5-YEAR STATUS REVIEW

Species Reviewed: ʻōpeʻapeʻa or Hawaiian hoary bat
(***Lasiurus cinereus semotus***)
Current Classification: Endangered

Federal Register Notice announcing initiation of this review:
[USFWS] U.S. Fish and Wildlife Service. 2018. Endangered and threatened wildlife and plants; initiation of 5-year status reviews of 18 species in Hawaii, Oregon, Washington, Montana, and Idaho. Federal Register 83(14):3014–3015.

Lead Region/Field Office:
Interior Region 12/Pacific Islands Fish and Wildlife Office (PIFWO), Honolulu, Hawaiʻi

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Methodology used to complete this 5-year status review:
This review was conducted by staff of the Pacific Islands Fish and Wildlife Office (PIFWO) of the U.S. Fish and Wildlife Service (Service/USFWS). The review is based on the recovery plan for the Hawaiian hoary bat (*Lasiurus cinereus semotus*) (ʻōpeʻapeʻa) (USFWS 1998), the previous 5-year Review Summary and Evaluation for the Hawaiian hoary bat (USFWS 2011), and current available information on Hawaiian hoary bats. The document was reviewed by the Animal Recovery Coordinator, the Conservation and Restoration Team Manager and the Field Supervisor for PIFWO approval; it was then reviewed by staff in the Regional Office Ecological Services program prior to being signed by the Assistant Regional Director - Ecological Services.

Background:
For information regarding the species listing history and other facts, please refer to the U.S. Fish and Wildlife Service's Environmental Conservation Online System (ECOS) database for threatened and endangered species (*http://ecos.fws.gov/tess_public*).

Review Analysis:
Please refer to the previous 5-year status review for *Lasiurus cinereus semotus*, published on September 30, 2011 (USFWS 2011; available at https://ecos.fws.gov/docs/five_year_review/doc3865.pdf), for a complete five-factor analysis and discussion of the species' status (including biology and habitat), threats, and management recommendations.

The ʻōpeʻapeʻa or Hawaiian hoary bat is an endangered endemic mammal found in the Hawaiian archipelago. Listed as a subspecies of the hoary bat (*Lasiurus cinereus*), the ʻōpeʻapeʻa is distributed across all of the major islands of the Hawaiian archipelago, including Kauaʻi, Oʻahu, Lānaʻi, Maui, Molokaʻi, and Hawaiʻi. Most recently, ʻōpeʻapeʻa have been observed visiting the island of Kahoʻolawe (KIRC 2017). ʻŌpeʻapeʻa roost alone or with dependent young in native and nonnative trees, typically more than 4.6 meters (15 feet) tall (Amlin and Siddiqi 2015). The pupping season extends from June to September; the Service and Hawaiʻi Division of Forestry and Wildlife (DOFAW) currently recommend avoiding tree-trimming from June 1 to September 15 while pups are unable to fly (Amlin and Siddiqi 2015). ʻŌpeʻapeʻa primarily feed on nocturnal moths and beetles (Jacobs 1999), which they hunt in flight across a wide array of habitat types and plant communities from sea level to at least 3,600 meters (11,800 feet) above sea level (Todd 2012, Gorresen *et al*. 2013, Bonaccorso *et al*. 2015, Gorresen *et al*. 2015, Bonaccorso *et al*. 2016, Todd *et al*. 2016, Johnston *et al.* 2019). No historical or current population estimates exist for this subspecies, although recent studies and ongoing research have shown the bats to be distributed across all of the Hawaiian archipelago. The ʻōpeʻapeʻa was listed as endangered in 1970 (35 FR 16046), based on apparent habitat loss and limited knowledge of its distribution and life history requirements. At the time of listing, no population estimate was given. New genetic information and the current status and threats for ʻōpeʻapeʻa are summarized below.

New status information:

<u>Genetics, Colonization and Morphology</u>
Until 2015, published genetic studies on *Lasiurus cinereus* were limited to an analysis of species-level variation within the genus *Lasiurus* by Baker *et al*. (1988) and a separate analysis by Morales and Bickham (1995) that supported the taxonomic distinction of North American, South American, and Hawaiʻi populations at the subspecies level. In the past few years several publications have analyzed the genetic relationships of the ʻōpeʻapeʻa within the larger *Lasiurus* complex and within the State of Hawaiʻi (Russell *et al*. 2015, Baird *et al*. 2015, Baird *et al*. 2017, Pinzari *et al*. 2020). These studies have identified geographic variation in genetics across the range of ʻōpeʻapeʻa within Hawaiʻi.

Based on the mitochondrial DNA (mtDNA) and nuclear DNA sequences of the samples analyzed, Russell *et al*. (2015) identified two geographically overlapping clades: one was found across the Hawaiian archipelago, but not on the North American continent, and the other was found on Maui, Oʻahu and the North American continent. In a different study, Baird *et al*. (2015) analyzed mtDNA sequences of 9 ʻōpeʻapeʻa from the islands of Maui and Hawaiʻi, 13 different hoary bat representatives from North America, 1 representative from South America, and additional outgroup species. Individuals from the Hawaiian islands formed two distinct clades: one consisting of only ʻōpeʻapeʻa (*L. c. semotus*) from the islands of Maui and Hawaiʻi, and one consisting of other individuals from Maui and all of the sampled North American specimens (*L. c. cinereus*). Based on this study, Baird *et al*. (2015) recommended that the three subspecies of *L. cinereus* (*L. c. cinereus*, *L. c. semotus*, and the South American *L. c. villosissimus*) each be raised to species status.

Baird *et al*. (2017) conducted further analyses using mtDNA and nuclear DNA from CMA1 (chymase gene) alleles. *Lasiurus c. cinereus* and *L. c. semotus* have distinct CMA1 lineages. Results show the presence of two clades that are primarily associated with the *L.c. semotus* mtDNA haplotypes unique to Hawaiʻi. Also, more genetic diversity in the CMA1 gene exists within Hawaiʻi than occurs in the North American *L. c. cinereus* grouping. Baird *et al*. (2017) identified a few individuals that possessed mtDNA haplotypes of the clade that appears to be limited to the Hawaiian islands and possessed nuclear alleles from the Maui/North America clade, and vice versa. These mismatched individuals were considered to have a hybrid ancestry, suggesting some hybridization between the Hawaii-limited group and the North American group (15 percent; 4 out of 27 individuals) (Baird *et al*. 2017).

Baird *et al*. (2015) concluded that the original colonization, represented by the presumed *Lasiurus cinereus semotus* clade, occurred between 400,000 and 1.8 million years ago, while the observation of two distinct North American *L. c. cinereus* haplotypes on Maui supported at least one and possibly two more recent events. In contrast, the results from Russell *et al*. (2015) suggested that Hawaiian *Lasiurus* populations resulted from at least two relatively more recent dispersal events from North American populations of *L. c. cinereus*, with the first colonization occurring no more than 10,000 years ago and the second perhaps 800 years ago. To address these marked inconsistencies between the results by Russell *et al*. (2015) and Baird *et al*. (2015), Baird *et al*. (2017) incorporated data from the Russell *et al.* (2015) study and examined four mtDNA and two nuclear DNA markers to further investigate the timing of colonization of the Hawaiian Islands by hoary bats. This analysis proposed that hoary bats colonized from North America around 1.35 million years ago to Kauaʻi, Oʻahu, or Maui (the islands existing at the time), giving rise to the *L. c. semotus* clade, additional arrivals of *L. c. cinereus* occurred more recently, and a notable population increase in the Hawaiʻi-limited clade occurred 20,000 years ago (Baird *et al*. 2017). Population increase may have been due to colonization of new islands and/or expansion after a bottleneck (Baird *et al*. 2017).

Results presented by Baird *et al*. (2015) and Russell *et al*. (2015) indicate that the geographic origin for *Lasiurus* in the Hawaiian islands is North America, confirming the previous suggestion by Morales and Bickham (1995) based on a specimen also sequenced by Baird *et al*. (2015). Bonaccorso and McGuire (2013) modeled energetics and water balance of simulated colonization flights for *L. c. cinereus* founders arriving in Hawai'i. They concluded that physical conditions (tradewind velocity and direction) and physiological conditions during fall migration (fat storage, energy consumption, and water balance) would allow for long-distance dispersal from the Pacific coast of North America (rather than from other parts of its range), and suggested that multiple colonization events may have been possible despite the energetic and physical constraints on dispersers.

Analysis of mtDNA and microsatellite DNA by Pinzari (2019) also found high heterozygosity within the bat populations in Hawaiʻi, indicating that the bats remain genetically diverse. She also identified haplotypes in addition to the 10 previously identified by Russell *et al*. (2015) and Baird *et al*. (2015, 2017). The existence of unique haplotypes on each island supports an island-specific population structure. Like Baird *et al*. (2017), this study also suggested some hybridization, or mixing, between groups.

The most recent study by the U.S. Geological Survey - Pacific Island Ecosystem Research Center (USGS-PIERC), based on a whole-genome analysis of single nucleotide polymorphisms (Pinzari *et al*. 2020; C. Pinzari and F. Bonaccorso, pers. comm. 2018), provides higher resolution of taxonomic divergence from the North American hoary bat, indicating that hoary bats in Hawaiʻi likely constitute a single species, sharing a common ancestor from a single colonization event. They concluded that an initial colonization of Maui occurred approximately 1 million years ago, after which populations dispersed to the islands of Hawaiʻi, Oʻahu, and Kauaʻi between 500,000 and 200,000 years ago and diverged genetically. This study found that genomic variation in ʻōpeʻapeʻa is primarily associated with divergence among island populations rather than being associated with mtDNAlineages or clades, and suggested that previous inferences of multiple colonization events were a result of analyzing a relatively small number of genetic markers, where lineages had been incompletely sorted with ancestral polymorphism being retained.

Jacobs (1996) reported morphological divergence in the ʻōpeʻapeʻa from the North American subspecies involving characteristics related to flight and feeding. According to Jacobs (1996), the ʻōpeʻapeʻa has a 45 percent reduction in body size with allometric responses in the size of its wings when compared to the continental North American subspecies, *L. c. cinereus.* The wing changes have resulted in a lower ratio of weight to wing area, and are expressed as long, narrow wings relative to the continental North American subspecies. This physical trait permits slower and more maneuverable flight near vegetation and enduring flight in open areas. This increased flexibility in flight behavior has allowed the ʻōpeʻapeʻa to expand its foraging habitat to include both open habitats similar to those typical of *L. c. cinereus*, and closed habitats not used by *L. c. cinereus*. Skeletal features related to feeding also diverge, with ʻōpeʻapeʻa having relative increases in the size of the mouth opening (gape), the size of the muscle that closes the jaw (masseter muscle) and the height of the coronoid process relating to the structure of the jawbone. These changes give the jaw more crushing power for more efficient processing of larger and hard-bodied prey. This has enabled the ʻōpeʻapeʻa, despite a marked reduction in body size, to include large, hard-bodied insects such as beetles that are not taken by *L. c. cinereus* in its diet.

Measurements of the skull length and forearm length of ōpeʻapeʻa collected throughout Hawaiʻi have shown no significant morphological variation between the two clades identified in Hawaiʻi (Pinzari 2019). Hybridization between the clades and the small sample size available may have contributed to the lack of variation observed (C. Pinzari, pers. comm. 2019). Sexual dimorphism (females larger than males) has been observed in ōpeʻapeʻa collected across all of the islands (Pinzari 2019).

<u>ʻŌpeʻapeʻa detection and its limitations</u>

Detectability refers to the capability of detecting a bat if it is present. Although detectability is not directly related to the recovery or status of the species, it is important to recognize the limits of tools for detecting these highly mobile, solitary bats when conducting population trend and occupancy studies using acoustic monitoring. Acoustic and video findings from a study by Gorresen *et al*. (2015) show that ʻōpeʻapeʻa can be acoustically cryptic (8 percent chance of detection on a given night if it was present during the study). In multiple instances, bats flew close to microphones but were not recorded; frequently, video evidence of foraging-like behavior was not accompanied by concurrent detection of recorded feeding calls (Gorresen *et al*. 2015). Thus, acoustic detection was highly inefficient at detecting bat presence. Most recently, Gorresen *et al.* (2018a, b) confirmed that video-derived observations provided higher and more accurate estimates of the prevalence of bat flight activity and feeding events than acoustic sampling methods, although each method has its own limitation. This study demonstrated the usefulness of monitoring flight with videography, in addition to feeding activity with acoustic detectors for occupancy modeling (Gorresen *et al.* 2018b).

Barclay *et al*. (1999) found that, on average, ʻōpeʻapeʻa give higher-frequency calls (26.2 to 29.8 kHz) compared to mainland hoary bats (20.1 kHz). The reported frequency range varies from 23 to 46 kHz and this may not encompass the complete range of echolocation frequency. The same study found the frequency range varied depending on the island and area where the detection occurred. More recently, Corcoran and Weller (2018) demonstrated that hoary bats (*L. cinereus*) in North America use a novel call type called "micro calls" that has three orders of magnitude less sound energy than other bat calls used during typical echolocation in open habitats. Acoustic modelling indicates the bats are not producing calls that exceed 70 to 75 dB at 0.1-meter (4-inch) distance, indicating bats sometimes fly without echolocation. A possible benefit of hoary bats shifting from normal to micro calls is that it would make bats far less conspicuous to predators and conspecifics. However, at this level, the call would have little or no known use for a bat flying in the open at high speeds. A micro-calling bat should have sufficient time to detect and avoid large obstacles such as tree branches at close range, but they would have difficulty avoiding smaller objects, mist nets, or rapidly moving wind turbine blades (Corcoran and Weller 2018).

<u>Distribution and Seasonal Behavior by Island</u>

Island of Kauaʻi

A comprehensive acoustical survey of military land in the western portion of the island, including Barking Sands, Mākaha Ridge, and Kokeʻe, demonstrated year-round use of all these areas by ʻōpeʻapeʻa, although different seasonal values indicate varying use throughout the year (Bonaccorso and Pinzari 2011). Bats appeared to be using low-elevation habitats (Barking Sands) primarily during the summer and fall, but then showed increased activity at higher elevations (Mākaha Ridge and Kokeʻe) during the winter months (Bonaccorso and Pinzari 2011). Bats were also detected in acoustical surveys at Hanapepe Armory and Kekaha Firing Range (Montoya-Aiona *et al.* 2020).

ʻŌpeʻapeʻa activity was also monitored across the Service's National Wildlife Refuge (NWR) complexes in Hawaiʻi from January to December 2017, with 22 stationary acoustic detectors (Wolfe 2018). Bat activity was detected almost every night at Hanalei NWR on Kauaʻi. At Hulēʻia NWR and Kīlauea Point NWR, bats were detected on the majority of nights throughout the year, indicating high occupancy at all three of these lowland sites year-round.

Island of O'ahu

The 'ōpe'ape'a was once thought to be extirpated from O'ahu (USFWS 1998). However, a 2013 capture of a lactating female with two dependent pups near Waimea Valley on the north shore of O'ahu was the first direct evidence of breeding on O'ahu (H. T. Harvey & Associates 2013). Additional detections of 'ōpe'ape'a have been made across O'ahu, including on military lands in both the Ko'olau and Wai'anae mountain ranges, as well as Waikīkī, Ford Island, the north shore of O'ahu, and the NWR complex that includes the James Campbell NWR, the Kalaeloa Unit of the Pearl Harbor NWR, and the O'ahu Forest NWR (Pinzari 2014, O'ahu Army Natural Resource Program 2016, Wolfe 2018, Montoya-Aiona *et al.* 2020).

Although little movement data has been published from the island, Gorresen *et al*. (2015) studied the landscape distribution of 'ōpe'ape'a in the northern Ko'olau Mountains of O'ahu from May 2013 to May 2014, integrating acoustic monitoring and thermal videography. Acoustic detections were consistently low from October through February and increased at most north shore sites, peaking in April through August (Gorresen *et al*. 2015). From July 10 to August 10, 2017, Gorresen *et al.* (2018a, b) deployed thermal video with acoustic detectors in the northern Ko'olau Mountains of O'ahu, within the footprint of the Kawailoa Wind project. Elevated levels of acoustic activity by 'ōpe'ape'a were found to be related primarily to beetle biomass in this study.

The preliminary findings from an island-wide study, conducted in 2018 with 83 randomly placed acoustical detectors across O'ahu, resulted in 5,135 'ōpe'ape'a detections between June 8, 2017 and June 29, 2018 (Starcevich *et al*. 2019). At least 1 detection was recorded at 61 percent of the 83 sites. The level of detections recorded at each site ranged from 0 to 1,703, suggesting site usage by bats is highly variable. The highest number of detections occurred during the lactation period. Detections occurred across the island though the highest concentrations of detections were in the northern Ko'olau and Wai'anae Mountain ranges (Starcevich *et al*. 2019).

The combined results of these relatively recent various O'ahu observations and studies constitute new knowledge that 'ōpe'ape'a are breeding on O'ahu and are widely distributed.

Island of Moloka'i

Since the previous 5-year status review in 2011, recent surveys led by the Kalaupapa National Historical Park have reported detections of 'ōpe'ape'a across the island and in all months of the year (Hosten and Poland 2018), indicating that a resident population likely exists on the island. Bats were also detected in acoustical surveys at Kaunakakai Armory (Montoya-Aiona *et al.* 2020). In addition, Wolfe (2018) surveyed bats at Kakahai'a NWR and detected their presence on 14 nights during the course of a year. The year-round detection of bats on the island suggests presence of a breeding population.

Island of Maui

On Maui, a comprehensive distribution study was conducted by Todd *et al*. (2016) on the upper leeward slopes of Haleakalā. Baseline occupancy and habitat-use acoustic surveys were conducted prior to the restoration of 3,200 hectares (8,000 acres) of habitat for bats in the Kahikinui Forest Reserve and adjoining Nakula Natural Area Reserve (KFR-NNAR) (State of Hawaiʻi 2015a, 2015b). ʻŌpeʻapeʻa vocalizations were recorded from July 2012 to November 2014 at 14 locations in the KFR-NNAR (Todd *et al*. 2016). The study area included remnants of recovering mesic montane forest with interspersed grasses from 1,250 to 1,850 meters (4,100 to 6,070 feet) and xeric subalpine shrubland plant communities from 1,860 to 2,800 meters (6,100 to 9,200 feet). Detections occurred on 65 percent of nights and in every month of the study, with monthly detection probability values highest from July to November 2012, and greater detections occurring in the remnant forests than in the shrubland for most months. Significantly higher detection probability for bat calls during 2012 and particularly in July and August of that year coincided with at least two environmental variables: low rainfall and presence of high ungulate density in the reserve. According to Todd *et al*. (2016), the reserve experienced very low average annual rainfall in 2012 followed by higher annual rainfall in 2013 and 2014.

Todd *et al*. (2016) also postulated that a high density of ungulates may have been positively linked to high detection numbers in the KFR-NNAR in July 2012. By the end of 2012, ungulates had been removed and exclusion fencing was in place at the NNAR. The presence of high ungulate densities have been shown in other studies to be associated with increased insectivorous bat presence and foraging activity (as reviewed by Downs and Sanderson 2010). In particular, dung-feeding beetles and flies that associate with cattle and other herding ungulates are important food items for a number of insectivorous bats (Shiel *et al*. 1991). Scarab beetles and flies have been identified from fecal pellets of ʻōpeʻapeʻa captured near cattle farms on the island of Hawaiʻi (Todd 2012). Thus, Todd *et al*. (2016) concluded that ʻōpeʻapeʻa, like other insectivorous bats, find sufficient resources in areas with ungulates, like cattle. Yet, Montoya-Aoina *et al.* (2020) did not detect an association between ʻōpeʻapeʻa abundance and ungulate use. Thus, it is possible that the reduction in bat activity in 2013 and 2014 was associated with the elimination of ungulates in KFR-NNAR. Alternatively, the reduction in activity could be a temporary phenomenon and bat presence and foraging activity may rise over time as forest recovery resulting from ungulate exclusion and the associated turnover in plant and insect communities occurs. Noted as a generalist aerial insectivore feeding principally on moths and a diverse array of beetles in Hawaiʻi (Whitaker and Tomich 1983; Jacobs 1999; Todd 2012), ʻōpeʻapeʻa are expected to benefit in the long term as the insect fauna increases due to forest productivity increases across the KFR-NNAR. In addition, weather patterns over the course of these years may also have accounted for this pattern, as the first year had a higher number of clear nights with lower rainfall, and the subsequent years had higher rainfall (Todd *et al*. 2016). Follow up surveys for ʻōpeʻapeʻa will be conducted at KFR-NNAR to monitor the effect of restoration activities on bat activity, which may enable a more definitive answer to this question.

Acoustic monitoring and capture of ʻōpeʻapeʻa in the Waihou mitigation area of east Maui, including the Puʻu Makua Restoration Area, indicates that bats are present and actively foraging year-round, feeding primarily on moths (Pinzari *et al*. 2019).

Results from an extensive radio-tracking project by Johnston *et al*. (2018) show that bats are active at low and high elevations summer through winter. No seasonal correlation with elevation was detected, suggesting that at least some bats may not "shift" to the high elevations during the late fall on Maui (Johnston *et al*. 2018), as has been seen on other islands (Menard 2001, Bonaccorso and Pinzari 2011, Todd 2012, Gorresen *et al*. 2013). The highest level of acoustic activity characterized by feeding buzzes in this study was detected in low-density urban and gulch areas (Johnston *et al.* 2019). Bats use varied habitats with different levels of human impact which is largely driven by ephemerality and abundance of insect prey (Johnston *et al*. 2019). Bat activity was detected acoustically almost nightly at Keālia Pond NWR in the coastal isthmus of Maui (Wolfe 2018). Bats were also detected in acoustical surveys at Pu'unēnē Training Facility and Ukumehame Firing Range (Montoya-Aiona *et al.* 2020). The cumulative results of these and previous studies conducted across Maui suggest bats have a wide distribution and forage across fragmented habitats.

Island of Lānaʻi
ʻŌpeʻapeʻa have been documented on Lānaʻi in studies conducted by Castle & Cooke (2008, as reported by Tetra Tech and Towill 2008). The occurrence of pupping on the island has not been established, but there is no biological reason to assume breeding on Lānaʻi would be implausible.

Island of Kahoʻolawe
Acoustic detectors placed by the Kahoʻolawe Island Reserve Commission (KIRC) first detected vocalizations of the ʻōpeʻapeʻa in June 2016 (KIRC 2017). Additional acoustic detections were noted in August, September and October, before dropping in December and January. These data suggest that ʻōpeʻapeʻa occur seasonally on the island and at least some appear to travel to Kahoʻolawe after dusk and then return to either Maui or Lānaʻi before dawn. Peak detections occurred around 10:00 PM. It is yet unknown if breeding occurs on the island (KIRC 2017).

Island of Hawaiʻi
Surveys for ʻōpeʻapeʻa have been extensive on the island of Hawaiʻi. Todd (2012) found ʻōpeʻapeʻa activity varied seasonally among elevations. ʻŌpeʻapeʻa are most active at elevations of less than 1,000 meters (less than 3,300 feet) from late spring through summer and early fall, which coincides with the reproductive period. Sites at middle elevations had the highest bat activity during the reproductive period and had the largest decrease in bat activity during the non-reproductive period. High-elevation sites generally had the least ʻōpeʻapeʻa activity during the reproductive period. In general, this indicates that activities related to reproduction and pup rearing tend to take place in the low- to mid-elevations and movement to higher elevations occurs after pups fledge. This is supported by ʻōpeʻapeʻa activity at low-elevation sites being higher during the

reproductive period than during the non-reproductive period. Notably, bat activity at high elevation sites remained constant throughout the year.

Similarly, Gorresen *et al*. (2013) concluded hoary bats concentrate in the coastal lowlands of the island of Hawaiʻi during the pupping season, May through October, and then move to interior highlands during the winter. This was based on acoustic recordings of ʻōpeʻapeʻa collected over a 5-year period (2007 to 2011) from 25 survey areas across the island. This data also supports the previous conclusion by Menard (2001). ʻŌpeʻapeʻa occupy and forage at elevations between 2,200 and 3,600 meters (7,200 and 11,800 feet) during November through March (F. Bonaccorso, personal observation, cited by Gorresen *et al*. 2013). Highest occupancy in the coastal lowlands peaked in mid-September across the 5-year average, which corresponded to the August to September fledging season of the young from that year (Gorresen *et al*. 2013). Although the ʻōpeʻapeʻa is a habitat generalist species (Tomich 1986) and occurs from sea level to the highest volcanic peaks on the island of Hawaiʻi, there was a significant association between occupancy and the prevalence of mature forest cover. Overall, the trend in occupancy, while strongly suggestive, but not conclusive, was that the population on the island was stable to slightly increasing based on breeding season records over the 5 years of surveys. This was based on a threshold for ecological significance at 25 percent change in occupancy over 25 years (Gorresen *et al.* 2013).

Acoustic surveys were also conducted at the coastal Kaloko-Honokōhau National Historical Park (Pinzari *et al*. 2014, Montoya-Aiona *et al.* 2019). Of the four sites surveyed, Kaloko Fishpond (wetland shoreline habitat) and ʻAimakapā Fishpond (wetland shoreline habitat) had substantially more ʻōpeʻapeʻa activity than did the xeric lava beds at the park's south boundary (lava and fountain grass [*Cenchrus setaceus*] habitat) and the Northern Māmalahoa Trail (lava and haole koa [*Leucaena leucocephala*] habitat; Pinzari *et al*. 2014). Wolfe's (2018) study on NWRs across the island of Hawaiʻi detected bat activity almost nightly at Hakalau Forest NWR, indicating bats occur year-round in this area.

Bats were also detected in acoustical surveys at Kealakekua Armory and Keaukaha Military Reservation (Montoya-Aiona *et al.* 2020).

Additional radio-tracking, roosting, breeding demographics, and diet studies are being conducted on the island of Hawaiʻi by the USGS on National Park land, and as part of compensatory mitigation of wind energy projects, to better understand ʻōpeʻapeʻa biology and habitat needs (USGS-PIERC 2016a, Montoya-Aiona *et al*. 2019).

Roosting habitat and behavior

The day-roost habitat requirement for ʻōpeʻapeʻa is tall (crown height greater than 4.6 meters [15 feet]), shady trees. Tree species used frequently include mature native ʻōhiʻa, but also include a wide variety of introduced species such as lychee (*Litchi chinensis*), various species of eucalyptus, mango (*Mangifera indica*), and numerous other tree species (Bonaccorso *et al*. 2015). Roost trees used by radio-tracked bats on Maui include blue gum eucalyptus (*Eucalyptus globulus*), African tulip tree (*Spathodea campanulata*), and Monterey cypress (*Cupressus macrocarpa*) (Johnston *et al*. 2018).

The roosting behavior of five solitary adult bats was observed during the summer of 2017 (Moura *et al*. 2018) using thermal imagery and surveillance video. They typically entered shallow torpor during the day while maintaining a mean body temperature slightly above ambient temperature. The slight difference between body temperature and ambient temperature can make detection using thermal methods difficult. Spikes in body temperature can be associated with arousal from sleep and activity such as urination or grooming (Moura *et al*. 2018).

Diet

ʻŌpeʻapeʻa consume a wide variety of insects (Whitaker and Tomich 1983, Jacobs 1999, Todd 2012). Todd (2012) identified seven orders of insects (Insecta) in the diet of ʻōpeʻapeʻa: moths (Lepidoptera), beetles (Coleoptera), termites (Blattodea), flies (Diptera), true bugs (Hemiptera), bees and wasps (Hymenoptera), and lacewings (Neuroptera). Moths and beetles were the most frequently consumed prey, and together constituted 99 percent by volume of the total prey items consumed in this study. Moths dominated the insect fauna at middle and high elevations, and were also consumed by ʻōpeʻapeʻa significantly more than any other insect taxon at low elevations (Todd 2012). ʻŌpeʻapeʻa at low elevations selected moths and beetles in proportion to their availability in the environment. However, at middle-elevation sites, beetles accounted for 43 percent of the ʻōpeʻapeʻa diet, even though beetles comprised only 3.5 percent of the total insect availability at these sites. This pattern suggests that bats forage less selectively at low elevations and more selectively at middle elevations (Todd 2012), and may be partially caused by the presence of stressors at low elevations in this study area, such as coqui frogs (*Eleutherodactylus coqui*) that consume a large percentage of the available insect fauna in these areas (Beard 2007, Todd 2012).

A massive outbreak of the koa moth (Geometridea: *Scotorythra paludicola*) defoliated more than a third of the koa forest on the island of Hawaiʻi from 2013 to 2014. Although ʻōpeʻapeʻa detectability was notably lower during the outbreak year than in any year of the 5-year study conducted by Gorresen *et al*. (2013) at both Hakalau and Laupāhoehoe, Banko *et al*. (2014) suggested that this may have be due to the relative ease in which ʻōpeʻapeʻa reached satiation during the koa moth outbreak. Echolocation calls associated with searching and attacking insect prey peaked abnormally early in the night during the outbreak at Laupāhoehoe. Bats actively foraged over longer portions of the night and at lower success rates during non-outbreak times when prey (moth) densities were orders of magnitude lower. Elevated acoustic detections of ʻōpeʻapeʻa on Oʻahu have been associated with the presence of beetles (Gorresen *et al.* 2018a, 2018b).

Foraging and movement

Many studies have looked at how ʻōpeʻapeʻa move, forage, and use habitats across the islands (e.g. Todd 2012, Gorresen *et al*. 2013, Bonaccorso *et al*. 2015, Gorresen *et al.* 2015, Bonaccorso *et al*. 2016, Todd *et al*. 2016, Johnston *et al.* 2019). These studies found that, overall, bat activity and movements on the landscape are not determined by one variable, but by an interaction of a complex array of environmental factors. Seasonal changes in temperature, rainfall, wind, insect abundance and energetic costs associated with reproduction of ‘ōpe‘ape‘a all play important roles in movements and habitat use.

Spaces in which ‘ōpe‘ape‘a forage are extremely varied in physical structure, including forest gaps and clearings, forest edges along planted windrows of trees, above forest canopies, and along roads. These areas can occur in a range of habitats including undisturbed native forest, mature eucalyptus plantations having mixed understory trees and shrubs, lowland forest dominated by introduced trees, suburban and urban areas planted with ornamental trees, grassland/pasture, river gorges, arboretums, macadamia nut orchards, and coastal bays (Bonaccorso *et al*. 2015, Gorresen *et al*. 2013).

Gorresen *et al*. (2013) found a significant association between ‘ōpe‘ape‘a occupancy and the prevalence of mature forest cover. However, native vegetation was not related to occupancy. This might be due to the fact that lowland forests on the island of Hawai‘i, which are important for pupping, are almost exclusively nonnative vegetation, whereas the majority of the native forest remaining in Hawai‘i occurs at montane elevations.

Bonaccorso *et al*. (2015) examined the movement of 28 radio-tagged ‘ōpe‘ape‘a along the windward side of the island of Hawai‘i during the summer and fall. One-way movements by ‘ōpe‘ape‘a within a night covered distances of up to 11.3 kilometers (km) (7.0 miles). The mean foraging range was 230.7 ± 72.3 hectares (570.1 ± 178.7 acres) (n = 28 bats) which included two outliers, an adult male with a foraging area of 1,593 hectares (3,936 acres) and a subadult male with a foraging area of 1,316 hectares (3,252 acres). Foraging areas of individual bats overlapped in some cases (Bonaccorso *et al*. 2015). These findings suggest that foraging range is largely based on the prey quality present in the area; the poorer the quality of foraging, the further a bat needs to travel to acquire the resources it needs for survival. A bat’s full foraging range is likely important for its survival (F. Bonaccorso, pers. comm. 2011).

During summer, the mean core use area (the area that a bat used intensively for 50 percent of the time while it was radio-tracked) averaged 25.5 ± 6.9 hectares (63.0 ± 17.1 acres) (n = 28 bats), or about 11 percent of the mean foraging range (Bonaccorso *et al*. 2015). One subadult male had an unusually large core use area of 176 hectares (435 acres). Excluding this outlier, core use area averaged 19.9 hectares (49.2 acres) (n = 27 bats). Winter ranges were not determined. Core use areas did not typically overlap between radio-tagged individuals, which aligns with the territoriality observed between conspecifics. The core use area may represent an area in which a bat shows greater territoriality, because of the amount of time spent in the area. However, other ‘ōpe‘ape‘a that were not radio-tagged could have been present in the core use areas. The size of the core use area also varies with the quality of the resources within that area.

In east Maui, Johnston *et al.* (2019) mist-netted 20 bats in 78 nights of effort over a year-long period. Captures consisted of 12 adult males, 2 adult females, and 5 subadults. The group radio-tagged 16 bats and mapped 11 ranges, some of which appeared to overlap. In their preliminary analysis Johnston *et al*. (2018) found foraging areas on that island can range from 1,200 to 26,000 hectares (3,000 to 64,000 acres). The average size of the regular foraging area was about 1,200 hectares (3,000 acres).

The wide variability in size of foraging ranges and core use areas may be influenced by the highly fragmented Hawaiian landscape and the ability of ‘ōpe‘ape‘a, in the absence of any other bats, to exploit different food resources in a large number of diverse habitats depending on nutritional needs (Bonaccorso *et al*. 2015, Todd 2012; Gorresen *et al.* 2018b). Suitable foraging areas can be quite widely separated, and ‘ōpe‘ape‘a easily move within a night from sea level to elevations above the cloud inversion layer (approximately 1,700 meters [approximately 5,600 feet]) in order to forage in dry weather (Bonaccorso *et al*. 2015). This bat species is known to fly more than 19 km (12 miles) one-way in the course of a night, usually returning to its original roost site by sunrise (Barclay 1989). One radio-tracked male foraged at different altitudes on several nights, allowing it to avoid rainfall at low elevations (Bonaccorso *et al*. 2015).

As such, temperature, wind and rainfall all appear to influence ‘ōpe‘ape‘a foraging activity and movements (Todd 2012, Gorresen *et al*. 2015, Todd *et al*. 2016, Bonaccorso *et al*. 2016). Todd (2012) found a temperature and rainfall model the best predictor for ‘ōpe‘ape‘a activity on Hawai‘i. However, temperature may be a stronger environmental influence on bat activity as some bats move to different elevations seasonally. Individual bats can and do fly more than 18 km (11 miles) in less than a half hour (Bonaccorso *et al.* 2012, as cited by Todd 2012), a distance greater than a round trip from the ocean to the summit of Mauna Kea. ‘Ōpe‘ape‘a may easily roost at high elevations and forage at low elevations or vice versa during any time of the year in order to obtain optimal foraging conditions (Gorresen *et al*. 2013, Gorresen *et al*. 2015). Additional studies have demonstrated that Hawaiian hoary bats can range between habitats and elevations within a single night to target optimal local foraging opportunities, with bats spending 20 to 30 minutes hunting in a feeding range before moving on to another (Bonaccorso 2010).

Females are solely responsible for rearing young, and energy demands increase significantly from pregnancy through lactation (Barclay 1989). Dependent pups left hanging in a tree are susceptible to drops in temperature. As lactation progresses, bats with dependent young spend more time foraging per night and less time roosting with their young (Barclay 1989). Barclay postulated that the time spent foraging by females is constrained by the need to keep newborn young warm at the roost. Because of this constraint the females are adjusting their foraging behavior to meet the current energy demand, rather than foraging to store energy for use at a later time (Barclay 1989). Menard (2001) observed progressively earlier times of emergence in the bat population as the period of lactation advanced from June to August, although not strongly correlated with ambient temperature.

Environmental conditions and the physiological status of the ‘ōpe‘ape‘a influence its activity. However, correlating particular environmental or physiological variables with specific flight activities and behavior remains difficult. No single environmental condition can be used to predict a specific behavior or activity. A variety of variables (e.g., humidity, rain, light, temperature, and prey availability) and their respective ranges influence behavior. Gorresen *et al*. (2015) found higher rates of bat detection on O‘ahu when nightly wind speeds dropped to a low relative to the previous night; mean speeds were less than 4.6 meters per second (m/s) (10.3 miles per hour [mph]); and maximum speeds were less than 8.2 m/s (19.0 mph). The conditions that favored the highest proportion of bat detections included conditions where maximum wind speeds were less than or equal to 7.7 m/s (17.2 mph), or between 7.7 and 8.7 m/s (17.2 and 19.5 mph) when temperatures were greater than 21.5 °Celsius (C) (71 °Fahrenheit (F). Conditions that favored the lowest bat activity included humidity levels greater than 90.0 percent and maximum wind speeds greater than 8.7 meters per second (19.5 mph), or humidity levels less than or equal to 90.0 percent and maximum wind speeds greater than 12 meters per second. The proportion of detections were also low where wind speeds were between 7.7 and 8.7 meters per second (17.2 and 19.5 mph) and temperatures were less than or equal to 21.5 °C (71° F). With regard to precipitation, the highest rates of activity were when nightly maximum wind speeds were less than or equal to 8.3 meters per second (19 mph) and cumulative rain less than or equal to 0.8 millimeters (0.03 inches). Conditions that favored the lowest activity rates included maximum wind speeds greater than 9.8 meters per second (22 mph), where humidity levels were greater than 85.0 percent, and temperatures were less than or equal to 21.5 °C (71° F). In general, lower wind speeds and warmer temperatures appear to increase detection likelihood. ‘Ōpe‘ape‘a were more likely to be detected when barometric pressure was relatively low (less than or equal to 972 millibars), but rising over a period of at least 24 hours. Rising barometric pressure may indicate improved conditions for foraging and overall activity and/or increased availability of insect prey. The results indicate that relatively higher bat activity occurred as storm fronts passed and weather conditions were improving. Video detections of bats at wind energy turbines declined with increasing humidity. A likely biological explanation for fewer bat detections at high levels of humidity is that foraging by echolocation may be less efficient in wet air.

Bonaccorso *et al*. (2015) documented that flight activity ceased during periods of rain within a night, as bats sheltered in night roosts until conditions improved. ‘Ōpe‘ape‘a activity increased at low and middle elevations during periods of lower mean rainfall, and increased at high elevations during non-reproductive periods with higher seasonal mean rainfall. On Hawai‘i, movement into high elevations during winter provides better foraging conditions, as rainfall at high elevations at this season is half of that at low elevations, while the availability of insect prey is the same as at low elevations. Low annual rainfall with increased clear, calm nights can lead to improved conditions for bat foraging, which possibly contributed to locally increased bat activity in a Maui study in 2012 (Todd *et al*. 2016). Conversely, in the 2 following years higher rainfall and possibly other climatic variables may have contributed to decreased time foraging within the study area (Todd *et al*. 2016).

Bonaccorso *et al*. (2016) examined altitudinal movements involving previously unknown use of caves by ‘ōpe‘ape‘a during winter and spring (November 2012 to April 2013) in the Mauna Loa Forest Reserve (MLFR) on the island of Hawai‘i. Thirteen lava tube cave entrances at elevations between 2,200 and 3,600 meters (7,200 and 11,800 feet) were acoustically monitored each month. The occurrence of feeding buzzes around cave entrances and visual observations of bats flying in an "acrobatic fashion" in cave interiors point to the use of these spaces as foraging sites (Bonaccorso *et al*. 2016). *Peridroma* moth species (Family: Noctuidae), the only abundant nocturnal flying insect sheltering in large numbers in rock rubble and on cave walls in the MLFR, apparently serve as the principal prey attracting ʻōpeʻapeʻa during winter to these lava tube caves. Bat foraging activity, evidenced by the amount of search and feeding buzz calls in the MLFR, is correlated with relatively low wind speeds, air temperatures above 6 °C (43° F), and conditions believed to be free of heavy fog and rain, similar to what Gorresen *et al*. (2015) observed on Oʻahu. Winds above 6 m/s (13.4 mph) generally reduce vespertilionid bat flight activity (Arnett *et al*. 2008, Schuster *et al*. 2015).

Visual searches found no evidence of ʻōpeʻapeʻa sheltering by day in the caves, nor were there signs of hibernacula (Bonaccorso *et al*. 2016). However, the presence of over 300 skeletons and mummies of bats in cave interiors indicated ʻōpeʻapeʻa do occasionally fly deep into the caves. One possible way for ʻōpeʻapeʻa in Hawaiʻi to avoid inclement weather conditions while hunting for aerial nocturnal insects is to fly to elevations above the cloud inversion layer, a condition frequently occurring above the 1,700-meter (5,600-foot) elevation in the MLFR (Giambelluca and Schroeder 1998). Bats make particularly heavy use of the high-elevation caves in the MLFR during December and January (Bonaccorso *et al*. 2016); thus the MLFR and other areas of similar elevation with lava tube caves may be particularly important as winter foraging areas. Altitude has previously been shown to be a significant determinant of the timing of emergence for foraging each night by ʻōpeʻapeʻa during the pre-pregnancy period (January to March), but not during any other period (Menard 2001).

Seasonal torpor in ʻōpeʻapeʻa has not been researched extensively. Understanding the role of torpor and how bats in Hawaiʻi facilitate it at different elevations and temperatures will provide important ecological answers to habitat use and offer insight into determining times for timber harvest that minimize impact on the bat. For example, if bats choose to move to higher elevations during winter months in order to induce long-term torpor then these areas may not be suitable for tree harvest during the winter months. Understanding torpor also will be important when examining the possible effects of climate change.

New Threats:

Wind Turbines
In 2015, the State of Hawaiʻi passed a bill (HB623) setting a target of achieving 100 percent renewable energy. The Hawai'i Clean Energy Initiative (HRS 196-10.5) and Renewable Portfolio Standards (HRS 269-92) specifies that the State of Hawai'i will establish a renewable portfolio standard of 100 percent of net electricity sales from renewable sources by 2045. Wind energy currently accounts for 29 percent of the renewable energy produced Statewide and may be expected to increase due to this goal. The Hawaiian Electric Company issued a renewable energy request for proposals seeking to develop an additional 60 megawatts (MW) of renewable energy on Maui (HECO 2018). No new wind energy projects were identified for Maui as a result of this process. It is not known if a similar request will be initiated in the future, but the start of operations of a new project in the next 5 years is unlikely given that no projects were identified in 2018. All new proposed wind projects that would pose a risk to Hawaiian hoary bats would be expected to offset authorized take impacts that could not be avoided, through an approved habitat conservation plan (HCP) under the Endangered Species Act (ESA).

Quantification of the number of bat fatalities is largely limited to those associated with wind energy projects because they conduct systematic and rigorous compliance monitoring. Based on fatality monitoring, land-based wind energy facilities that operate between dusk and dawn are presently the greatest known source of ʻōpeʻapeʻa fatalities that is being quantified and tracked. The impacts of land-based wind energy projects are substantially higher than was anticipated at the time of the previous 5-year review. Currently, there are eight operating wind facilities and one under construction in Hawaiʻi. Of those nine facilities, six have approved HCPs and Incidental Take Permits (ITP) under section 10(a)(1)(B) of the ESA, one (Kahuku) is under a Federal Biological Opinion and State-approved HCP, and one (Lalamilo) has applied for an ITP and has a draft HCP, but has reportedly ceased nighttime operation. The other operating facility (Hawi) is developing a draft HCP. As of 2020, there had been 95 observed ʻōpeʻapeʻa fatalities at the 6 facilities systematically monitoring and reporting bat fatalities. The modeled number of fatalities adjusts for imperfect detection of fatalities and includes indirect take from the loss of dependent young. Based on an 80 percent credibility standard used for modeling fatalities, the number of direct and indirect bat fatalities at all existing commercial wind projects on Maui are estimated to not exceed 11.3 bats per year (Auwahi Wind 2020, Kaheawa Wind Power I 2020, Kaheawa Wind Power II 2020). On Oʻahu, direct and indirect bat fatalities at all existing and permitted wind projects are estimated to not exceed 14.7 bats per year (Kawailoa Wind Power 2020, Kahuku Wind Power 2020; Tetra Tech 2016). On the island of Hawaiʻi, direct and indirect bat fatalities at the one facility conducting monitoring are estimated to not exceed 3.2 bats per year (SWCA 2018). The other facilities do not have ITPs and do not report systematic monitoring results to the Service. The numbers provided for the annual estimations do not represent actual observed fatalities; rather, the numbers represent what the Service is confident has not been exceeded per year because of imperfect detection. Additional wind energy facilities on Oʻahu or Maui, or repowering of existing facilities on Hawaiʻi may occur in the future. Such projects would be expected to seek ITPs with associated

HCPs. We do not currently anticipate construction of wind energy projects on Kauaʻi, Molokaʻi, or Lānaʻi.

Bat behavior at turbines also plays a role in risk of fatality. Cryan *et al*. (2014) observed that wind speed and the speed of the rotating turbine blades influences the way bats approach the turbines. Bats approached turbines less frequently when the blades were spinning fast. To avoid and minimize ʻōpeʻapeʻa fatalities in Hawaiʻi, wind facilities are using low wind speed curtailment (LWSC) and blade feathering. Wind turbine generators have a manufacturer-designated curtailment wind speed (also known as "cut-in speed"), below which the turbine blades are idle or rotating very slowly and not producing power, and above which the blades overcome drag to produce lift during rotation and produce power. Manufacturer's cut-in speeds typically range from 3.0 to 4.0 m/s (6.7 to 8.9 mph). By raising the cut-in speed to a higher wind-speed setting, the risk to bats flying in wind speeds below the cut-in speed is diminished. Increasing cut-in speeds 1.5 to 3.0 m/s (3.3 to 6.7 mph) above the manufacturers' cut-in speed have been correlated with a reduction in number of bat fatalities in areas where bat fatalities are frequent on the mainland United States (U.S.) and abroad (Good *et al.* 2011, Arnett *et al.* 2013). Modifying the acceleration and deceleration profile of the turbine blades when wind speeds are below the cut-in speed has also been associated with reduced bat fatalities. Feathering the blades when wind speeds are below the cut-in speed reduces the wind force on the blades and slows the rotation of the blades to 0 to 3 rotations per minute. Many studies have shown reductions in bat fatalities may be achieved by feathering blades to be parallel to the wind, or by a low rotational-speed idle approach (Baerwald *et al*. 2009; Young *et al.* 2011, 2012, 2013; Good *et al.* 2012). Studies have also evaluated the benefits of combined feathering and low wind speed curtailment. Significant reductions in bat fatality rates have been demonstrated on the mainland and abroad when cut-in speeds are raised incrementally from 3.5 to 4.5 to 5.0 and 5.5 m/s (8 to 10 to 11 and 12 mph) (Arnett *et al.* 2009, 2010, 2011, Good *et al.* 2013, 2015, 2016, 2017, 2018, Good and Adachi 2014, Hein *et al.* 2014). In Hawaiʻi, the cut-in speeds that wind energy facilities use vary from 5.2 m/s (11.6 mph) to 6.9 m/s (15.4 mph), depending on facility and time of year.

The goal of these operational approaches is to limit the time turbines are spinning during periods of lower wind speed, that is, when bats are more likely to be flying. While LWSC and feathering the blades does appear to reduce the number of bat fatalities at wind facilities locally, it is difficult to determine exactly how effective the methods are due to the infrequency (rarity) of fatalities, stochastic variability, and lack of non-curtailed turbines to compare against in Hawaiʻi. ʻŌpeʻapeʻa can and do fly in wind speeds that are above the wind speeds used as the cut-in speeds. According to Gorresen *et al.* (2015), ʻōpeʻapeʻa behaviors, including close approaches to turbine monopole, blades, and nacelle, occur across a range of wind speeds typically from 0 to 9.6 m/s (0 to 21.5 mph), and occasionally from 12 to 15 m/s (26.8 to 33.6 mph). In general, bats were detected more frequently at low blade-rotation speeds (less than 1.0 m/s [2.2 mph]) and less frequently at intermediate (1 to 10 m/s; 2.2 to 22.4 mph) and high speeds above 10 m/s (22.4 mph) (Gorresen *et al.* 2015). Prevailing wind speeds in the group's Oʻahu study site ranged from 5.5 to 8 m/s (12.3 to 17.9 mph) and may have contributed to the upper limit at which bats were observed flying. The timing of operational minimization actions (feathering and low wind-speed curtailment) also plays a role in reducing the risk to bats.

Gorresen *et al.* (2015) found the hourly rate of nightly ʻōpeʻapeʻa detection (number/hour/turbine) was highly variable but more than doubled from mid-May to mid-November. Acoustic and thermal video detection and lack of roosting resources suggests bats are not constantly present at a wind project but may use sites opportunistically or intentionally, depending on resources and season. In Hawaiʻi, implementation of LWSC is not based on actual bat presence each night, as there is no way to reliably detect bats and shut down blade rotation in response to their presence. Rather, LWSC is implemented year-round. While this is assumed to reduce the risk of bat fatalities, it also reduces electricity generated, and it does not eliminate all fatalities.

The use of ultrasonic deterrents, which would deter bats from flying in the immediate vicinity of spinning turbines, is undergoing evaluation and refinement on the U.S. mainland. There the technologies have shown mixed efficacy depending on location and bat species (Weaver *et al*. 2018; BCI 2019). Most recently, Kawailoa Wind announced their intent to deploy bat deterrents at Kawailoa Wind facility on Oʻahu for evaluation. This is the first multi-turbine evaluation of its kind in Hawaiʻi, attempting to avoid and minimize ʻōpeʻapeʻa fatalities while still producing the energy necessary under the existing power purchase agreement with the local power company. Systematic monitoring will be used to determine the level of efficacy the deterrents have in minimizing ʻōpeʻapeʻa fatalities over time. At the time of this 5-year status review, the only approach that would definitively avoid take of ʻōpeʻapeʻa would be to fully curtail all turbines, on all islands, from one hour prior to dusk to one hour after dawn. This strategy, while effective, is not considered a long-term strategy for existing wind facilities because of project viability and power production constraints imposed by local power utilities.

In order to offset unavoidable take, wind facilities operating under an ITP implement a variety of conservation projects, including land purchase and protection, forest and wetland restoration, and targeted research projects for the ʻōpeʻapeʻa. The implementation of such projects is anticipated to offset impacts, resulting in "no net loss" for the species. However, given the limited information on basic life history needs and difficulty in tying land-based mitigation projects to a specific increase in ʻōpeʻapeʻa numbers, some uncertainty remains regarding the effectiveness of land-based mitigation projects for ʻōpeʻapeʻa. Compensatory mitigation projects currently rely on adaptive management programs to ensure measures of success are met and fatalities are effectively offset using best science available. The selected research projects are expected to contribute to our collective understanding of the species' needs and life history parameters for the long term. These research needs are considered some of the highest priority recovery actions for ʻōpeʻapeʻa.

Timber Harvest

Timber harvest of trees greater than 4.6 meters (15 feet) in height when ‘ōpe‘ape‘a and their dependent pups are present continues to be a threat (Amlin and Siddiqi 2015). Non-volant (not yet able to fly), dependent pups are reliant on their mother to move them out of a roost tree during timber harvest. The ability of a female ‘ōpe‘ape‘a to accomplish this move is constrained by the weight of the pup and perception of the threat. Detection of roosting bats in trees with thermal imaging is limited by canopy structure and relatively small differences between ambient temperatures and ‘ōpe‘ape‘a body temperatures. Silviculture and biomass harvest operations exist primarily on the islands of Kaua‘i and Hawai‘i. The Service and DOFAW recommend not cutting trees above 4.6 meters (15 feet) between June 1 and September 15 to avoid impact to dependent (non-volant) bat pups (Amlin and Siddiqi 2015). ‘Ōpe‘ape‘a roost in a wide variety of trees (native and nonnative) and are widely distributed across all islands, thus limited removal of trees outside of the pupping season is not currently anticipated to result in adverse effects to Hawaiian hoary bat populations. However, removal of functioning habitat that has taken years to develop, might be expected to have impacts on the activity and territoriality of bats. Degradation or removal of roosting and foraging resources may increase the distance ‘ōpe‘ape‘a need to travel to obtain necessary sustenance for survival and reproduction, and may reduce fitness. The Service is working with the timber industry to avoid, minimize, and mitigate impacts should harvest occur during the pupping period.

Coqui Frogs

Coqui frogs, introduced to the State of Hawai‘i in the late 1980s (Woolbright *et al*. 2006), are widely established on the island of Hawai‘i, and are found in smaller areas on the islands of Maui, O‘ahu, and Kaua‘i (Hawai‘i Invasive Species Council 2018). The highest densities of frogs (20,000 to 40,000 individuals per hectare [8,000 to 16,000 per acre]) are found at elevations lower than 670 meters (2,200 feet) above sea level (Beard *et al*. 2009), but the frogs are now spreading to mid-elevation forests (900 to 1,200 meters [3,000 to 3,900 feet]) and have the ability to thrive and successfully overwinter at higher elevations in Hawai‘i (Kraus and Campbell 2002, Hawai‘i Invasive Species Council 2018). They have a limited number of predators (mongoose, rats, and feral cats), which enables these frogs to become successful invaders across wet forest habitats and allows their populations to grow extraordinarily dense, in comparison to their native habitat of Puerto Rico (Woolbright *et al*. 2006). The spread to higher elevations poses increased threat to insect resources that overlap with the ‘ōpe‘ape‘a. An analysis of coqui frog diets at lowland sites on the islands of Hawai‘i and Maui found that many of the invertebrates consumed by the frogs were leaf litter insects, as well as a large number of flying insects, indicating that these frogs are actively foraging while climbing trees (Beard 2007). Dietary analysis of the coqui frog on the island of Hawai‘i showed that aerial insects make up 33.8 percent of the diet (Bernard and Mautz 2016). High frog densities (20,000 to 40,000 frogs per hectare [8,000 to 16,000 per acre]) result in the ability to consume 4,500 to 56,000 prey items per hectare (1,800 to 23,000 per acre) every night, with 1,500 to 19,000 of these per hectare (600 to 7,700 per acre) being aerial insects (Bernard and Mautz 2016). As determined from the USGS aerial arthropod data used by Todd (2012), low-elevation study sites had an estimated 17,000 to 21,000 available aerial insects per hectare (6,900 to 8,500 per acre), and the high-elevation sites were estimated to have 20,000 to 74,000 available aerial insects per hectare (8,000 to 30,000 per acre). At low elevation, coqui frogs could potentially consume up to 91 percent of the available aerial arthropods. While the diet of ʻōpeʻapeʻa is consistently dominated by moths at both high and low elevations, the bats displayed foraging preference at high elevations rather than taking prey proportional to availability as they do at low elevations (Todd 2012). In addition, the ground insect feeding behavior of the frogs can result in the consumption of larval stages of moths and beetles, thereby reducing the adult aerial prey availability of moths and beetles. Increases in coqui frog densities at higher elevations has the potential to change the foraging patterns of ʻōpeʻapeʻa. Bats were found to consume fewer Coleoptera prey at low elevations where there were dense coqui frog populations compared to areas with few to no frogs (Whitaker and Tomich 1983). While the overall degree of dietary overlap between the ʻōpeʻapeʻa and the coqui frog was relatively low, the percentage of total available aerial arthropods shared by both species could be up to 64.9 percent (Bernard and Mautz 2016). This estimate identifies the range of competition the ʻōpeʻapeʻa may have in low-elevation sites shared with the coqui frog. The competitive impact of the invasive frog predator on the ʻōpeʻapeʻa may be measurable in areas that overlap with coqui frog occupancy, due to the high population densities the frog achieves and its continued elevational spread throughout the islands.

Climate Change

Climate change may exacerbate the impacts of coqui frogs by allowing an expansion of their numbers into higher elevation areas (Gayle 2020), where they would compete with ʻōpeʻapeʻa by changing the composition of the insect fauna available to forage. Other impacts from climate change to ʻōpeʻapeʻa are unknown. Warmer temperatures may allow an expansion of pupping habitat into higher-elevation areas, but may also affect habitat conditions by effecting changes to the prey base, resulting in suboptimal foraging conditions. These impacts may be mitigated by the ability of the ʻōpeʻapeʻa to range widely in search of resources and its generalist diet.

New management actions:
Conservation actions and compensatory mitigation for ʻōpeʻapeʻa include reforestation and restoration of native ecosystems, creating foraging opportunities, and long-term protection of areas from clearing and development.

- Compensatory mitigation for take of ʻōpeʻapeʻa at wind turbines on Maui contributes to habitat and food resource improvement at the Kahikinui Forest Reserve (FR) and the adjoining Nakula Natural Area Reserve (NAR) (Kaheawa Wind Power II 2020). A management plan was developed for the area to improve 3,200 hectares (8,000 acres) of habitat, through 11.7 km (7.3 miles) of fencing for exclusion of nonnative herbivores, restoration of native vegetation, weed control, and predator removal (State of Hawaiʻi 2015b). Ungulates were cleared from the fenced area in 2016, and monitoring facilitates fence maintenance and ingress detection, while restoration of the forest through weeding and outplanting continues. Additional monitoring of ʻōpeʻapeʻa is planned to determine the effectiveness of the restoration as compared to pre-restoration baseline levels. Kahikinui is protected from development by a permanent conservation easement.

- Another compensatory mitigation project for the ‘ōpe‘ape‘a funded by a wind energy project on Maui is restoration of approximately 52 hectares (128 acres) of pastureland at Pu‘u Makua, located in the Waihou area of Maui. Restoration actions began in 2012 and have included installation of an ungulate-proof fence, ungulate removal, removal of invasive vegetation, and native plant restoration (Auwahi Wind 2020). This parcel was also placed into a conservation easement held by the Hawaiian Islands Land Trust to be protected in perpetuity.

- On O‘ahu, compensatory mitigation from a wind energy project has resulted in the restoration and management of 32 hectares (79 acres) of the ‘Uko‘a wetland area to increase its foraging habitat value for ‘ōpe‘ape‘a, as well as managing 16 hectares (40 acres) surrounding the wetland to create foraging lanes and increase native tree species favorable to bat roosting. The management plan was finalized in August 2014 (H.T. Harvey & Associates and SWCA 2014), and amended in March 2016. The wetland was fenced and maintained to keep the area inside ungulate-free. Invasive vegetation, primarily water hyacinth (*Eichhornia crassipes*), has been removed from the open-water areas of the wetland to improve insect production for bat foraging. Quarterly maintenance visits will continue through year 2032 to remove any small areas of water hyacinth that have regenerated. Nonnative trees were removed to create 5-meter (16-foot) wide corridors that have been shown to support bat foraging (Jantzen 2012, Kawailoa Wind 2017). Insect collection was conducted in June to October 2014 and June to October 2015 and submitted for analysis to establish baselines for ‘ōpe‘ape‘a prey levels and composition prior to the removal of invasive vegetation and restoration actions. Baseline acoustical monitoring for ‘ōpe‘ape‘a at the site began in April 2012 and is ongoing (Kawailoa Wind 2017).

- Approximately 1,142 hectares (2,822 acres) of the Helemano Wilderness Area located near Wahiawā, in central O‘ahu, was acquired in October 2018, protecting the area from development for perpetuity. The land will be managed for multiple uses, including for the benefit of the ‘ōpe‘ape‘a and other protected and native species. Helemano Wilderness Area includes significant tracts of native forest habitat within the documented range of the ‘ōpe‘ape‘a that are at risk due to encroachment of invasive plant and animal species and potential anthropogenic activities (e.g., residential development). The property also includes non-forested fallow agricultural areas suitable for forest restoration and this mix of forested lands and fallow agricultural lands is anticipated to provide foraging and roosting habitat for ‘ōpe‘ape‘a. ‘Ōpe‘ape‘a have been detected in, and are highly likely to occupy, the immediate areas surrounding the property. The area will also support the movement of bats between central O‘ahu and the North Shore, along the major forested parcels in the Ko‘olau mountains.

- Additional protection of ‘ōpe‘ape‘a habitat occurs on the North Shore of O‘ahu at Pūpūkea Mauka. The upper portions of the Waimea River watershed (Waimea Native Forest) was purchased by the State of Hawai‘i (DLNR *in litt.* 2019) and will be managed for conservation of ‘ōpeʻapeʻa and other native species. This 1,504-hectare (3,716-acre) property consists predominantly of native forest and ‘ōpeʻapeʻa have been documented regularly in and around the property at high occupancies.

- Approximately 1,389 hectares (3,433 acres) of the Kamehamenui Forest located on the leeward side of Haleakalā, Maui, was acquired in 2020 by DOFAW, protecting the area from development and enhancing mitigation opportunities for ‘ōpeʻapeʻa on the island. The DOFAW will fence portions of the property, followed by ungulate control and forest restoration. Management of the natural resources in the area will include: (1) conservation of the native subalpine habitat including fencing, ungulate removal, and restoration for ‘ōpeʻapeʻa and other endangered species and native communities; and (2) native forest restoration below the subalpine habitat to connect existing habitat for ‘ōpeʻapeʻa. The Kamehamenui Forest is likely occupied by ‘ōpeʻapeʻa, based on detections on all sides of the property in similar terrains. The property borders Haleakalā National Park, the Kula Forest Reserve, and nearby open ranches to provide transit connectivity for ‘ōpeʻapeʻa movement.

Five-Factor Analysis (threats, conservation measures, and regulatory mechanisms)

A. Present or threatened destruction, modification or curtailment of its habitat or range

Overall, over the last 8 years, ‘ōpeʻapeʻa have been documented to occur over a much broader range than was known at the time of listing or when the species’ recovery plan was finalized. In most locations where acoustic monitoring has been conducted, ‘ōpeʻapeʻa have been present at some point during the year, including in urban, semi-urban, and agricultural areas. However, the ability of bats to move large distances nightly to take advantage of dispersed food resources, as well as documented seasonal movements, make interpreting these results challenging. While there are no monitoring methods that can quantify the abundance of ‘ōpeʻapeʻa on each island, all of the major Hawaiian islands are now recognized as providing roosting, breeding, and/or foraging habitat for the species.

Tree trimming and timber harvest of trees above 4.6 meters (15 feet) in height poses a threat to non-volant, dependent bat pups if they are present. Based on existing data, the Service recognizes the period of time for pups to be non-volant as June 1 through September 15. Therefore, timber harvest during this time likely results in some level of mortality annually. Silviculture and biomass harvest operations exist primarily on the islands of Kauaʻi and Hawaiʻi. The Service recommends not cutting trees above 4.6 meters (15 feet) between June 1 and September 15 to avoid impact to dependent (non-volant) bat pups. ‘Ōpeʻapeʻa roost in a wide variety of trees (native and nonnative) and are widely distributed across all islands, thus limited removal of trees outside of the pupping season is not currently anticipated to result in adverse effects to ‘ōpeʻapeʻa populations. However, removal of a functioning habitat that has taken years to develop might be expected to have impacts on the activity and territoriality of bats.

An estimated 597,000 hectares (1.475 million acres) of forest habitat occurs across the major Hawaiian islands (Reeves and Amidon 2018). About 50 percent or 300,000 hectares (700,000 acres) of dry, mesic, and wet forest habitat is owned by County, State, or Federal government agencies. Of that total forested habitat, about 79 percent or 470,000 hectares (1.163 million acres) is on O‘ahu, Maui, and Hawai‘i, the three islands where wind facilities are located and where almost all of the cumulative effects to ‘ōpe‘ape‘a are occurring. Of that, about 54 percent or 255,000 hectares (630,000 acres) are owned by government agencies and about 17 percent or 81,000 hecatres (200,000 acres) are currently designated as conservation lands. Additional privately-held acreage occupied by the ‘ōpe‘ape‘a is protected by conservation easements throughout the State.

B. Overutilization for commercial, recreational, scientific, or educational purposes

This factor is not considered a threat to the ‘ōpe‘ape‘a.

C. Disease or predation

Predation is likely to be a source of mortality but has not been quantitatively documented and is not known to be a significant threat to the ‘ōpe‘ape‘a. Predation on bats by owls has been documented outside Hawaii (e.g., Roulin and Christe 2013) and it is likely that some mortality of ‘ōpe‘ape‘a occurs from predation by introduced barn owls (*Tyto alba*) or the native pueo (*Asio flammeus sandwichensis*). Dependent pups of ‘ōpe‘ape‘a are also likely to be vulnerable to predation at roost sites by introduced rats (*Rattus* spp.) Feral cats have been observed attacking a grounded bat.

White-nose syndrome, a disease of bats caused by the invasive fungus *Pseudogymnascus destructans*, was introduced from Europe to North America in 2006 and has caused mass mortality of multiple species of hibernating bats. To date there is no evidence that this fungus has been introduced to the Hawaiian islands, and screening of lava tube caves on the island of Hawai‘i resulted in no positive tests (Zhelyazkova *et al.* 2019). However, this disease could be a significant threat to ‘ōpe‘ape‘a if the fungus were to become established in the Hawaiian islands.

D. Inadequate existing regulatory mechanisms

Hawai‘i has regulations in place to protect the ‘ōpe‘ape‘a, but the implementation of current biosecurity programs is inadequate to prevent the introduction and spread of invasive species, such as coqui frog (Beard 2007, Bernard and Mautz 2016), that affect ‘ōpe‘ape‘a habitat and food resources.

E. Natural or manmade factors affecting its continued existence

Collisions
Collisions with man-made objects are a source of mortality to ‘ōpe‘ape‘a. Bats colliding with fishing line, vehicles or vehicle antennas, though rarely reported, have been documented. The impacts from these sources are largely unquantified because of the lack

of systematic monitoring and reporting. Based on the incidents reported, the impact is believed to be minor.

On the other hand, bat fatalities attributable to collisions with wind turbines operating between dusk and dawn are quantified. Projects conduct mandatory systematic weekly or twice-weekly monitoring for downed wildlife at all operating turbines and the fatalities are counted. In addition modeling is used to estimate the number of fatalities that may have occurred because of imperfect detection and to account for the loss of dependent young during the breeding season. The presence of wind farms operating at night poses a risk to bats on the islands of Oʻahu, Maui, and Hawaiʻi. Movement of bats between islands is expected to be low. The absence of commercial wind facilities on Kauaʻi, Lānaʻi, and Molokaʻi suggests that the bat populations on those islands do not face the same level of risk. The entire Statewide population of ʻōpeʻapeʻa are not at direct risk of extirpation from the limited operation of the wind farms on the islands of Oʻahu, Maui and Hawaiʻi, as not all individuals are likely to enter wind project sites and be killed.

However, wind turbines operating at night when ʻōpeʻapeʻa are active could potentially cause a localized reduction in the population. The extent of this reduction is largely unknown because it depends on how rapidly a niche vacated by a fatality is filled, and on the behavior of the resident ʻōpeʻapeʻa population. ʻŌpeʻapeʻa are highly mobile, capable of using fragmented habitats, and are more widespread than previously thought. Acoustic monitoring at wind facilities has not shown a decrease in bat activity. While bats have been known to travel up to 20 km (12 miles) in a night, the bats tend to focus their activity in areas where food and sheltering resources are available and spend the majority of their time in their core use area. A local effect on the bat population is possible if the core use area overlaps with the turbine sites because of the slightly higher probability of turbine encounter during nightly usage. This local effect on population could impact the species, either by reducing genetic diversity or by reducing the local population below a threshold that, with the contribution of other mortality factors, would cause the population to decline. Mobility of the bats provides an adaptive ability to sustain gene flow, at least within an island. Mortality of adult bats also results in loss of their future productivity. Bats may live up to 10 years, although it is unknown if they breed each year, nor for how many years they may produce young. The loss of an adult bat would also foreclose future additional recruitment by its progeny into future generations of bats on that island. In the case of wind energy-associated fatalities, mitigation actions are focused on the island on which the take is occurring, to minimize possible impacts to genetic diversity within an island population.

Acoustic detection has its limitations as discussed earlier in this document. Reduced echolocation further complicates monitoring hoary bat activity using passive acoustic monitoring (Corcoran and Weller 2018). Gorresen *et al*. (2015, 2018b) conducted a side-by-side comparison between thermal imaging videography and acoustical monitoring and found that the acoustical detectors detected only 8 percent of the bat activity captured by thermal videography, though thermal videography has its own limitations, such as limited field of view (Gorresen *et al.* 2015). Over the last decade, acoustic monitoring has been used to assess pre-construction fatality risk of wind turbines and post-construction bat activity (Weller and Baldwin 2012). Acoustic detections at operating wind energy facilities on O‘ahu (Kahuku Wind Power 2020, Kawailoa Wind 2020) and Maui (Kaheawa Wind Power I 2020, Kaheawa Wind Power II 2020; Auwahi Wind 2020) did not document a decline in general bat activity since operations began, although recordings do not necessarily equate to the number of bats in the area, and some improvements to microphones and location and number of detectors has since occurred. The evidence provided by Corcoran and Weller (2018) showed hoary bats on the mainland sometimes fly without echolocation or use micro calls that are not detected by acoustical detectors. This silence may help to explain the inconsistent ability to predict the potential for post-construction ‘ōpe‘ape‘a fatalities at wind facilities (Kaheawa Wind Power I 2020, Kaheawa Wind Power II 2020, Kawailoa Wind Power 2020, Auwahi Wind 2020, Hein *et al*. 2013). The inability to acoustically detect bat activity also complicates evaluations of habitat use related to management actions.

Wind energy facilities with approved HCPs include mitigation actions that are expected to help offset the authorized incidental take impacts to Covered Species. These actions include: (1) reforestation and restoration of foraging and roosting habitats, installation of water features, and removal of invasive species that degrade water sources, roosting, and foraging habitat of the bats; (2) conducting high-priority research to inform and improve management for the benefit of bats; and/or (3) acquisition of suitable habitat and protection of that land for perpetuity. The required measures of success for reforestation or restoration activities are objective and based on best science to appropriately gauge progress toward habitat improvements. All pending and approved ITPs and associated HCPs must include monitoring to document impacts to the ‘ōpe‘ape‘a and the effectiveness of mitigation actions in addition to adaptive management. This combination of monitoring and adaptive management allows the Service and State wildlife agencies to track compliance with the ITP, State incidental take license, and HCP, respond to conditions that indicate take or mitigation is not meeting the success criteria, and take corrective actions to ensure mitigation needs are met. Accordingly, project-related take impacts associated with these HCPs are likely to be avoided, minimized, and mitigated using the best available scientific practices and adaptive management.

Barbed Wire

Bat mortality caused by individuals becoming snagged on barbed wire has been documented. Annual mortality estimates range from 0 to 0.8 ‘ōpe‘ape‘a per 100 kilometers (0 to 1.3 per 100 miles) of barbed wire (Zimpfer and Bonaccorso 2010). Most barbed wire fences are not systematically monitored and the bat fatalities due to snagging may be quickly taken by predators or scavengers. In addition, the surrounding landscape may affect the risk of bat collisions with a barbed wire fence. Although observed bat fatalities are uncommon, the extent of the impact of barbed wire fences is largely unknown. The Service recommends removal or replacement of barbed wire with smooth wire when providing technical assistance and in all formal and informal consultations. Barbed wire usage is expected to decrease Statewide, but the amount of remaining barbed wire in use Statewide is unknown. Based on the low estimates of mortality related to bat impalement on barbed-wire fences and the decrease in barbed wire use, this impact is not expected to contribute significantly to cumulative effects to this species.

Pesticides
Pesticide use may have an impact on ʻōpeʻapeʻa by reducing or altering the prey population, or through biomagnification (concentration of toxins from food sources) via the prey base. Effects are mostly unknown. Trace amounts of rodenticide residues have been detected in tissues from 2 out of 21 ʻōpeʻapeʻa carcasses examined (USFWS unpubl. data), but there is currently no data available in Hawaiʻi to evaluate the potential impact on ʻōpeʻapeʻa populations by island or Statewide.

Other factors
Unquantified threats to the ʻōpeʻapeʻa include the incidental introduction and establishment of nonnative and invasive species that have likely reduced bat roosting habitat, foraging habitat, and/or prey availability (USFWS 2011). Resort or housing developments, farming, road construction, and pesticides are expected to persist into the future and have the potential to result in further habitat loss or alteration. Wildfires can cause direct loss of adult bats and dependent young that are unable to escape a forest fire. Historically, conversion of native forests to large-scale agriculture or the expansion of human development has resulted in an appreciable reduction in ʻōpeʻapeʻa roosting, potential foraging habitat, and possible changes in insect prey populations (USFWS 1998). An estimated 600,000 hectares (1.475 million acres) of forest currently occurs across the major Hawaiian Islands (Reeves and Amidon 2018), although portions of the forest have been degraded or fragmented over time. The high mobility of the bat provides capability to utilize fragmented landscapes and the bats have been shown to use areas of low development (Johnston *et al.* 2019).

Synthesis:
Since the 5-year review in 2011, significant new information on the genetics, seasonal movements, foraging and diet, and distribution of the ʻōpeʻapeʻa has been collected. Ongoing research to develop and refine reliable detection tools, management and conservation actions, and bat deterrents to reduce the threats posed by wind energy turbines continues. There is no population estimate and gaps remain in our understanding of the species' abundance, life history parameters, limiting factors, and overall population trend.

Recent genetic studies indicate geographic variation in genetic structure across the State of Hawaiʻi. This may potentially reflect divergence of ʻōpeʻapeʻa populations among the various islands, or the presence of multiple clades with partially overlapping island distributions and some hybridization among the groups. No accepted consensus on taxonomic classification at species or subspecies level is yet apparent; however, evidence of hybridization among groups suggests species divergence is not completely established. The presence of multiple alleles at several of the loci examined in the genetic analyses suggest genetic diversity is present, at least at the loci evaluated. We have no clear basis for reclassifying ʻōpeʻapeʻa currently due to genetics. Should taxonomic reclassification of the ʻōpeʻapeʻa be advised in the future, each taxon would require an evaluation to determine its respective species status, biological characteristics, range and distribution, and appropriate management or recovery actions. Such evaluation might ultimately result in reassessment of the ESA listing status of one or more taxa, which would involve analysis of the five listing factors for each taxon in accordance with section 4(a)(1) of the ESA, formal publication of proposed and final listing rules in the Federal Register, and opportunity for public comment. Until then, recovery actions should focus on protection and conservation of the ʻōpeʻapeʻa Statewide while recognizing the need to maintain the genetic diversity that each island's population represents.

The ʻōpeʻapeʻa was listed as endangered in 1970 based on apparent habitat loss, but with limited knowledge of its life history requirements, and without population estimates. Substantial monitoring efforts are underway to better understand the distribution and population trends of the ʻōpeʻapeʻa on several of the major islands, namely Oʻahu, Maui, and Hawaiʻi. Although the population size remains unknown, the ʻōpeʻapeʻa appears to range widely across the islands. Being a generalist, the ʻōpeʻapeʻa feeds on a variety of insects and may move seasonally or daily in search of resources. They roost in a wide variety of native and nonnative trees, and have been documented in urban, semi-urban, and agricultural areas, in addition to native and nonnative forests. Due to this flexibility in habitat use, roosting habitat is not believed to be a significant limiting factor for the species.

Aside from roosting needs, there is limited understanding of the ecological needs of the species and whether it differs by island or season. On the island of Hawaiʻi, at least some individuals make daily movements above tree line to feed on moths in high-elevation caves, a habitat not available on other islands. Other observations indicate that ʻōpeʻapeʻa use discrete core use areas within a larger foraging range, but these areas may shift seasonally or even nightly depending on local climatic and weather conditions. Overall, the current information points to a species that is well adapted to a range of environments, possesses mobility to use fragmented landscapes, and is resilient to small-scale changes in habitat condition and available resources.

Breeding populations of ʻōpeʻapeʻa are known from the islands of Kauaʻi, Oʻahu, and Hawaiʻi. Breeding populations likely occur across Maui Nui where subadults are frequently documented and conditions that support breeding are present. Gorresen *et al.* (2015) found stable to slightly increasing occupancy based on the breeding season during a 5-year study on the island of Hawaiʻi. Interisland movement is thought to be low, with the possible exception of movement between Maui or Lānaʻi and the island of Kahoʻolawe, where bats were recently observed. Based on typical litter sizes for the hoary bat *Lasiurus cinereus*, ʻōpeʻapeʻa are likely to usually bear twins, as also evidenced by observations of two pups, and of a mother and her newly volant young flying together. Little additional new information exists for other ʻōpeʻapeʻa demographic characteristics such as longevity, fecundity, and survival rate.

Intensive monitoring has shown that nighttime operation of wind energy facilities in Hawaiʻi has resulted in a greater number of ʻōpeʻapeʻa fatalities than originally anticipated when commercial wind energy turbines began operating in Hawaiʻi. Because interisland movement of the ʻōpeʻapeʻa is considered to be low, localized impacts to the population may be expected to be greater on islands with wind energy facilities operating at night. Because of the protected status of the bat, fatalities associated with the operation of a wind energy facility are required to be minimized and mitigated to offset the loss and not jeopardize the existence of the species. Mitigation actions are carried out on the island where the fatalities occur in an effort to sustain the island's representative population. Effectiveness of compensatory mitigation remains uncertain and requires continued research, monitoring, feedback, and adaptive management to ensure the mitigation meets the success criteria and the needs of the bat. ʻŌpeʻapeʻa that are resident on islands without wind energy facilities are not believed to be at direct risk by wind energy due to limited interisland movement.

Several other threats to ʻōpeʻapeʻa are largely unquantified. Barbed wire-associated bat fatalities have been documented but unlike wind energy turbines, most barbed wire fences are not monitored, or at best are monitored infrequently. The impacts of pesticides and rodenticides, historically or currently, are not known at this time. These possible threats would impact the ʻōpeʻapeʻa Statewide. With the changes in agriculture and agricultural pesticide regulations, it may be expected that this threat is decreasing. However, trace amounts of rodenticide were found in 2 out of 21 bat carcasses. Tree trimming and timber harvest of trees above 4.6 meters (15 feet) in height poses a threat to non-volant, dependent bat pups if present. Silviculture and biomass harvest operations exist primarily on the islands of Kauaʻi and Hawaiʻi. The Service recommends not trimming or cutting trees above 4.6 meters (15 feet) between June 1 and September 15 to avoid impacts to dependent (non-volant) bat pups. Because ʻōpeʻapeʻa roost in a wide variety of trees (native and nonnative) and are broadly distributed across all islands, limited removal of trees outside of the pupping season is not currently anticipated to result in adverse effects to ʻōpeʻapeʻa populations. However, removal of a functioning habitat that has taken years to develop, would be expected to have impacts on the activity, prey base, and territoriality of bats.

Overall, ʻōpeʻapeʻa have a much wider distribution than was thought at the time of listing, are highly mobile, capable of using fragmented habitats, and appear adapted to a range of environments and variable habitat and resource conditions. The species moves widely both nightly and seasonally (at least on some islands), which obscures our ability to determine population trends given current sampling technology. The ʻōpeʻapeʻa breed on at least five of the islands and possibly more. There is uncertainty with regard to what factors limit the ʻōpeʻapeʻa and the archipelago's carrying capacity. Rotating turbine blades at night pose a known fatality risk to bats if they fly through the rotor sweep zone. Efforts are made to track, minimize, and mitigate for the fatalities associated with the nighttime operation of wind energy turbines. In addition, currently available genetic data indicates geographic variation in genetic structure but does not clearly support taxonomic reclassification of ʻōpeʻapeʻa.

Given these considerations, we believe that the taxon warrants continued protection under the ESA in light of genetic, population, and new threat uncertainties. Under the ESA [§3(6)], an endangered species is one which is in danger of extinction throughout all or a significant portion of its range. A threatened species is defined as any species that is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. The ‘ōpe‘ape‘a appears to possess resilience, redundancy, and representation across the islands such that it is not on the brink of extinction. Therefore, we conclude that the ‘ōpe‘ape‘a appropriately meets the definition of threatened under the ESA.

Recommendations for Future Actions:

New and Ongoing Research

The Recovery Plan for the Hawaiian Hoary Bat identifies determining the actual population status and habitat requirements of the ‘ōpe‘ape‘a as an interim goal (USFWS 1998). As such, significant research is underway to determine life history traits, ecological requirements, distribution, population trend, and taxonomic structure of the ‘ōpe‘ape‘a.

An initial step in examining occupancy and population trends was completing a power analysis to determine the approximate annual sample size of sites required to detect ‘ōpe‘ape‘a occupancy trends of various magnitudes (WEST 2015). A pilot data set from a 5-year study of ‘ōpe‘ape‘a in Hawai‘i provided the basis for the power simulation (Gorresen *et al*. 2013). The simulations indicated that the annual sample size of sites is more important than the number of within-year revisits to a site for improving the ability to detect trends. This analysis will assist in the development of projects that can monitor the population trends of the ‘ōpe‘ape‘a over the long-term.

A large, multi-year monitoring study deployed acoustic detectors in 2018 to determine the distribution of ‘ōpe‘ape‘a across the landscape of O‘ahu (WEST 2016). Over the first year of acoustic detector deployment, about 61 percent of the echolocation devices resulted in bat detection. This study is demonstrating that bats are utilizing a wide landscape of O‘ahu and that some areas have higher detection rates than others (Starcevich *et al*. 2019). The study is expected to expand our knowledge of the distribution and habitat use of O‘ahu by ‘ōpe‘ape‘a and provide insight into occupancy trends as more years of data are collected.

Gorresen *et al.* (2018a, 2018b) recently showed the importance of multi-state occupancy modeling to improve assessments of habitat use and site quality. Thermal videography provided more accurate estimates of the prevalence of bat flight activity and feeding events than did acoustic detectors. These findings may inform the way inference is made about species-resource relationships, habitat quality and the extent to which species intensively use areas for activities such as foraging (Gorresen *et al.* 2018b).

Genetic research on the ‘ōpe‘ape‘a is ongoing, including sexing of bat carcasses and evaluating genetic variability, intra-island divergence, genetically distinct populations, effective population size, and recent evolutionary bottleneck events (*e.g*., USGS-PIERC 2016b). Recent genetic studies (Pinzari *et al*. 2020) are providing additional insight into the work published by Russell *et al*. (2015), Baird *et al*. (2015, 2017), and Pinzari (2019), and future analyses should continue to inform assessment of taxonomic status and geographical differentiation of ‘ōpe‘ape‘a. The results of the DNA-based testing for sexing of ‘ōpe‘ape‘a fatalities currently informs ITP-associated take estimations for assessing loss for dependent young for females taken between April 1 and September 15.

Additional ecological field projects seeking to expand the knowledge base on the life history of this species are taking place primarily on the islands of Maui and Hawai‘i. Research as compensatory mitigation for wind facility impacts is being conducted in the Pu‘u Makua restoration area of Maui to examine seasonality, prey base, diet analysis, and occupancy over time as restoration proceeds in the area (Pinzari *et al.* 2019, USGS-PIERC 2017). Another project on Maui is examining ‘ōpe‘ape‘a home ranges, seasonal movements, habitat utilization, diet, and prey availability (H.T. Harvey & Associates 2016, Johnston *et al.* 2019). This project is using acoustic monitoring and habitat associations, insect collection within the habitat types and barcoding to determine taxa, radio-telemetry studies of 16 to 20 bats, and analysis of habitat occupancy across a variety of habitat and elevations. Preliminary results indicate ‘ōpe‘ape‘a home range averages about 1,200 hectares (2,967 acres) and can range from 1,200 to 26,000 hectares (3,000 to 64,000 acres) (Johnston *et al*. 2019). Of the nine habitats being evaluated, grasslands, gulches, and low-density developed areas have the highest concentration of detections. The features shared by these three habitats are openness, allowing for unobstructed prey detection, and warmer temperatures, which are believed to be conducive to insect flight (Johnston *et al*. 2019).

Similar comprehensive studies on the movements, roosting behavior, and diet of the ‘ōpe‘ape‘a are being conducted on the island of Hawai‘i (USGS-PIERC 2016a; Montoya-Aiona *et al*. 2019). Assessments of insect prey and acoustic surveys for habitat use and foraging patterns have been completed in dry habitat on the leeward western shore of the island (Montoya-Aiona *et al*. 2019). Further objectives include radio-tagging and collecting data from up to 48 bats per year to look at seasonal and annual home range and movement patterns on the eastern slope of the island, conducting a fecal analysis with molecular barcoding for diet composition and food availability, identifying habitats used for foraging, roosting, and breeding, and mother-pup demographics and predation at maternity roosts (USGS-PIERC 2016a). This study has the potential to verify and refine previous movement studies, while also collecting key life-history data where significant data gaps currently exist.

Studies using acoustic monitoring in combination with thermal videography to monitor bat interactions with wind turbines are needed. These bat monitoring methods are being used to assess the behavior of bats at turbines with higher cut-in speeds. In addition, these monitoring methods are being deployed to evaluate the behavior of bats at wind turbines where ultrasonic bat deterrents have been installed for the first time in Hawai‘i.

Additional future actions needed

- Assess current research on the genetic structure and taxonomic status of ōpeʻapeʻa throughout the State of Hawaiʻi and if appropriate, revise taxonomic classification or designate distinct population segments.

- Quantify biological characteristics of the ʻōpeʻapeʻa, including pup and adult survival rates, longevity, mating, breeding, fecundity, and heterothermy (variability in body-temperature self-regulation).

- Implement and refine bat deterrent technology to minimize wind fatalities.

- Develop and refine improved detection technologies for the ʻōpeʻapeʻa.

- Determine the population and trend of ʻōpeʻapeʻa; should the taxonomic classification of the ʻōpeʻapeʻa be revised in the future, each taxon would require an evaluation to determine its respective population and trend.

- Conduct standardized surveys on Kauaʻi, Lānaʻi, Molokaʻi, and Kahoʻolawe to determine the distribution of ʻōpeʻapeʻa and presence or absence of established breeding populations based on officially recognized taxonomic and genetic conclusions.

- Determine ʻōpeʻapeʻa patterns of interisland and intraisland movement.

- Determine habitat and dietary needs and revise management actions to benefit ʻōpeʻapeʻa recovery.

- Preserve and manage existing and suitable future ʻōpeʻapeʻa habitat. Forests at all elevations may need protection and management to provide a year-round prey base. Low-elevation forests are under the greatest threat due to development, agriculture, and high numbers of habitat-altering invasive plant and invertebrate species.

- Determine the significance of barbed wire mortality to ʻōpeʻapeʻa populations and identify any risk factors that increase a fence's likelihood to snag bats (location, slope, surrounding habitat type, number of strands of barbed wire, distance from certain habitat features, etc.)

- Collect additional information on the threat from pesticides and/or rodenticides on the ʻōpeʻapeʻa, to help elucidate the threat that these may pose, directly through chemical exposure or indirectly through prey reduction, to the bat populations in the islands.

- Collect additional information on other limiting factors for this species, such as predation from nonnative mammals (i.e., cats or rats) or avian species (i.e., barn owls) and competition from introduced amphibians (coqui frogs) or ants (little fire ants).

- Assess the impacts of climate change on ‘ōpe‘ape‘a diet availability and thermal requirements necessary for biological success.

- Develop alliances and expand partnerships throughout all islands of the State to avoid and minimize fatality risks and maximize benefit to the ‘ōpe‘ape‘a.

Table 1. Status and trends of *Lasiurus cinereus semotus* from listing through current 5-year status review.

Date	No. Wild Individuals	Interim* Downlisting and Delisting Criteria Identified in Recovery Plan	Downlisting or Delisting Criteria Completed?
1970 (listing)	No population given	No recovery plan developed	N/A
1998 (Recovery plan)	Population size unknown. ʻōpeʻapeʻa historically inhabited the islands of Hawaiʻi, Maui, Oʻahu, Kauaʻi, and Molokaʻi, but may be resident on only Hawaiʻi, Kauaʻi, and Maui; no verified records for Lānaʻi and Kahoʻolawe	Interim downlisting criterion: A widespread population of ʻōpeʻapeʻa must be naturally reproducing, and stable or increasing in size on the island of Hawaiʻi for a minimum of 5 consecutive years before downlisting is considered.	No
		Interim delisting criterion: ʻōpeʻapeʻa populations on Hawaiʻi, Kauaʻi, and Maui must be well distributed, naturally reproducing, and stable or increasing in size for at least 5 consecutive years following downlisting before delisting is considered.	Partial
2011 (5-year status review)	ʻōpeʻapeʻa is widely distributed on island of Hawaiʻi and is naturally reproducing. However, it is not yet clear whether the ʻōpeʻapeʻa population is stable or increasing in size. At this time, there is insufficient data from Maui or Kauaʻi on ʻōpeʻapeʻa distribution, breeding, or population trend.	Interim downlisting criterion: A widespread population of ʻōpeʻapeʻa must be naturally reproducing, and stable or increasing in size on the island of Hawaiʻi for a minimum of 5 consecutive years before downlisting is considered.	No
		Interim delisting criterion: ʻōpeʻapeʻa populations on Hawaiʻi, Kauaʻi, and Maui must be well distributed, naturally reproducing, and stable or increasing in size for at least 5 consecutive years following downlisting before delisting is considered.	No

Date	No. Wild Individuals	Interim* Downlisting and Delisting Criteria Identified in Recovery Plan	Downlisting or Delisting Criteria Completed?
2020 (5-year status review)	Population size unknown. Breeding known on islands of Kauaʻi, Oʻahu, and Hawaiʻi, and likely on Maui and Molokaʻi. Known to be present on Lānaʻi and Molokaʻi and known to make nighttime visits to Kahoʻolawe. Study conducted from 2007 to 2011 suggests population on island of Hawaiʻi is stable to increasing (Gorresen *et al*. 2013). Trends not established for Maui or Kauaʻi.	Interim downlisting criterion: A widespread population of ʻōpeʻapeʻa must be naturally reproducing, and stable or increasing in size on the island of Hawaiʻi for a minimum of 5 consecutive years before downlisting is considered.	Yes, ʻōpeʻapeʻa on the island of Hawaiʻi are known to be widespread, reproducing, and occupancy believed to be stable to increasing
		Interim delisting criterion: ʻōpeʻapeʻa populations on Hawaiʻi, Kauaʻi, and Maui must be well distributed, naturally reproducing, and stable or increasing in size for at least 5 consecutive years following downlisting before delisting is considered.	Partial; ʻōpeʻapeʻa are known to be well-distributed on these three islands, but we have no recent population trend information for Maui or Kauaʻi islands

* Because there was limited knowledge of the life history of this subspecies with respect to short-term and long-term survival, only tentative criteria for downlisting and delisting were established in the recovery plan. Research addressing these questions must be undertaken prior to consideration of delisting and is ongoing.

Table 2. Threats to ʻōpeʻapeʻa (*Lasiurus cinereus semotus*) and ongoing conservation efforts.

Threat	Listing factor	Current Status	Conservation/ Management Efforts
Ungulate degradation of habitat	A	Unknown	Partial. Some areas known to be used by ʻōpeʻapeʻa for foraging and roosting have been fenced and ungulates removed. Monitoring is conducted to maintain zero ungulate presence at these sites. Fencing of additional areas Statewide are planned.
Established ecosystem altering invasive plant species degradation of habitat	A	Ongoing	Partial. Protection of habitat and restoration projects are ongoing on the islands of Oʻahu, Maui, and Hawaiʻi specifically for the ʻōpeʻapeʻa.
Climate change	A, E	Increasing	Partial. Climate change effects on ʻōpeʻapeʻa may include expansion of habitat for invasive competitors or changes in food availability needed during different seasons.
Predation	C	Ongoing	No. Little is known about the severity of threats from predation by introduced rat, barn owls, the native pueo, or cats.
Disease	C	Potential	White-nose syndrome not known from Hawaii; limited monitoring has been done to screen for occurrence.
Lack of adequate regulations	D	Ongoing	No. While Hawaiʻi does have regulations in place to protect this species, the current biosecurity program is inadequate to prevent the introduction of invasive species, such as coqui frog (Beard 2007, Bernard and Mautz 2016) that affect ʻōpeʻapeʻa habitat and food resources.
Alien Competitors	D, E	Ongoing	No. ʻŌpeʻapeʻa face resource competition from nonnative coqui frogs, which feed on insects also consumed by the bat. Existing regulations offer inadequate protection to the ʻōpeʻapeʻa from the introduction of nonnative competitors and the loss of their food resources.
Human Disturbance of roosts during pupping	E	Ongoing	Partial. Education, guidance, and technical assistance are provided to parties that contact the Service about the removal of trees over 4.6 meters (15 feet) in height during the pupping season from June 1 through September 15. Additionally, protection of forests through purchase or mitigation easements reduces this threat.
Environmental Catastrophes	E	Ongoing	No. Risk of catastrophic events such as storms is ongoing. Management to prevent such events is generally not feasible, although some effects can be mitigated.

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U.S. FISH AND WILDLIFE SERVICE
SIGNATURE PAGE for 5-YEAR STATUS REVIEW of
ʻōpeʻapeʻa or Hawaiian hoary bat (*Lasiurus cinereus semotus*)

Pre-1996 DPS listing still considered a listable entity? __N/A______________

Recommendation resulting from the 5-year status review:

____________ Delisting
______X_____Reclassify from Endangered to Threatened status
____________ Reclassify from Threatened to Endangered status
____________ No Change in listing status

PACIFIC ISLANDS FISH AND WILDIFE OFFICE APPROVAL:

MARY ABRAMS
Digitally signed by MARY ABRAMS
Date: 2021.03.09 16:37:32 -07'00'
__________________________________ Date __________________
Field Supervisor

REGIONAL OFFICE APPROVAL:

Digitally signed by MARJORIE NELSON
Date: 2021.03.16 13:54:58 -06'00'
__________________________________ Date 3/16/21
Acting Assistant Regional Director, Ecological Services