). Pollution, primarily from wood pulp, paper mills, and other industries, has also severely impacted many streams within the historical range of the species.

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging, and can be affected by a variety of factors. Please refer to the Species Detectability section for the flat pigtoe above for the descriptions of these factors. The green blossom was a medium-sized mussel most often found buried in substrate in shallow riffle and shoal areas. However, mussels can be found in sub-optimal conditions, depending on where they dropped off of the host fish.

Survey Effort

As of 1984, freshwater mussel surveys by numerous individuals had failed to document any living populations of green blossom in any Tennessee River tributary other than the Clinch River. The recovery plan cites several freshwater mussel surveys (which took place between 1972 and 2005) of the Powell River; North, South, and Middle Forks of the Holston River; Big Moccasin Creek; Copper Creek; Nolichucky River; and French Broad River, all of which failed to find living or freshly dead green blossom specimens (USFWS 1984, p. 5). Annual surveys continue to be conducted in the Clinch River since 1972. Biologists conducting those surveys have not reported live or freshly dead individuals of the green blossom (Ahlstedt *et al.* 2016, entire; Ahlstedt *et al.* 2017, entire; Jones *et al.* 2014, entire; Jones *et al.* 2018, entire).

Time Since Last Detection

The last known record for the green blossom was a live individual collected in 1982, in the Clinch River at Pendleton Island, Virginia.

III. Analysis

Habitat within the historical range of the green blossom has been significantly altered by water impoundments, siltation, and pollution, including at Pendleton Island on the Clinch River, the site of the last known occurrence of the species (Jones *et al.* 2018, pp. 36–56). The last known collection of the species was 38 years ago, and numerous surveys have been completed within the known range of the species over these 38 years. Although other federally listed mussels have been found by these experts during these surveys, no live or freshly dead specimens of the green blossom have been found (Ahlstedt *et al.* 2016, pp. 1–18; Ahlstedt *et al.* 2017, pp. 213–225). Mussel experts conclude that the species is likely to be extinct.

IV. Conclusion

We conclude the green blossom is extinct and, therefore, should be delisted. This conclusion is based on lack of detections during surveys and searches conducted throughout the species' range since the green blossom was last observed in 1982, and the amount of significant habitat alteration that has occurred within the range of the species, rendering most of the species' historical habitat unlikely to support the species.

Tubercled Blossom (Epioblasma torulosa torulosa)

I. Background

The tubercled blossom (pearly mussel), *Epioblasma torulosa torulosa,* was listed as endangered on June 14, 1976 (41 FR 24062), and the final recovery plan was completed on January 25, 1985. At the time of listing, the greatest factor contributing to the species' decline was the alteration and destruction of stream habitat due to impoundments. Two 5-year reviews were completed in 1991 (initiated on November 6, 1991; see 56 FR 56882) and 2011 (initiated on September 20, 2005; see 70 FR 55157); both reviews recommended the species maintain its endangered status, although the 2011 review did conclude the species was likely extinct. The most recent 5-year review was completed in 2017 (initiated on March 25, 2014; see 79 FR 16366), indicated that the species was presumed extinct, and recommended delisting. The Service solicited peer review from three peer reviewers for the 2017 5-year review from Federal and State biologists with known expertise and interest in blossom pearly mussels (the tubercled blossom was one of four species assessed in this 5-year review). All three peer reviewers indicated there was no new information on the species, all populations of the species were extirpated from their respective States, and the species was presumed extinct.

The tubercled blossom was described as *Amblema torulosa* from the Ohio and Kentucky Rivers (Rafinesque 1820; referenced in USFWS 1985, p. 2). All records for this species indicate it was widespread in the larger rivers of the eastern United States and southern Ontario, Canada (USFWS 1985, p. 2). Records for this species included the Ohio, Kanawha, Scioto, Kentucky, Cumberland, Tennessee, Nolichucky, Elk, and Duck Rivers (USFWS 1985, pp. 3–6). Historical museum records gathered subsequently add the Muskingum, Olentangy, Salt, Green, Barren, Wabash, White, East Fork White, and Hiwassee Rivers to its range (Service 2011, p. 5). The total historical range includes the States of Alabama, Illinois, Indiana, Kentucky, Ohio, Tennessee, and West Virginia. This species was abundant in archaeological sites along the Tennessee River in extreme northwestern Alabama, making it likely that the species also occurred in adjacent northeastern Mississippi where the Tennessee River borders that State (Service 2011, p. 5).

The tubercled blossom was medium-sized, reaching about 3.6 inches (9.1 centimeters) in shell length, and could live as long as 50 years or more. The shell was irregularly egg-shaped or elliptical, slightly sculptured, and corrugated with distinct growth lines. The outer surface was smooth and shiny; was tawny, yellowish-green, or straw-colored; and usually had numerous green rays (Parmalee and Bogan 1980, pp. 22–23).

The genus *Epioblasma* as a whole has suffered extensively because members of this genus are characteristic riffle or shoal species, typically found only in streams that are shallow with sandy-gravel substrate and rapid currents (Parmalee and Bogan 1980, pp. 22–23). Eight species of *Epioblasma* were presumed extinct at the time of the 1985 recovery plan. The elimination of these species has been attributed to impoundments, barge canals, and other flow alteration structures that have eliminated riffle and shoal areas (USFWS 1985, p. 1).

The single greatest factor contributing to the decline of the tubercled blossom is the alteration and destruction of stream habitat due to impoundments for flood control, navigation, hydroelectric power production, and recreation. Siltation is another factor that has severely affected the tubercled blossom. Increased silt transport into waterways due to strip mining, coal washing, dredging, farming, logging, and road construction increased turbidity and consequently reduced the depth of light penetration and created a blanketing effect on the substrate. The 1985 recovery plan documented numerous coal operations within the range of the tubercled blossom that were causing increased silt runoff. A third factor is the impact caused by various pollutants. An increasing number of streams throughout the blossom's range receive municipal, agricultural, and industrial waste discharges.

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging, and can be affected by a variety of factors. Please refer to the Species Detectability section for the flat pigtoe above for the descriptions of these factors. The tubercled blossom was a large-river species most often found inhabiting parts of those rivers that are shallow with sandy-gravel substrate and rapid currents. However, mussels can be found in sub-optimal conditions, depending on where they dropped off of the host fish.

Survey Effort

All three rivers where the species was last located have been extensively sampled in the intervening years without further evidence of this species' occurrence, including Kanawha River, Nolichucky River, and Green River (Service 2011, p. 5).

Based on this body of survey information in large rivers in the Ohio River system, investigators have been considering this species as possibly extinct since the mid-1970s. Probably the best reach of potential habitat remaining may be in the lowermost 50 miles of the free-flowing portion of the Ohio River, in Illinois and Kentucky. This reach is one of the last remnants of large-river habitat remaining in the entire historical range of the tubercled blossom. In our 2011 5-year review for the tubercled blossom, we hypothesized that this mussel might be found in this stretch of the Ohio River. Unfortunately, mussel experts have not reported any new collections of the species (USFWS 2017, p. 8). Additionally, State biologists have conducted extensive surveys within the Kanawha Falls area of the Kanawha River since 2005, and have found no evidence that the tubercled blossom still occurs there (USFWS 2017, p. 4). This species is presumed extirpated.

Time Since Last Detection

The last individuals were collected live or freshly dead in 1969, in the Kanawha River, West Virginia, below Kanawha Falls; in 1968, in the Nolichucky River, Tennessee; and in 1963, in the Green River, Kentucky.

III. Analysis

The tubercled blossom has not been seen since 1969, despite extensive survey work in nearly all of the rivers of historical occurrence, prompting many investigators to consider this species as possibly extinct. According to the last two 5-year reviews, experts indicate that the species is presumed extinct throughout its range.

IV. Conclusion

We conclude the tubercled blossom is extinct and, therefore, should be delisted. This conclusion is based on the lack of detections during surveys and searches conducted throughout the species' range since the tubercled blossom was last sighted in 1969, and the significant habitat alteration that has occurred within the range of the species, rendering most of the species' habitat unable to support the life-history needs of the species.

Turgid Blossom (Epioblasma turgidula)

I. Background

The turgid blossom (pearly mussel), *Epioblasma turgidula,* was listed as endangered on June 14, 1976 (41 FR 24062), and the final recovery plan was completed on January 25, 1985 (USFWS 1985). At the time of listing, the single greatest factor contributing to the species' decline was the alteration and destruction of stream habitat due to impoundments. Two 5-year reviews were completed in 2007 (initiated on September 20, 2005; see 70 FR 55157) and 2017 (initiated on August 30, 2016; see 81 FR 59650); both reviews recommended delisting due to extinction. The Service solicited peer review from eight peer reviewers for the 2017 5-year review from Federal and State biologists with known expertise and interest in blossom pearly mussels (the turgid blossom was one of four species assessed in this 5-year review). All eight peer reviewers indicated there was no new information on the species, all populations of the species were extirpated from their respective States, and the species was presumed extinct.

The turgid blossom was described (Lea 1858; referenced in USFWS 1985, p. 2) as *Unio turgidulus* from the Cumberland River, Tennessee, and the Tennessee River, Florence, Alabama. According to the recovery plan, this species was historically relatively widespread with a disjunct distribution occurring in both the Cumberlandian and Ozarkian Regions (USFWS 1985, p. 7). It has been reported from the Tennessee River and tributary streams including Shoal and Bear Creeks, and Elk, Duck, Holston, Clinch, and Emory Rivers (Ortmann 1918, 1924, 1925; Stanberry 1964, 1970, 1971, 1976a; Johnson 1978, as cited in USFWS 2017, entire). Additional records are reported from the Cumberland River (Ortmann 1918; Clench and van der Schalie 1944; Johnson 1978, as cited in USFWS 2017, entire) and from the Ozark Mountain Region, including Spring Creek, and Black and White Rivers (Simpson 1914; Johnson 1978, as cited in USFWS 2017, entire).

The turgid blossom was a medium-river, Cumberlandian-type mussel that was also reported from the Ozarks. These mussels could live as long as 50 years or more. The species was strongly dimorphic; males and females differed in shape and structure. This species seldom exceeded 1.6 inches (4.1 centimeters) in shell length. Shells of the male tended to be more elliptical or oval, while females tended to be more rounded. Valves were inequilateral, solid, and slightly inflated. The outer shell was shiny yellowish-green with numerous fine green rays over the entire surface. The shell surface was marked by irregular growth lines that are especially strong on females. The inner shell surface was bluish-white (Parmalee and Bogan 1980, pp. 22–23).

The genus *Epioblasma* as a whole has suffered extensively because members of this genus are characteristic riffle or shoal species, typically found only in streams that are shallow with sandy-gravel substrate and rapid currents (Parmalee and Bogan 1980, pp. 22–23). Eight species of *Epioblasma* were presumed extinct at the time of the 1985 recovery plan. The elimination of these species has been attributed to impoundments, barge canals, and other flow alteration structures that have eliminated riffle and shoal areas (USFWS 1985, p. 1). The last known population of the turgid blossom occurred in the Duck River and was collected in 1972, at Normandy (Ahlstedt 1980, pp. 21–23). Field notes associated with this collection indicate that it was river-collected 100 yards above an old iron bridge. Water at the bridge one mile upstream was very muddy, presumably from dam construction above the site (Ahlstedt *et al.* 2017, entire). Additionally, surveys in the 1960s of the upper Cumberland Basin indicated an almost total elimination of the genus *Epioblasma,* presumably due to mine wastes (Neel and Allen 1964, as cited in USFWS 1985, p. 10).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging, and can be affected by a variety of factors. Please refer to the Species Detectability section for the flat pigtoe above for the descriptions of these factors. The turgid blossom was a small-sized mussel most often found buried in substrate in shallow riffle and shoal areas. However, mussels can be found in sub-optimal conditions, depending on where they dropped off of the host fish.

Survey Effort

This species has not been found in freshwater mussel surveys conducted on the Duck River since the time of the Normandy Dam construction (Ahlstedt 1980, pp. 21–23), nor has it been reported from any other stream or river system. The most recent 5-year review notes that the Tennessee Wildlife Resources Agency had completed or funded surveys (1972–2005) for blossom pearly mussels in the Cumberland, Tennessee, Clinch, Duck, Elk, Emory, Hiwassee, Little, and Powell Rivers, yet there were no recent records of turgid blossom (USFWS 2017, p. 4). Surveys in the Ozarks have not observed the species since the early 1900s (USFWS 1985, p. 7).

Time Since Last Detection

The last known collection of the turgid blossom was a freshly dead specimen found in the Duck River, Tennessee, in 1972 by a biologist with the TVA. The species has not been seen in the Ozarks since the early 1900s (USFWS 1985, p. 7).

III. Analysis

Habitat within the historical range of the turgid blossom has been significantly altered by water impoundments, siltation, and pollution. The last known collection of the species was more than 45 years ago. Mussel experts conclude that the species is likely to be extinct. Numerous surveys have been completed within the known range of the species over the years. Although other federally listed mussels have been found by experts during these surveys, no live or freshly dead specimens of the turgid blossom have been found.

IV. Conclusion

We conclude the turgid blossom is extinct and, therefore, should be delisted. This conclusion is based on the lack of detections during surveys and searches conducted throughout the species' range since the turgid blossom was last sighted in 1972, and the significant habitat alteration that occurred within the range of the species, rendering most of the species' habitat unlikely to support the species.

Yellow Blossom (Epioblasma florentina florentina)

I. Background

The yellow blossom (pearly mussel), *Epioblasma florentina florentina,* was listed as endangered on June 14, 1976 (41 FR 24062), and the final recovery plan was completed on January 25, 1985. At the time of listing, the single greatest factor contributing to the species' decline was the alteration and destruction of stream habitat due to impoundments. Two 5-year reviews were completed in 2007 (initiated on September 20, 2005; see 70 FR 55157) and 2017 (initiated on March 25, 2014; see 79 FR 16366); both reviews recommended delisting due to extinction. The Service solicited peer review from eight peer reviewers for the 2017 5-year review from Federal and State biologists with known expertise and interest in blossom pearly mussels (the yellow blossom was one of four species assessed in this 5-year review). All eight peer reviewers indicated there was no new information on the species, all populations of the species were extirpated from their respective States, and the species was presumed extinct.

The yellow blossom was described (Lea 1857; referenced in USFWS 1985, pp. 2–3) as *Unio florentinus* from the Tennessee River, Florence and Lauderdale Counties, Alabama, and the Cumberland River, Tennessee. According to the recovery plan, this species was a Cumberlandian-type mussel historically widespread in the Tennessee and Cumberland Rivers and tributaries to the Tennessee River. The yellow blossom was reported from Hurricane, Limestone, Bear, and Cypress Creeks, all tributary streams to the Tennessee River in northern Alabama (Ortmann 1925 p. 362; Bogan and Parmalee 1983, p. 23). This species was also reported from larger tributary streams of the lower and upper Tennessee River, including the Flint, Elk, and Duck Rivers (Isom *et al.* 1973, p. 439; Bogan and Parmalee 1983, pp. 22–23) and the Holston, Clinch, and Little Tennessee Rivers (Ortmann 1918, pp. 614–616). Yellow blossoms apparently occurred throughout the Cumberland River (Wilson and Clark 1914, p. 46; Ortmann 1918, p. 592; Neel and Allen 1964, p. 448).

The yellow blossom seldom achieved more than 2.4 inches (6 centimeters) in length. The slightly inflated valves were of unequal length, and the shell surface was marked by uneven growth lines. The shell was a shiny honey-yellow or tan with numerous green rays uniformly distributed over the surface. The inner shell surface was bluish-white (Bogan and Parmalee 1983, pp. 22–23).

The genus *Epioblasma* as a whole has suffered extensively because members of this genus are characteristic riffle or shoal species, typically found only in streams that are shallow with sandy-gravel substrate and rapid currents (Bogan and Parmalee 1983, pp. 22–23). Eight species of *Epioblasma* were presumed extinct at the time of the 1985 recovery plan. The elimination of these species has been attributed to impoundments, barge canals, and other flow alteration structures that have eliminated riffle and shoal areas (USFWS 1985, p. 1).

The single greatest factor contributing to the decline of the yellow blossom, not only in the Tennessee Valley but in other regions as well, is the alteration and destruction of stream habitat due to impoundments for flood control, navigation, hydroelectric power production, and recreation. Siltation is another factor that has severely affected the yellow blossom. Increased silt transport into waterways due to strip mining, coal washing, dredging, farming, logging, and road construction increased turbidity and consequently reduced light penetration, creating a blanketing effect on the substrate. The 1985 recovery plan documented numerous coal operations within the range of the yellow blossom. A third factor is the impact caused by various pollutants. An increasing number of streams throughout the mussel's range receive municipal, agricultural, and industrial waste discharges (USFWS 2017, p. 5).

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Detection of rare, cryptic, benthic-dwelling animals like freshwater mussels is challenging, and can be affected by a variety of factors. Please refer to the Species Detectability section for the flat pigtoe above for the descriptions of these factors. The yellow blossom was a small-sized mussel most often found buried in substrate in shallow riffle and shoal areas. However, mussels can be found in sub-optimal conditions, depending on where they dropped off of the host fish.

Survey Effort

Since the last recorded collections in the mid-1960s, numerous mussel surveys (1872–2005) have been done by mussel biologists from the TVA, Virginia Tech, U.S. Geological Survey, and others in rivers historically containing the species. Biologists conducting those surveys have not reported live or freshly dead individuals of the yellow blossom.

Time Since Last Detection

This species was last collected live from Citico Creek in 1957, and the Little Tennessee River in the 1966 (Bogan and Parmalee, 1983, p. 23), and archeological shell specimens were collected from the Tennessee and Cumberland Rivers between 1976–1979 (Parmalee *et al.* 1980, entire).

III. Analysis

Habitat within the historical range of the yellow blossom has been significantly altered by water impoundments, siltation, and pollution. The last known collection of the species was over 50 years ago. Mussel experts conclude that the species is likely to be extinct. Numerous surveys have been completed within the known range of the species over the years. Although other federally listed mussels have been found by these experts during these surveys, no live or freshly dead specimens of the yellow blossom have been found.

IV. Conclusion

We conclude the yellow blossom is extinct and, therefore, should be delisted. This conclusion is based on lack of detections during surveys conducted throughout the species' range since the yellow blossom was last sighted in the mid-1960s and on the significant habitat alteration that occurred within the range of the species, rendering most of the species' habitat unlikely to support the species.

Plants

Phyllostegia glabra var. lanaiensis

I. Background

Phyllostegia glabra var. *lanaiensis* was listed as endangered on September 20, 1991 (56 FR 47686), and was included in the Lanai plant cluster recovery plan in 1995 (USFWS 1995).

At the time of listing, no wild individuals had been seen since 1914, although there was one questionable sighting from the 1980s that was later considered to be *P. glabra* var. *glabra* (USFWS 1995; 2012). Threats included habitat degradation and herbivory by feral ungulates, the establishment of ecosystem-altering invasive plant species, and the consequences of small population sizes (low numbers) (USFWS 1995). In 2000, designation of critical habitat was considered not prudent for *P. glabra* var. *lanaiensis* because this plant had not been observed in the wild in over 20 years and no viable genetic material was available for recovery efforts (65 FR 82086; December 27, 2000). Two 5-year status reviews have been completed; the 2012 review (initiated on April 8, 2010; see 75 FR 17947) recommended surveys within the historical range and within suitable habitat on Lanai, with no change in status. Despite repeated surveys of historical and suitable habitat by botanists since 2006, *P. glabra* var. *lanaiensis* has not been found (Plant Extinction Prevention Program (PEPP) 2012; Oppenheimer 2019, in litt.). In 2012, PEPP reported that *P. glabra* var. *lanaiensis* was likely extinct. The 5-year status review completed in 2019 (initiated on February 12, 2016; see 81 FR 7571) recommended delisting due to extinction.

Historically, *P. glabra* var. *lanaiensis* was known from only two collections from Lanai, one from the ''mountains of Lanai,'' and the other from Kaiholena Gulch, where it was last collected in 1914 (USFWS 1991, 1995, 2003; Wagner 1999; Hawaii Biodiversity and Mapping Program 2010). A report of this species from the early 1980s in a gulch feeding into the back of Maunalei Valley probably was erroneous and likely *P. glabra* var. *glabra* (USFWS 1995, 2003; Wagner 1999, p. 269). Very little is known of the preferred habitat or associated species of *P. glabra* var. *lanaiensis* on the island of Lanai. It has been observed in lowland mesic to wet forest in gulch bottoms and sides, often in quite steep areas, in the same habitat as the endangered *Cyanea macrostegia* ssp. *gibsonii* (listed as *C. gibsonii*) (USFWS 1995).

Phyllostegia glabra var. *lanaiensis* was a short-lived perennial herb. Flowering cycles, pollination vectors, seed dispersal agents, longevity, specific environmental requirements, and limiting factors of *P. glabra* var. *lanaiensis* remain unknown (USFWS 1995, 2003). *P. glabra* var. *lanaiensis* was described as a variety of *P. glabra* from specimens collected from Lanai by Ballieu, Munro, and Mann and Brigham. It differed from *P. glabra* var. *glabra* in its longer calyx (the collection of modified leaves that enclose the petals and other parts of a flower) (0.3 inches or 10–11 millimeters) and narrowly lanceolate leaves (Wagner *et al.* 1990, p. 816). No taxonomic changes have been made since the variety was described in 1934.

II. Information on Detectability, Survey Effort, and Time Since Last Detection

Species Detectability

Phyllostegia glabra var. *lanaiensis* was a short-lived perennial herb. This taxon differed from the other variety by its longer calyces and narrowly lanceolate leaves, suggesting that flowers should be present in order to confirm identification. Most congeners tend to flower year-round, with peak flowering from April through June, indicating that it would be easier to detect and confirm the species during this time period.

Survey Effort

The PEPP surveys and monitors rare plant species on Lanai; botanical surveys are conducted on a rotational basis, based on the needs for collections and monitoring. Opportunistic surveying is also conducted when botanists are within the known range and suitable habitat when other work brings them to that area. No observations of *P. glabra* var. *lanaiensis* have been reported since 1914. By 2012, PEPP determined that this variety was likely extirpated (PEPP 2012), with very little chance of rediscovery due to the restricted known range, thorough search effort, and extent of habitat degradation. However, botanists were still searching for this taxon on any surveys in or near its last known location and other suitable habitat, as recently as January 2019 (Oppenheimer 2019, in litt.).

Time Since Last Detection

All *P. glabra* identified since 1914 have been determined to be *P. glabra* var. *glabra,* and, therefore, *P. glabra* var. *lanaiensis* has not been detected since 1914.

III. Analysis

Threats to the species included habitat degradation and herbivory by feral ungulates, the establishment of ecosystem-altering invasive plant species, and the consequences of small population sizes. Despite repeated surveys of historical and suitable habitat by botanists from 2006 through 2019, *P. glabra* var. *lanaiensis* has not been found since 1914 (PEPP 2012; Oppenheimer 2019, in litt.). In 2012, PEPP reported that *P. glabra* var. *lanaiensis* was likely extinct. In 2019, the species was included on the list of possibly extinct Hawaiian vascular plant taxa (Wood *et al.* 2019).

IV. Conclusion

At the time of listing in 1991, *P. glabra* var. *lanaiensis* had not been detected in over 75 years. Since its last detection in 1914, botanical surveys have not detected the species. Available information indicates that the species was not able to persist in the face of environmental stressors, and we conclude that the best available scientific and commercial information indicates that the species is extinct.

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 **ADDRESSES**. To better help us revise the rule, your comments should be as specific as possible. For example, you should tell us the names 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 regulations adopted pursuant to section 4(a) of the Act. We published a notice outlining our reasons for this determination in the **Federal Register** on October 25, 1983 (48 FR 49244). This position was upheld by the U.S. Court of Appeals for the Ninth Circuit (*Douglas County* v. *Babbitt*, 48 F.3d 1495 (9th Cir. 1995), cert. denied 516 U.S. 1042 (1996)).

Government-To-Government Relationship With Tribes

In accordance with the President's memorandum of April 29, 1994

(Government-to-Government Relations with Native American Tribal Governments; 59 FR 22951), Executive Order 13175 (Consultation and Coordination with Indian Tribal Governments), and the Department of the Interior's manual at 512 DM 2, we readily acknowledge our responsibility to communicate meaningfully with recognized Federal Tribes on a government-to-government basis. In accordance with Secretarial Order 3206 of June 5, 1997 (American Indian Tribal Rights, Federal-Tribal Trust Responsibilities, and the Endangered Species Act), we readily acknowledge our responsibilities to work directly with Tribes in developing programs for healthy ecosystems, to acknowledge that Tribal lands are not subject to the same controls as Federal public lands, to remain sensitive to Indian culture, and to make information available to Tribes. The Seminole Tribe of Florida and the Miccousukee Tribe has expressed interest in the Bachman's warbler. We have reached out to these tribes by providing an advance notification prior to the publication of the proposed rule. We will continue to work with these and any other Tribal entities that expressed interest in these species during the development of a final rule to delist these species.

References Cited

Lists of the references cited in in this document are available on the internet at *http://www.regulations.gov* in the dockets provided above under *Public Comments* and upon request from the appropriate person, as specified under **FOR FURTHER INFORMATION CONTACT**.

Authors

The primary authors of this document are the staff members of the Branch of Delisting and Foreign Species, Ecological Services Program, as well as the staff of the Ecological Services Field Offices as specified under **FOR FURTHER INFORMATION CONTACT**.

List of Subjects in 50 CFR Part 17

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

Proposed Regulation Promulgation

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

PART 17—ENDANGERED AND THREATENED WILDLIFE AND PLANTS

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

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

§ 17.11 [Amended]

■ 2. Amend § 17.11(h), the List of Endangered and Threatened Wildlife:

■ a. Under MAMMALS, by removing the entry for ''Bat, little Mariana fruit'';

■ b. Under BIRDS, by removing the entries for ''Akepa, Maui'', ''Akialoa, Kauai'', ''Creeper, Molokai'', ''Nukupuu, Kauai'', ''Nukupuu'', Maui'', '' 'O'o, Kauai (honeyeater)'', ''Po'ouli (honeycreeper)'', ''Thrush, large Kauai'', ''Warbler (wood), Bachman's'', ''White-eye, bridled'', and ''Woodpecker, ivory-billed'';

■ c. Under FISHES, by removing the entries for ''Gambusia, San Marcos'' and ''Madtom, Scioto''; and

■ d. Under CLAMS, by removing the entries for ''Acornshell, southern'' and ''Blossom, green''; both entries for ''Blossom, tubercled'', ''Blossom, turgid'', and ''Blossom, yellow''; and the entries for ''Combshell, upland'', ''Pigtoe, flat'', and ''Stirrupshell''.

§ 17.12 [Amended]

■ 3. Amend § 17.12(h), the List of Endangered and Threatened Plants, under FLOWERING PLANTS, by removing the entry for ''*Phyllostegia glabra* var. *lanaiensis*''.

§ 17.85 [Amended]

■ 4. Amend § 17.85(a) by:

■ a. In the heading, removing the word ''*Seventeen*'' and adding in its place the word ''*Fourteen*'';

■ b. In the table, removing the entries for ''tubercled blossom (pearly mussel)'', ''turgid blossom (pearly mussel)'', and ''yellow blossom (pearly mussel)'';

■ c. In paragraph (a)(1)(i), by removing the number ''17'' and adding in its place the number ''14'';

■ d. In paragraph (a)(1)(ii), by removing the number ''17'' and adding in its place the number ''14''; and

■ e. In paragraph (a)(2)(iii), by removing the number ''17'' and adding in its place the number ''14''.

§ 17.95 [Amended]

■ 4. Amend § 17.95 by:

■ a. In paragraph (e), removing the entry for ''San Marcos Gambusia (*Gambusia georgei*)''; and

■ b. In paragraph (f), the entry for, ''Eleven Mobile River Basin Mussel Species: Southern Acornshell (*Epioblasma othcaloogensis*), Ovate Clubshell (*Pleurobema perovatum*), Southern Clubshell (*Pleurobema decisum*), Upland Combshell (*Epioblasma metastriata*), Triangular Kidneyshell (*Ptychobranchus greenii*), Alabama Moccasinshell (*Medionidus acutissimus*), Coosa Moccasinshell (*Medionidus parvulus*), Orange-nacre Mucket (*Lampsilis perovalis*), Dark Pigtoe (*Pleurobema furvum*), Southern Pigtoe (*Pleurobema georgianum*), and Fine-lined Pocketbook (*Lampsilis altilis*)'', revising the entry's heading, the first sentence of the introductory text of paragraph (f)(1), the introductory text of paragraph (f)(2)(i), the table at paragraph (f)(2)(ii), the introductory text of paragraph (f)(2)(xiv), paragraph (f)(2)(xiv)(B), the introductory text of paragraph (f)(2)(xv), paragraph (f)(2)(xv)(B), the introductory text of paragraph (f)(2)(xx), paragraph (f)(2)(xx)(B), the introductory text of paragraph (f)(2)(xxi), paragraph (f)(2)(xxi)(B), the introductory text of paragraph (f)(2)(xxiii), paragraph (f)(2)(xxiii)(B), the introductory text of paragraph (f)(2)(xxvi), paragraph (f)(2)(xxvi)(B), the introductory text of paragraph (f)(2)(xxvii), paragraph (f)(2)(xxvii)(B), the introductory text of paragraph (f)(2)(xxviii), and paragraph (f)(2)(xxviii)(B) to read as follows:

§ 17.95 Critical habitat—fish and wildlife.

* * * * *

(f) *Clams and Snails.*

* * * * *

Nine Mobile River Basin Mussel Species: Ovate clubshell (*Pleurobema perovatum*), southern clubshell (*Pleurobema decisum*), triangular kidneyshell (*Ptychobranchus greenii*), Alabama moccasinshell (*Medionidus acutissimus*), Coosa moccasinshell (*Medionidus parvulus*), orange-nacre mucket (*Lampsilis perovalis*), dark pigtoe (*Pleurobema furvum*), southern pigtoe (*Pleurobema georgianum*), and fine-lined pocketbook (*Lampsilis altilis*)

(1) The primary constituent elements essential for the conservation of the ovate clubshell (*Pleurobema perovatum*), southern clubshell (*Pleurobema decisum*), triangular kidneyshell (*Ptychobranchus greenii*), Alabama moccasinshell (*Medionidus acutissimus*), Coosa moccasinshell (*Medionidus parvulus*), orange-nacre mucket (*Lampsilis perovalis*), dark pigtoe (*Pleurobema furvum*), southern pigtoe (*Pleurobema georgianum*), and fine-lined pocketbook (*Lampsilis altilis*) are those habitat components that support feeding, sheltering, reproduction, and physical features for maintaining the natural processes that support these habitat components.
* * *

(2) * * *

(i) *Index map.* The index map showing critical habitat units in the States of Mississippi, Alabama, Georgia, and Tennessee for the nine Mobile River Basin mussel species follows:

(ii) * * *

Species	Critical habitat units	States
Ovate clubshell (*Pleurobema perovatum*)	Units 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 17, 18, 19, 21, 24, 25, 26.	AL, GA, MS, TN.
Southern clubshell (*Pleurobema decisum*)	Units 1, 2, 3, 4, 5, 6, 7, 8, 9, 13, 14, 15, 17, 18, 19, 21, 24, 25, 26.	AL, GA, MS, TN.
Triangular kidneyshell (*Ptychobranchus greenii*)	Units 10, 11, 12, 13, 18, 19, 20, 21, 22, 23, 24, 25, 26	AL, GA, TN.
Alabama moccasinshell (*Medionidus acutissimus*)	Units 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 15, 25, 26	AL, GA, MS, TN.
Coosa moccasinshell (*Medionidus parvulus*)	Units 18, 19, 20, 21, 22, 23, 24, 25, 26	AL, GA, TN.
Orange-nacre mucket (*Lampsilis perovalis*)	Units 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15	AL, MS.
Dark pigtoe (*Pleurobema furvum*)	Units 10, 11, 12	AL.
Southern pigtoe (*Pleurobema georgianum*)	Units 18, 19, 20, 21, 22, 23, 24, 25, 26	AL, GA, TN.
Fine-lined pocketbook (*Lampsilis altilis*)	Units 13, 16, 17, 18, 19, 20, 21, 22, 23, 24, 25, 26	AL, GA, TN.

* * * * *

(xiv) Unit 12. Locust Fork and Little Warrior Rivers, Jefferson, Blount Counties, Alabama. This is a critical habitat unit for the ovate clubshell, triangular kidneyshell, Alabama moccasinshell, orange-nacre mucket, and dark pigtoe.

* * * * *

(B) Map of Unit 12 follows:

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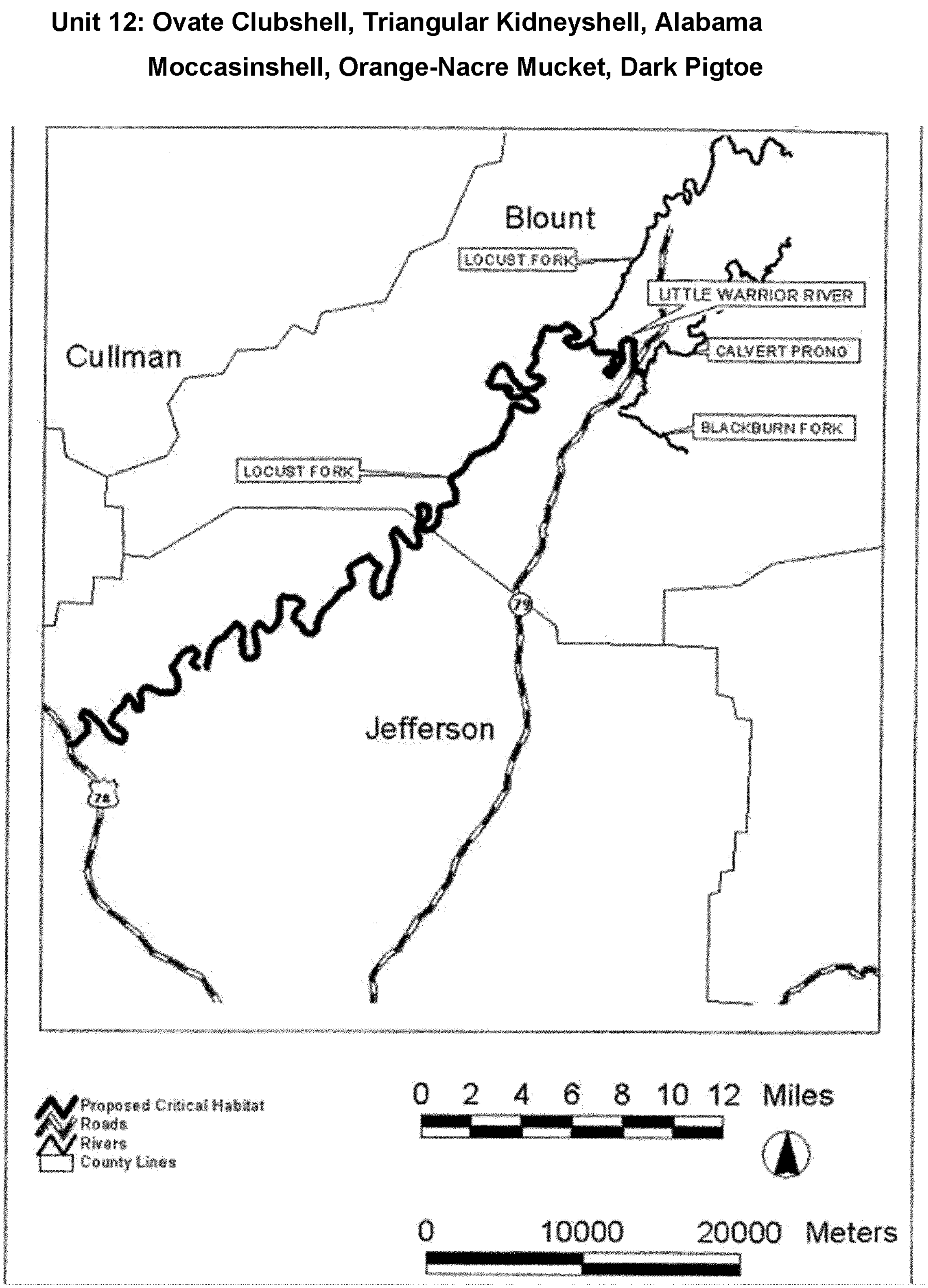
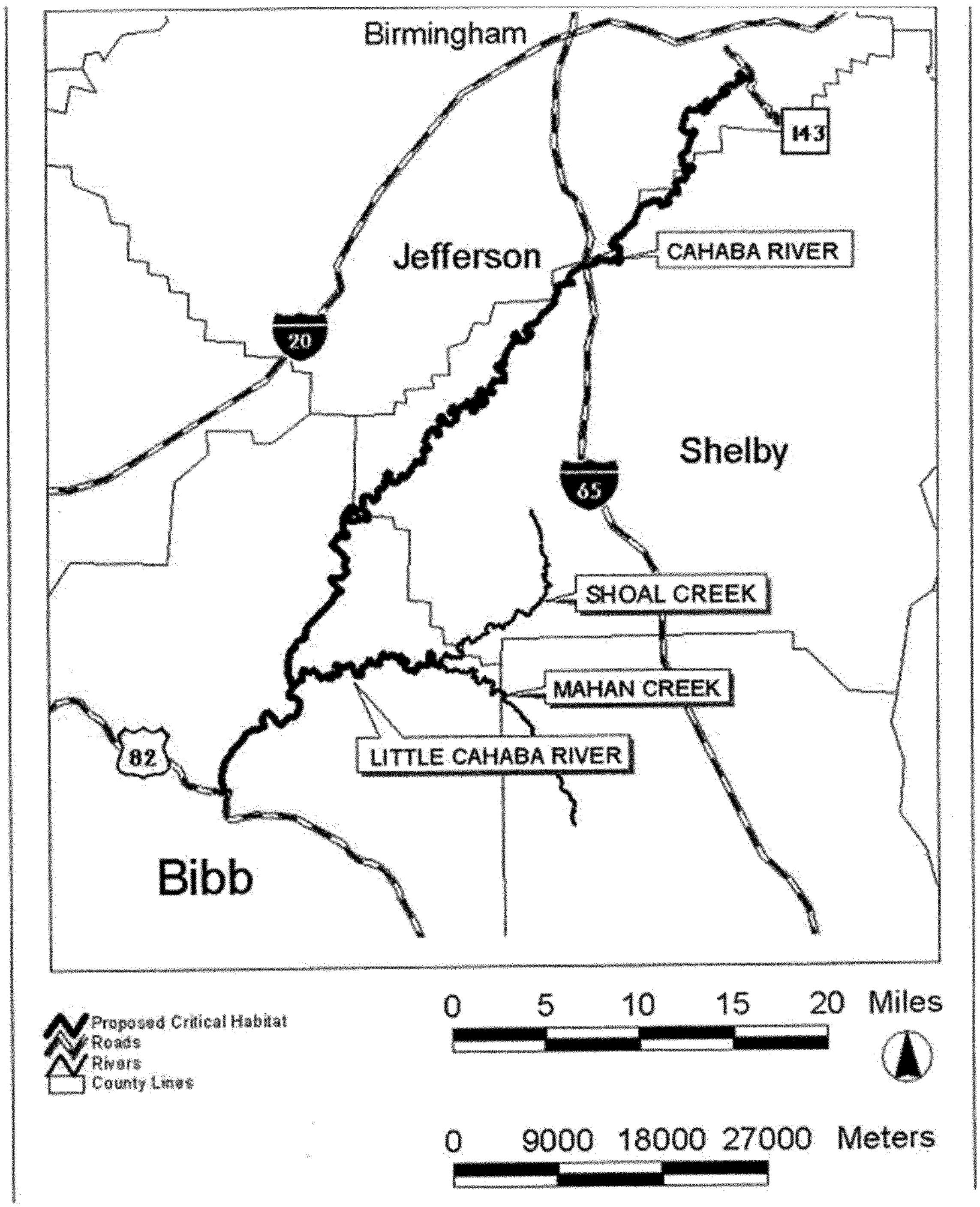


(xv) Unit 13. Cahaba River and Little Cahaba River, Jefferson, Shelby, Bibb Counties, Alabama. This is a critical habitat unit for the ovate clubshell, southern clubshell, triangular kidneyshell, Alabama moccasinshell, orange-nacre mucket, and fine-lined pocketbook.

* * * * *

(B) Map of Unit 13 follows:

Unit 13: Ovate Clubshell, Southern Clubshell, Triangular Kidneyshell, Alabama Moccasinshell, Orange-Nacre Mucket, Fine-Lined Pocketbook



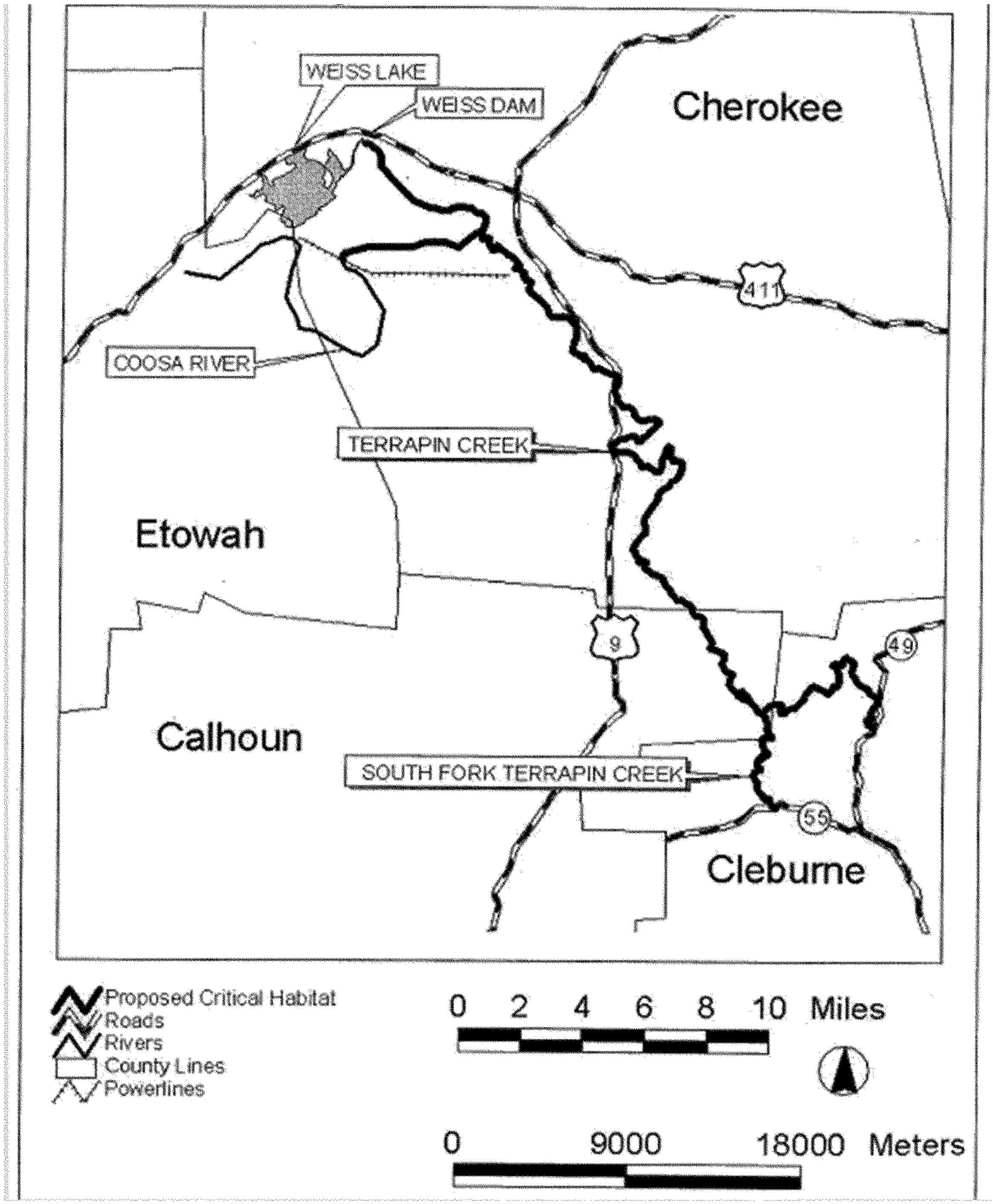
* * * * *

(xx) Unit 18. Coosa River (Old River Channel) and Terrapin Creek, Cherokee, Calhoun, Cleburne Counties, Alabama. This is a critical habitat unit for the ovate clubshell, southern clubshell, triangular kidneyshell, Coosa moccasinshell, southern pigtoe, and fine-lined pocketbook.

* * * * *

(B) Map of Unit 18 follows:

Unit 18: Ovate Clubshell, Southern Clubshell, Triangular Kidneyshell, Coosa Moccasinshell, Southern Pigtoe, Fine-Lined Pocketbook

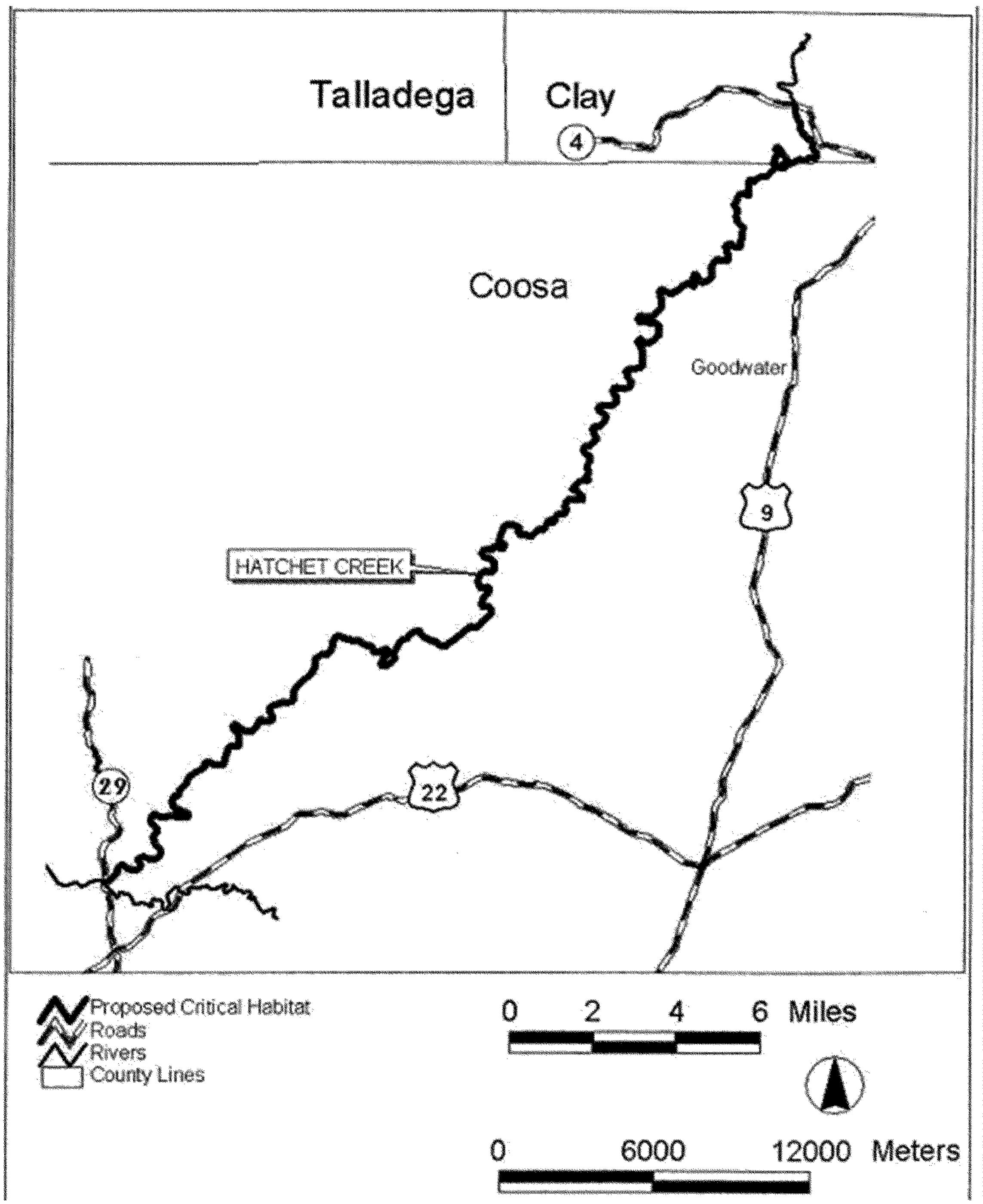


(xxi) Unit 19. Hatchet Creek, Coosa, Clay Counties, Alabama. This is a critical habitat unit for the ovate clubshell, southern clubshell, triangular kidneyshell, Coosa moccasinshell, southern pigtoe, and fine-lined pocketbook.

* * * * *

(B) Map of Unit 19 follows:

Unit 19: Ovate Clubshell, Southern Clubshell, Triangular Kidneyshell, Coosa Moccasinshell, Southern Pigtoe, Fine-Lined Pocketbook



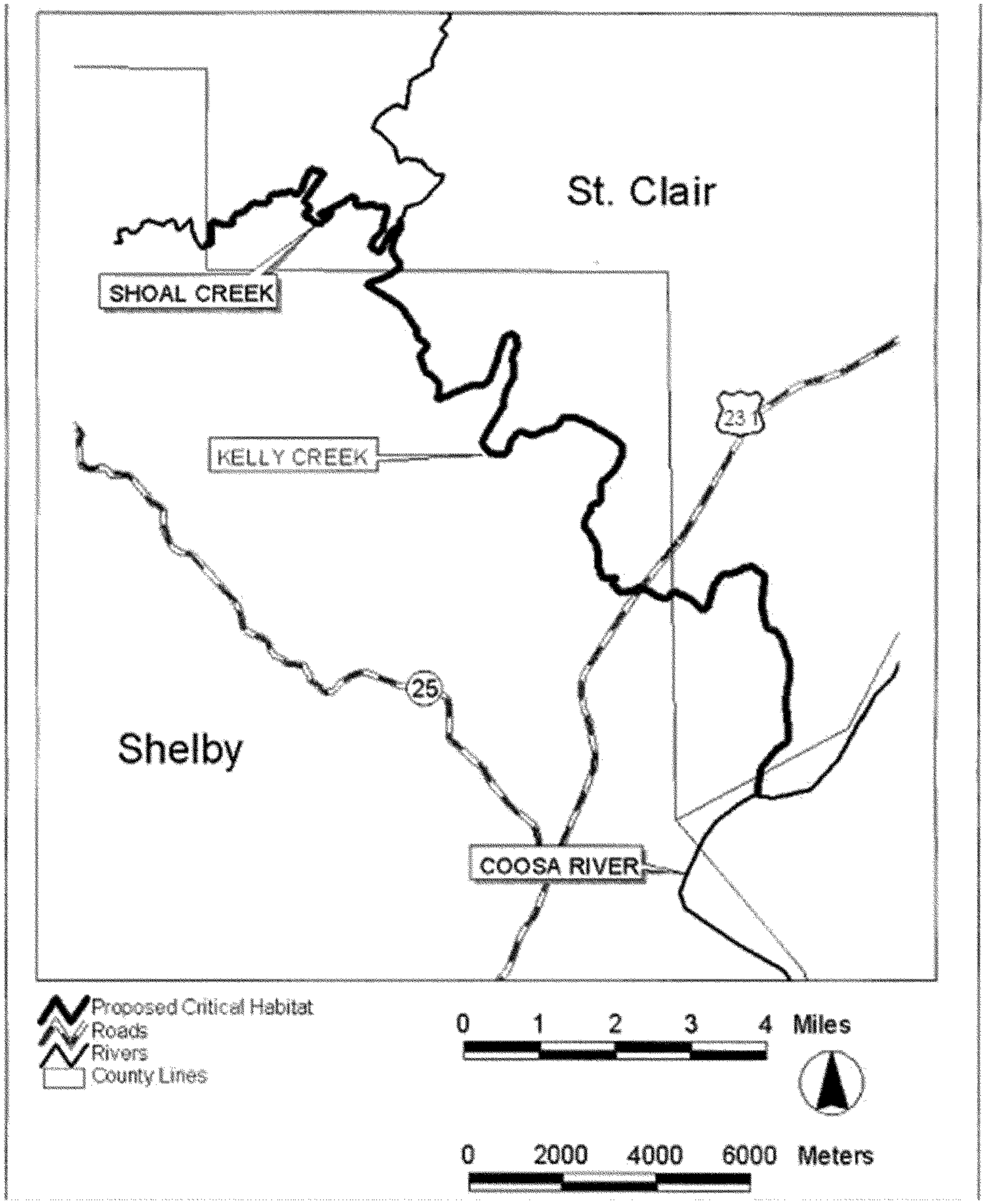
* * * * *

(xxiii) Unit 21. Kelly Creek and Shoal Creek, Shelby, St. Clair Counties, Alabama. This is a critical habitat unit for the ovate clubshell, southern clubshell, triangular kidneyshell, Coosa moccasinshell, southern pigtoe, and fine-lined pocketbook.

* * * * *

(B) Map of Unit 21 follows:

Unit 21: Ovate Clubshell, Southern Clubshell, Triangular Kidneyshell, Coosa Moccasinshell, Southern Pigtoe, Fine-Lined Pocketbook



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(xxvi) Unit 24. Big Canoe Creek, St. Clair County, Alabama. This is a critical habitat unit for the ovate clubshell, southern clubshell, triangular kidneyshell, Coosa moccasinshell, southern pigtoe, and fine-lined pocketbook.

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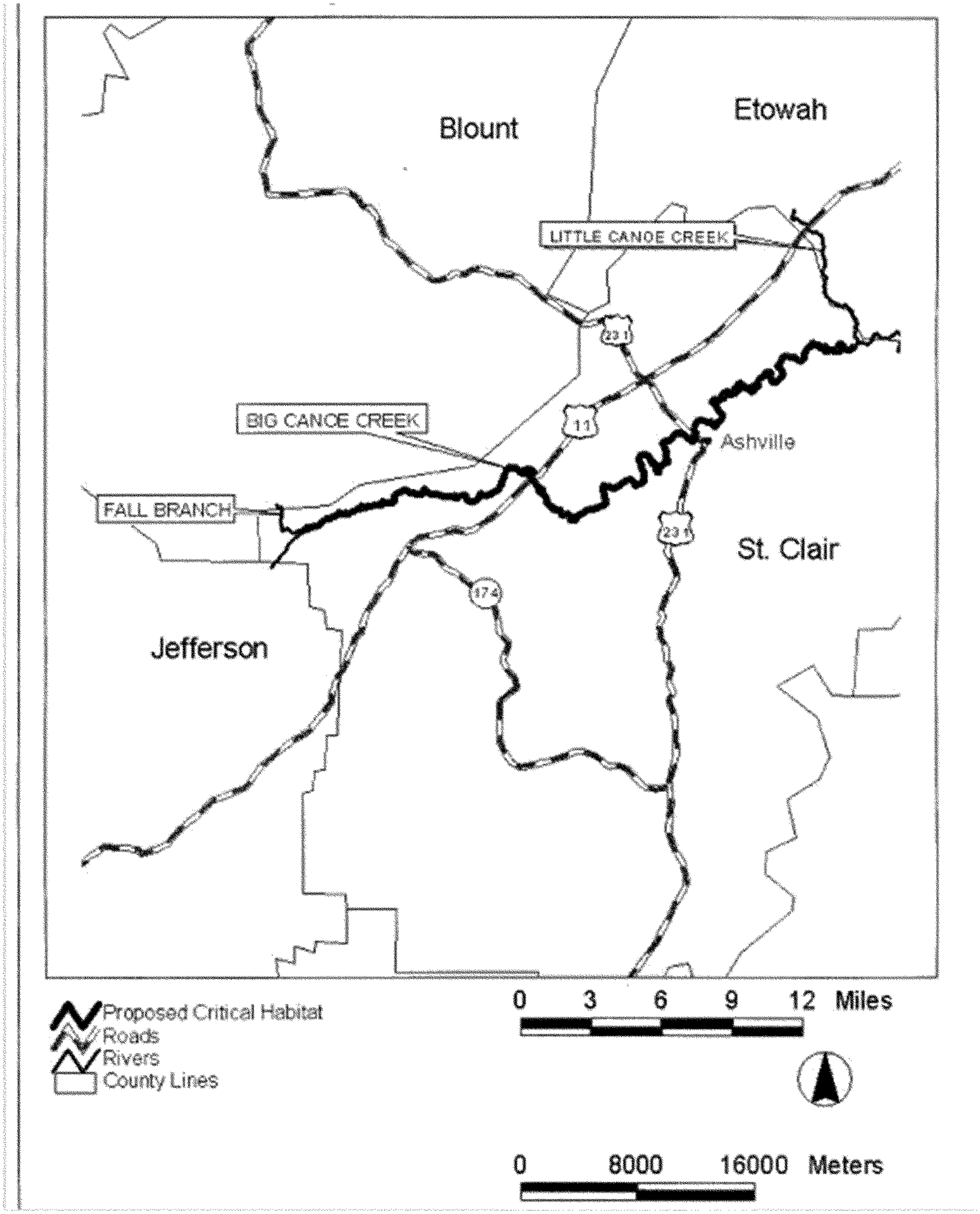
(B) Map of Unit 24 follows:

Unit 24: Ovate Clubshell, Southern Clubshell, Triangular Kidneyshell, Coosa Moccasinshell, Southern Pigtoe, Fine-Lined Pocketbook



(xxvii) Unit 25. Oostanaula, Coosawattee, and Conasauga Rivers, and Holly Creek, Floyd, Gordon, Whitfield, Murray Counties, Georgia; Bradley, Polk Counties, Tennessee. This is a critical habitat unit for the ovate clubshell, southern clubshell, triangular kidneyshell, Alabama moccasinshell, Coosa moccasinshell, southern pigtoe, and fine-lined pocketbook.

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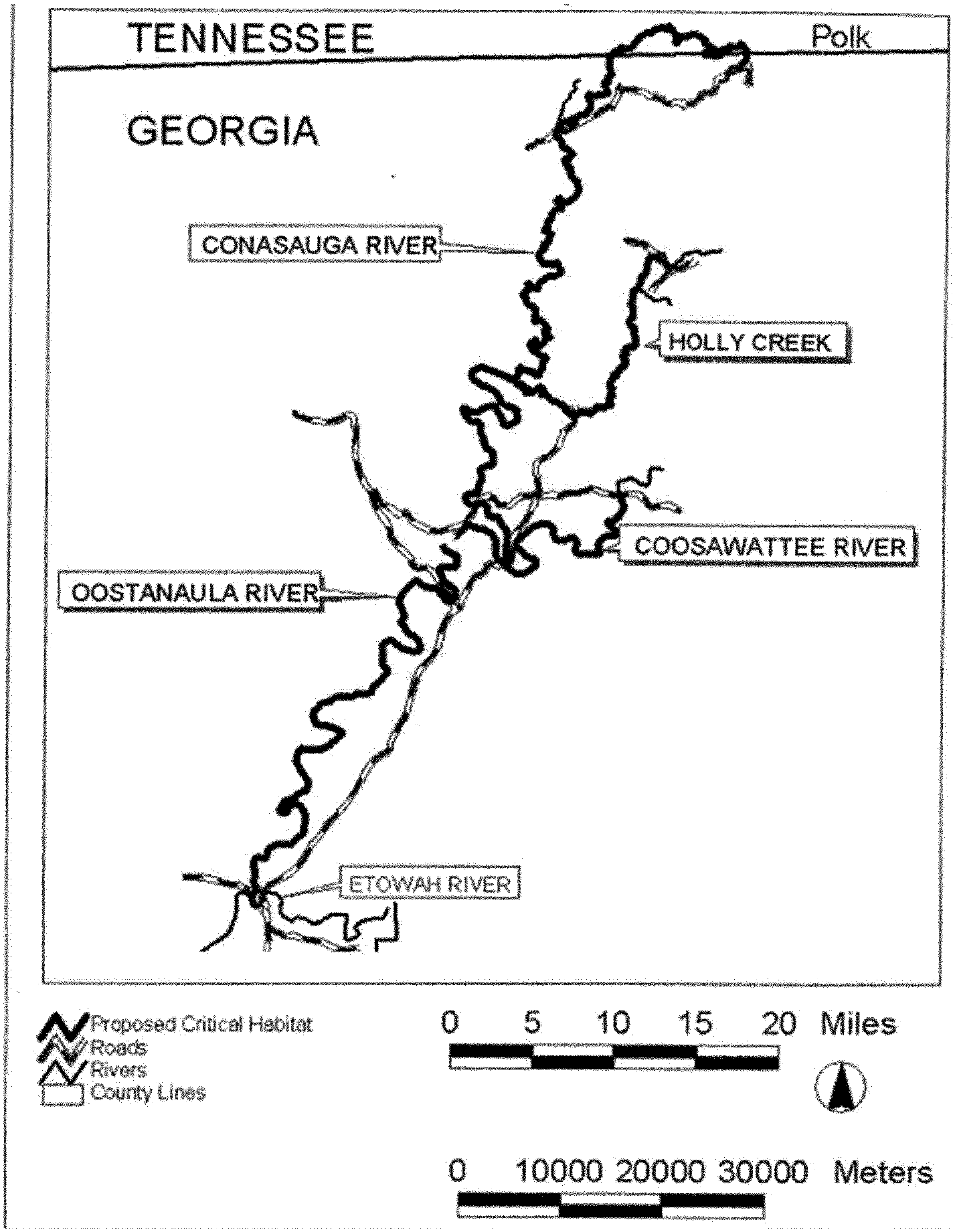
(B) Map of Unit 25 follows:

Unit 25: Ovate Clubshell, Southern Clubshell, Triangular Kidneyshell, Alabama Moccasinshell, Coosa Moccasinshell, Southern Pigtoe, Fine-Lined Pocketbook



(xxviii) Unit 26. Lower Coosa River, Elmore County, Alabama. This is a critical habitat unit for the ovate clubshell, southern clubshell, triangular kidneyshell, Alabama moccasinshell, Coosa moccasinshell, southern pigtoe, and fine-lined pocketbook.

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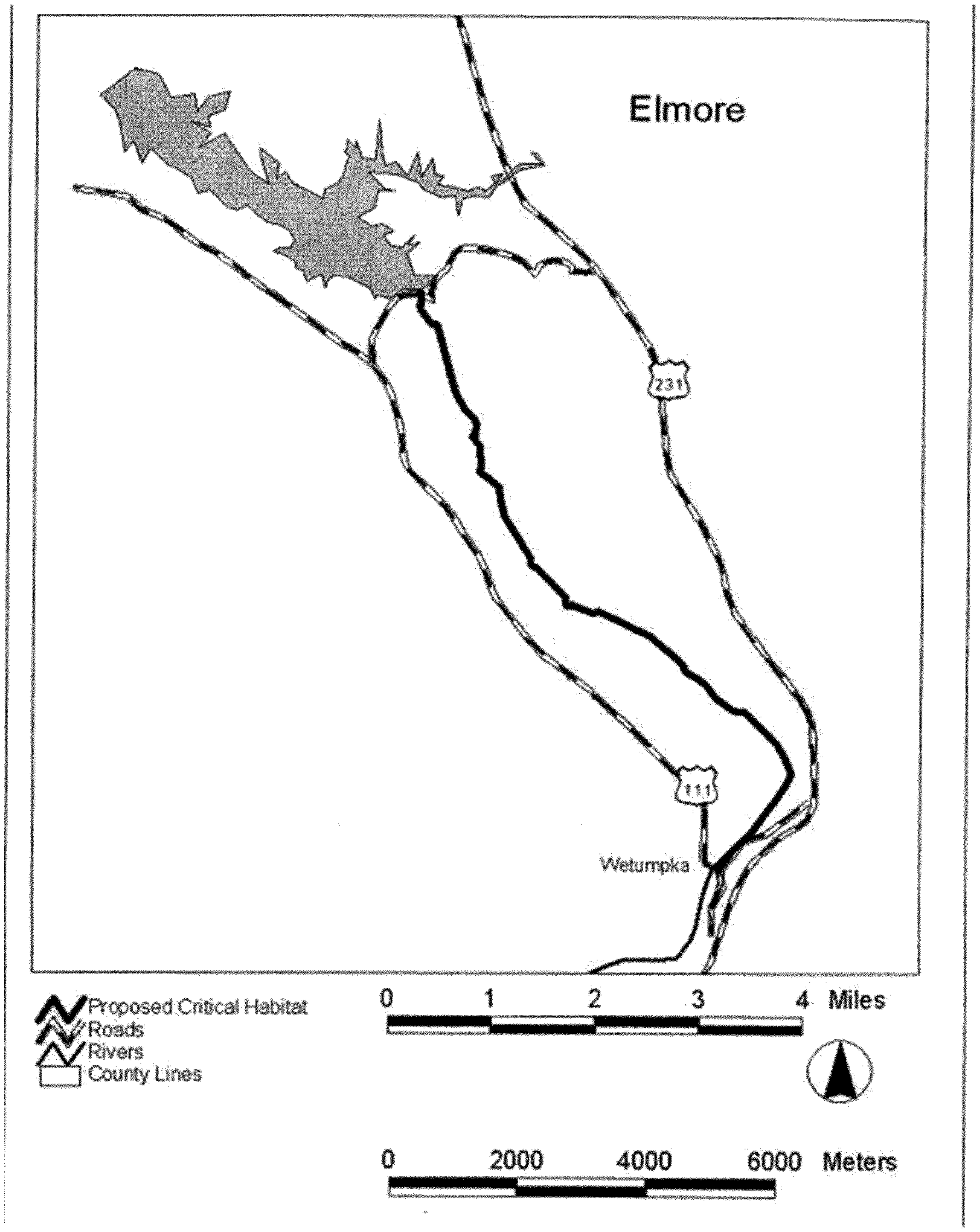
(B) Map of Unit 26 follows:

Unit 26: Ovate Clubshell, Southern Clubshell, Triangular Kidneyshell, Alabama Moccasinshell, Coosa Moccasinshell, Southern Pigtoe, Fine-Lined Pocketbook



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Martha Williams,

Principal Deputy Director, Exercising the Delegated Authority of the Director, U.S. Fish and Wildlife Service.

[FR Doc. 2021–21219 Filed 9–29–21; 8:45 am]

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