DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

50 CFR Part 17

[FWS–R1–JA–2008–007; 96100–1671–000; 1018–AT62]

Endangered and Threatened Wildlife and Plants; Final Rule To List Six Foreign Birds as Endangered

AGENCY: Fish and Wildlife Service, Interior.

ACTION: Final rule.

SUMMARY: We, the U.S. Fish and Wildlife Service (Service), determine endangered status for six avian species—black stilt (*Himantopus novaezelandiae*), caerulean paradise-flycatcher (*Eutrichomyias rowleyi*), giant ibis (*Pseudibis gigantea*), Gurney's pitta (*Pitta gurneyi*), long-legged thicketbird (*Trichocichla rufa*), and Socorro mockingbird (*Mimus graysoni*)—under the Endangered Species Act of 1973, as amended (Act). This rule implements the protection of the Act for these six species.

EFFECTIVE DATE: This final rule is effective February 15, 2008.

ADDRESSES: The supporting file for this rule is available for public inspection, by appointment, during normal business hours, Monday through Friday, in Suite 110, 4401 N. Fairfax Drive, Arlington, Virginia 22203.

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SUPPLEMENTARY INFORMATION:

Background

In this final rule, we determine endangered status for six foreign bird species under the Act (16 U.S.C. 1531 *et seq.*): Black stilt (*Himantopus novaezelandiae*), caerulean paradise-flycatcher (*Eutrichomyias rowleyi*), giant ibis (*Pseudibis gigantea*), Gurney's pitta (*Pitta gurneyi*), long-legged thicketbird (*Trichocichla rufa*), and Socorro mockingbird (*Mimus graysoni*).

Previous Federal Action

Section 4(b)(3)(A) of the Act requires us to make a finding (known as a ''90-day finding'') on whether a petition to add, remove, or reclassify a species from the list of endangered or threatened species has presented substantial information indicating that the requested action may be warranted. To the maximum extent practicable, the finding shall be made within 90 days following receipt of the petition and published promptly in the **Federal Register**. If we find that the petition has presented substantial information indicating that the requested action may be warranted (a positive finding), section 4(b)(3)(A) of the Act requires us to commence a status review of the species if one has not already been initiated under our internal candidate assessment process. In addition, section 4(b)(3)(B) of the Act requires us to make a finding within 12 months following receipt of the petition on whether the requested action is warranted, not warranted, or warranted but precluded by higher-priority listing actions (this finding is referred to as the ''12-month finding''). Section 4(b)(3)(C) of the Act requires that a finding of warranted but precluded for petitioned species should be treated as having been resubmitted on the date of the warranted but precluded finding, and is therefore subject to a new finding within 1 year and subsequently thereafter until we take action on a proposal to list or withdraw our original finding. The Service publishes an annual notice of resubmitted petition findings (annual notice) for all foreign species for which listings were previously found to be warranted but precluded.

On November 24, 1980, we received a petition (1980 petition) from Dr. Warren B. King, Chairman, United States Section of the International Council for Bird Preservation (ICBP), to add 79 bird species (19 native and 60 foreign) to the List of Endangered and Threatened Wildlife (50 CFR 17.11(h)), including the black stilt and the long-legged thicket bird (or, long-legged warbler, which was the common name used in the petition). In response to the 1980 petition, we published a positive 90-day finding on May 12, 1981 (46 FR 26464), for 77 of the species (19 domestic and 58 foreign), noting that 2 of the foreign species identified in the petition were already listed under the Act, and initiated a status review. On January 20, 1984, we published an annual review on pending petitions and description of progress on all petition findings addressed therein (49 FR 2485). In that notice, we found that listing all 58 foreign bird species from the 1980 petition, including the black stilt and the long-legged thicketbird, was warranted but precluded by higher-priority listing actions. On May 10, 1985, we published the first annual notice (50 FR 19761) in which we continued to find that listing all 58 foreign bird species from the 1980 petition was warranted but precluded. In our next annual notice, published on January 9, 1986 (51 FR 996), we found that listing 54 species from the 1980 petition, including the black stilt and the long-legged thicketbird, continued to be warranted but precluded, whereas new information caused us to find that listing four other species in the 1980 petition was no longer warranted. We published additional annual notices on the species included in the 1980 petition on July 7, 1988 (53 FR 25511); December 29, 1988 (53 FR 52746); April 25, 1990 (55 FR 17475); and November 21, 1991 (56 FR 58664), in which we indicated that the black stilt and the long-legged thicketbird continued to be warranted but precluded.

On May 6, 1991 (1991 petition), we received a petition from Alison Stattersfield, of ICBP, to list 53 additional foreign birds under the Act. The caerulean paradise-flycatcher, giant ibis, Gurney's pitta, and Socorro mockingbird were included in the 1991 petition. On December 16, 1991, we published a positive 90-day finding and announced the initiation of a status review of the 53 foreign birds listed in the 1991 petition (56 FR 65207). The 1991 petition included the giant ibis, Gurney's pitta, Socorro mockingbird, and caerulean paradise-flycatcher among the 53 foreign birds that the petitioner requested be listed under the Act. On March 28, 1994 (59 FR 14496), we published a proposed rule to list 30 African bird species from both the 1980 and 1991 petitions. In the same **Federal Register** document, we included a notice of findings in which we announced our determination that listing the 38 remaining species from the 1991 petition was warranted but precluded; this group included the giant ibis, Gurney's pitta, Socorro mockingbird, and caerulean paradise-flycatcher. On May 21, 2004 (69 FR 29354), we published an annual notice of findings on resubmitted petitions for foreign species and annual description of progress on listing actions (2004 annual notice) within which we ranked species for listing by assigning them a Listing Priority Number per the Service's listing priority guidelines, published on September 21, 1983 (48 FR 43098). Based on this ranking and priorities, we determined that listing five of the previously petitioned species—the black stilt, caerulean paradise-flycatcher, giant ibis, Gurney's pitta, and Socorro mockingbird—was warranted. In the same 2004 annual notice, we determined that the long-legged thicketbird and 16 other species no longer warranted listing on the basis that those species were likely extinct. In response to the 2004 annual notice, we received information indicating that the long-legged thicketbird had been rediscovered, in small numbers, in 2002. The magnitude of the threat to the species was perceived as high and the immediacy of threat imminent. Therefore, we assigned this species a listing priority ranking of 1, which ranking is reserved specifically for a monospecific genus, and determined that listing the species was warranted at that time.

On November 22, 2006 (71 FR 67530), we published a **Federal Register** notice to list black stilt, caerulean paradise-flycatcher, giant ibis, Gurney's pitta, long-legged thicketbird, and Socorro mockingbird as endangered. We implemented the Service's peer review process and opened a 60-day comment period to solicit scientific and commercial information on the species from all interested parties following publication of the proposed rule.

Summary of Comments and Recommendations

In the proposed rule of November 22, 2006 (71 FR 67530), we requested that all interested parties submit information that might contribute to development of a final rule. We received five comments: two from members of the public and one each from the governments of Cambodia, Fiji, and Mexico. In accordance with our policy, ''Notice of Interagency Cooperative Policy for Peer Review in Endangered Species Act Activities,'' published on July 1, 1994 (59 FR 34270), we also sought the expert opinion of at least three appropriate independent specialists regarding the proposed rule.

Comment 1: Four commenters supported the proposed listings, including the governments of Cambodia, Fiji, and Mexico. The government of Cambodia ''strongly endorsed[d] the proposal of giant ibis to be listed in [the] U.S. Endangered Species Act. The Fijian government noted that the benefits of listing the long-legged thicketbird under the Act are ''perhaps marginal'' but that a listing could help where species, such as the thicketbird, are not listed in the Appendices of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) because trade in the wild bird is not a concern at this time. The potential funding and technical support (see Available Conservation Measures) for the development of management programs for the conservation of species in foreign countries could be beneficial to the thicketbird in Fiji. Similarly, the government of Mexico commented that listing the Socorro mockingbird under the Act would support its ongoing efforts and additional actions to be undertaken by the Mexican government, including scientific investigations, in order to protect the species.

Our Response: While general support of a listing is not, in itself, a substantive comment that we take into consideration as part of our five-factor analysis, we appreciate the support of these range countries. Cooperation is important to the conservation of foreign species.

Comment 2: One researcher opposed the listing of the long-legged thicketbird on the basis that the species is not endangered, but merely elusive to the inexperienced or to those with an uneducated eye.

Our Response: We have taken into account in our review of the long-legged thicketbird the bird's elusive behavior. However, we believe that we have used the best available scientific information in our status review and have accurately determined the appropriate threat status for this species.

Comment 3: One commenter recommended that the term *kakï* be used to refer to the black stilt throughout the rule, as it is the preferred name in New Zealand.

Our Response: We have added this common name in the species description for the black stilt, but have chosen to use the common name ''black stilt'' throughout the rule and in the list because the federal listing will be categorized under the species grouping ''stilt.''

Several commenters provided additional information on the species. This information has been considered and incorporated into the rulemaking as appropriate (as indicated in the citations by ''in litt.'').

Species Information and Factors Affecting the Species

Under section 4(a) of the Act (16 U.S.C. 1533(a)(1)) and regulations promulgated to implement the listing provisions of the Act (50 CFR part 424.11), we may list a species as threatened and endangered on the basis of five threat factors: (A) Present or threatened destruction, modification, or curtailment of its habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) inadequacy of existing regulatory mechanisms; or (E) other natural or manmade factors affecting its continued existence. Listing may be warranted based on any of the above threat factors, either singly or in combination.

Under the Act, we may determine a species to be endangered or threatened. An endangered species is defined as a species which is in danger of extinction throughout all or a significant portion of its range. A threatened species is defined as a species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. Therefore, we evaluated the best available scientific and commercial information on each species under the five listing factors to determine whether they met the definition of endangered or threatened.

Following is a species-by-species analysis of these five factors. The species are considered in alphabetical order: Black stilt, caerulean paradise-flycatcher, giant ibis, Gurney's pitta, long-legged thicketbird, and Socorro mockingbird.

I. Black stilt (*Himantopus novaezelandiae*)

Species Description

The black stilt is a wading bird in the family Recurvirostridae. It is native to New Zealand and is locally known there by its Maori name ''*kaki*.'' Adults are characterized by long red legs, a slender bill and black plumage (BirdLife International (BLI) 2007a; New Zealand Conservation Management Group (NZ CMaG 2007). Adult males and females are generally regarded as having identical plumage (BLI 2007e); however, Elkington and Maloney (2000) determined that white flecking around their eyes and crown is generally indicative of older males. Juveniles have a white-plumed breast, neck, and head (BLI 2007e). Black and pied stilt (*Himantopus himantopus*) hybridize (see Taxonomy, below), and hybrids are more varied in color, with varying gradations of white and black plumage, and varying body characteristics, such as shorter legs and longer bills (BLI 2007e; Department of Conservation (DOC) 2007a; Maloney & Murray 2002; Reed *et al.* 2007).

The species can reach 16 inches (in) (40 centimeters (cm)) (BLI 2007e) in height, with a wingspan of 23 in (58 cm). The average age of birds in the current population is 6 years (BLI 2007e; Maloney & Murray 2002). The potential lifespan of the species is unknown, but the oldest recorded specimen, a banded female relocated in 1983, was estimated to be at least 12 years old (Pierce 1986b).

Taxonomy

The black stilt was first taxonomically described by Gould in 1841 and placed in the family Recurvirostridae. It is one of two stilt species in New Zealand, the other being the pied stilt (Pierce 1984a;

Reed *et al.* 1993a). Where their ranges overlap, the black stilt may interbreed with its close relative, the pied stilt (Reed *et al.* 1993a). It is generally accepted that hybridization between these two species has been occurring only in the last two centuries, as the pied stilt expanded its range from Australia to New Zealand in the early 19th century (Greene 1999; Pierce 1984a; Reed *et al.* 1993a). During the late 19th century, the frequency of hybrid sightings increased (Pierce 1984b) but observers of the time did not realize that the two species were hybridizing, and the taxonomy of *Himantopus* species of New Zealand was the subject of much debate (Buller 1874; Potts 1872; Travers 1871). In 1984, Pierce (1984b) concluded on the basis of morphological, ecological, and behavioral differences that the two species remained distinct. Genetic analysis in the 20th century confirmed that the two species were undergoing introgressive hybridization, wherein viable offspring produced from the successful mating of two distinct species were subsequently capable of mating with parental species (Greene 1999). From these studies, despite the genetic similarity between the two species, Greene (1999) concluded that the species remain distinct.

Habitat and Life History

Black stilt habitat includes riverbanks, lakeshores, swamps, and shallow ponds (Maloney & Murray 2002; Pierce 1982; Potts 1872; Reed *et al.* 1993a). The species' habitat preferences shift slightly depending on the seasons, which are: Breeding (braided rivers, side streams, and swamps), post-breeding (riverbeds and shallow tarns), and wintering (inland waters or river deltas) (Maloney & Murray 2002). However, these habitats are often located within the same watershed, and the species is considered a primarily sedentary, nonmigrating species (Maloney & Murray 2002; Pierce 1986b). About 90 percent of the black stilt population overwinters in the Upper Waitaki Basin (UWB; in the central region of the South Island) by moving to inland areas to continue feeding on aquatic insects, including larvae of mayfly (*Deleatidium* sp.) and caddisfly (*Olinga* sp.), and, to a lesser extent, on mollusks and fish (DOC 2007a; Reed *et al.* 1993a). Researchers believe that the black stilt's long legs allow them to wade out into the deeper, unfrozen sections of rivers where they can continue foraging throughout the winter (DOC 2007a; Reed *et al.* 1993a).

A small percentage (about 10 percent) of the population migrates to coastal Canterbury on South Island or Northern Island coastal areas in the winter, from February to June, before returning to the UWB to breed in July and August (BLI 2007e; Maloney & Murray 2002: NZ CMaG 2007; Pierce 1984a; Pierce 1996; Reed *et al.* 1993a). Reed *et al.* (1993a) believe that this migratory behavior has resulted from hybridization with the pied stilt (which migrates to coastal waters in the winter) (Dowding & Moore 2006). In the absence of a suitable mate of the same species, black stilts will mate and produce hybrid offspring with the pied stilt (BLI 2007e; DOC 2007a; Maloney & Murray 2002; Reed *et al.* 1993a). Mixed pairs (a black stilt paired with a pied stilt) and their offspring are more likely to participate in migratory behavior (Dowding & Moore 2006; Reed *et al.* 1993a). Hybridization is discussed further under Factor E.

Black stilts reach adulthood around 18 months of age, attaining sexual maturity between 2 and 3 years of age. They mate for life, nest in solitary pairs (often miles (kilometers) from another pair), and exhibit high nesting fidelity (returning to the same location to nest each year) (BLI 2007e; DOC 2007a; Maloney & Murray 2002; Pierce 1984a; Reed *et al.* 1993a). The breeding season begins in July or August and egg-laying occurs from September to December (BLI 2007e; Maloney & Murray 2002; NZ CMaG 2007). Ground-nesting birds, black stilts prefer open nesting sites, such as dry, stable riverbanks (Maloney & Murray 2002; Pierce 1982; Pierce 1986b; Reed *et al.* 1993a). They lay a typical clutch size of four eggs and have a lengthy fledging period of 40 to 55 days (the amount of time it takes birds to hatch and leave the nest) (Maloney & Murray 2002). Both sexes share the nesting responsibility (Maloney & Murray 2002; Pierce 1986b; Pierce 1996; Sanders & Maloney 2002). Eggs are incubated by both sexes for 25 days, and pairs will often re-nest if the first clutch is lost early in the season (BLI 2007e; Reed *et al.* 1993a; Maloney & Murray 2002; NZ CMaG 2007). Chicks are precocial (the young are relatively mature and mobile from the moment of hatching) and capable of feeding themselves within hours of hatching (DOC 2007a; Reed *et al.* 1993a). After fledging, chicks stay with parents until the beginning of the following breeding season (Maloney & Murray 2002).

The black stilt's breeding success in the wild is very low. For example, according to Maloney and Murray (2002), from 1977 to 1979, of 33 chicks that hatched in unmanaged nests, only 2 individuals (or 6.1 percent) survived to fledge (i.e., lived long enough to leave the nest). Overall breeding success (nesting success plus fledging success) for the same period was 0.9 percent. Recruitment, defined by Maloney and Murray (2002) as the number of chicks attaining 2 years of age, is only about 4 percent.

Reproductive potential does not appear to be the primary limiting factor to the black stilt's breeding success and recruitment rates. The black stilt has high reproductive capability, first reproducing at age 2 and continuing to produce multiple clutches in captivity to at least age 13 plus (Maloney & Murray 2002; Reed 1998). The species has high fecundity, producing clutches of one to four eggs every breeding season, and will re-nest if clutches are lost early in the season (BLI 2007e; Reed *et al.* 1993a; Maloney & Murray 2002). Moreover, a review of captive breeding records from two breeding seasons (1981 to 1982 and 2001 to 2002) found that the survival rate of captive-bred stilts reintroduced to the wild at 2 months and 10 months increased to 88 percent and 82 percent, respectively (Van Heezik *et al.* 2005).

Historical Range and Distribution

When it was described in 1841, the species' range included both the North and South Islands of New Zealand (Pierce 1984a). Its range has contracted twice in the 20th century: Once in the 1940s, when the breeding range became restricted to the South Island, and again in the 1960s, when the UWB became their only breeding area (Maloney & Murray 2002; Pierce 1984a; Reed *et al.* 1993a).

As the black stilt's range contracted, researchers noticed that the pied stilt's range had increased (Pierce 1984a). In the last quarter of the 19th century, both black and pied stilts were considered common across South Island (Buller 1874, 1878; Travers 1871). By the 1980–1981 breeding season, the estimated number of pied stilts in the UWB was between 1,500 and 2,000 (Pierce 1984a). At the same time, only 23 black stilt adults were known in the wild (Maloney & Murray 2002; Van Heezik *et al.* 2005). Experts considered whether the black stilts were being competitively excluded by the pied stilt and found that this was not the case. Black stilts and pied stilts prefer slightly different feeding areas (black stilts forage in riffles and pied stilts at pools) (Pierce 1986a); black stilts are better foragers than pied stilts (employing a greater variety of foraging techniques that allow them to obtain more food) (DOC 2007a; Pierce 1986a; Reed *et al.* 1993a); also, black stilts are territorially dominant over pied stilts when breeding areas overlap (Maloney & Murray 2002). From this work, researchers concluded that the decreasing range and numbers of black stilts in the face of the increasing pied stilt population reflected the black stilt's inability to adapt as readily to man-induced changes, namely, the introduction of predators and habitat modification (Pierce 1986a, 1986b; Maloney & Murray 2002: Reed *et al.* 1993a). Historical declines were attributed primarily to predation by mammals introduced in the 19th century and secondarily to habitat loss and hybridization with the pied stilt (Pierce 1984b; Reed *et al.* 1993a, 1993b).

For a primarily sedentary species, the black stilt requires a fairly large area for feeding and nesting. In counts conducted between 1991 and 1994, Maloney (1999) found less than one black stilt for every 3 mi (5 km) of river surveyed. The species' tendency to overwinter inland requires sufficiently large areas of river habitat to allow for continuous year-round feeding (DOC 2007a; Reed *et al.* 1993a). Life history traits, such as lifelong pair-bonding combined with high nesting fidelity (returning to the same location to nest each year) and solitary nesting combined with their preference for open nesting sites (often miles from another pair), contribute to the highly dispersed nature of the population and their resultant large habitat requirement (Maloney & Murray 2002; Pierce 1982, 1986b; Reed *et al.* 1993a).

Current Range and Distribution

The current range of the black stilt is estimated to be an 821 square mile (mi^2) (2,830 square kilometer (km^2)) area in the ''braided-river'' habitat of the UWB (BLI 2007e). Located on the eastern side of the Southern Alps, in central South Island, New Zealand, the following rivers and lakes comprise the braided river habitat: Tasman, Godley, Hopkins, Ahuriri, Tekapo, Cass, Dobson, Macaulay, Lower Ohau, Pukaki and Upper Ohau, as well as Lakes Ohau and Pukaki (Maloney *et al.* 1997). The UWB population is sometimes referred to in the literature as the Mackenzie Basin population (for example, in Reed *et al.* 1993a). According to Dr. Richard Maloney of the Department of Conservation, Twizel, New Zealand (in litt. November 2007), although the two areas represent slightly different geographical boundaries, the black stilt population being referred to is the same in either instance. Because habitat quality in the species' present range is considered to be higher than in other former localities, the species is managed in situ (Maloney & Murray 2002).

The black stilt is considered locally extinct in 9 of the 13 Department of Conservation Conservancy Districts, occurring only in 2 districts (Canterbury and Otaga) on the South Island and 2 (Waikata and Bay of Plenty) on the North Island (Hitchmough 2002). The majority of the population remains in the UWB, on the South Island, year round (BLI 2007e; Maloney & Murray 2002: Pierce 1984a; Reed *et al.* 1993a; NZ CMaG 2007), and their breeding range is now entirely confined to the wetlands and rivers of the UWB (Maloney & Murray 2002; Pierce 1984a).

Population Estimates

The wild black stilt population has undergone severe reductions in numbers concomitant with the reduction in range area. In the 1950s, the total population was estimated at 500 to 1,000 birds; however, within one decade the population decreased to between 50 to 100 birds (Pierce 1996).

Since 1981, the New Zealand Department of Conservation has intensively managed the wild black stilt population, including the establishment of a captive population (Maloney & Murray 2002; Reed 1998; Reed *et al.* 1993a, 1993b). The captive breeding program entails the transfer of ''eggs, chicks, juveniles and sub-adults from one part of the range to any other part of the range'' (R. Maloney in litt. October 2007). For further discussion on the captive breeding program, see ''Management Plans,'' under Factor D.

Since the establishment of the captive breeding program, the Department of Conservation has managed the global population of black stilts, including captive-held and wild birds, as a single breeding population (R. Maloney in litt. November 2007). Wild and reintroduced birds are free to move across the full geographical range of the species. Thus, the number of adults in the wild should be considered in conjunction with the number of breeding pairs held in captivity. According to Dr. Maloney (in litt. October 2007), a total wild population number, including immature individuals, ''is not informative'' because the total wild population is dependent on how many young the breeding program produces and releases each year. The number of breeding pairs is more informative as an indicator of the status of the population (R. Maloney in litt. November 2007). The number of available females is particularly important because of the species' tendency to hybridize with pied stilt when male black stilts are unable to find suitable mates (see Factor E) (Maloney & Murray 2002).

Wild population estimates: From 1975 to 1979, there were an estimated 50 to 60 adults in the wild (Pierce 1984a); by 1981, only 23 adults remained in the wild (Maloney & Murray 2002; Van Heezik *et al.* 2005). In August 2000, there were 48 adults in the wild, of which 15 to 18 were females. As of February 2007, the wild adult population consisted of 87 adults, including 17 productive pairs and a total of 41 females (DOC 2007b).

Captive-held population numbers: Throughout the 1980s, an average of 15 birds was managed in captivity (Reed *et al.* 1993a). In 1998, the number of managed birds reached 48 individuals. At that time, it was decided that the captive-held population should be maintained at approximately 6 breeding pairs. It was further determined that, in order to maintain a genetic diversity among the breeding stock, a base population of at least 18 breeding adults and juveniles would be maintained as replacement stock and, barring a catastrophic loss of the wild population, only first-generation captive stock would be used for breeding (Reed 1998). As of 2007, the captive breeding program consisted of 15 adults, including 6 productive pairs (DOC 2007b).

The black stilt is considered to be one of the rarest wading birds in the world (BLI 2007e; Caruso 2006; Reed et al. 1993a). Since 1994, the species has been categorized by the World Conservation Union (IUCN) as ''Critically Endangered'' (BLI 2007a). The species' continued existence in the wild today is considered a direct result of the captive breeding program (Maloney & Murray 2002; Reed *et al.* 1993a; Van Heezik *et al.* 2005). According to the priority management ranking system devised by Molloy and Davis (1992) for the New Zealand Department of Conservation, the species was ranked as a Category ''A'' species, which includes the ''highest priority threatened species'' (Hitchmough et al. 2005; Reed *et al.* 1993a). Under New Zealand Department of Conservation's management system devised in 2002, the black stilt is classified as ''Nationally Critical'' (Hitchmough *et al.* 2005). In the 2004 to 2005 breeding season, 7 pairs of captive-held black stilt and 12 pairs in the wild produced ''up to 100 birds per year for release into the wild'' (NZ CMaG 2007).

Summary of Factors Affecting the Black Stilt

A. The Present or Threatened Destruction, Modification, or Curtailment of the Black Stilt's Habitat or Range

Today, it is estimated that only 10 percent of New Zealand's wetlands remain intact (Caruso 2006). The braided river habitat of UWB is a globally rare ecosystem. With an estimated area of 3,664 mi^2 (9,490 km^2), the UWB may account for 50 to 60 percent of the remaining suitable braided river habitat in New Zealand (Caruso 2006; Maloney et al. 1997). The UWB is the only breeding ground for the black stilt and most of the population remains in the UWB year-round (Maloney & Murray 2002; Pierce 1984a; Reed *et al.* 1993a).

Several factors affect the quality of black stilt breeding and nesting grounds. Among the most significant impacts to the UWB has been the diversion of rivers for hydroelectric power (HEP) development (Caruso 2006; Collar *et al.* 1994a; Maloney 1999). Since 1935, eight HEP plants have been built on rivers, floodplains, and wetlands associated with the UWB (Caruso 2006). The damming of rivers for HEP and flood control projects has reduced river flows and interrupted the natural flooding cycles vital to the creation and maintenance of the open gravel braided river system of the UWB. It is estimated that floodplains have been reduced by 17 percent in the 11 major rivers of the UWB (Caruso 2006; Maloney & Murray 2002).

Disturbance by recreational users of riverbeds and riversides also affects black stilt habitat within the UWB (Maloney & Murray 2002). The riverine habitat where black stilts live and nest is a prime outdoor recreation area. According to the New Zealand Ministry for the environment (NZ MFE 2007), recreational activities include water sport fishing, mountain biking, four-wheel driving, and jet skiing. Central South Island Fish and Game New Zealand manages the Waitaki Catchment (which includes rivers of the UWB and associated wetlands) and considers the Catchment to be ‘‘outstanding publicly accessible game bird hunting and waterfowl habitat’’ (NZ MFE 2007). According to the New Zealand Ministry for the Environment (NZ MFE 2007), recreational use and impacts on the areas of the Waitaki Catchment are predicted to increase. The New Zealand Ministry for the Environment (2007) does not address the effect that increased recreational activities will have on the black stilt or other native species (See also Factor D). Maloney and Murray (2002) indicate that the species does not tolerate human disturbance. Recreational activities that are disruptive to the black stilt's life cycle are considered to be a potentially serious threat to the species (R. Maloney in litt. February 2007). Indiscriminate use of off-road vehicles and jet-boats, disturbance by hikers and dogs, and fishing and camping activities are disruptive to black stilts (Maloney & Murray 2002). Recreational use of riverbed sites disturbs nesting birds and prevents successful rearing of offspring (BLI 2007e).

Additional impacts on black stilt habitat include drainage for fields or irrigation, overgrazing of wetlands, and water extraction for agricultural irrigation (Caruso 2006; Collar *et al.* 1994a; Maloney & Murray 2002). Since 1850, 40 percent of UWB wetlands have been drained for farming (Caruso 2006). Proliferation of introduced weeds is a problem (Maloney & Murray 2002). Invasive plants, especially the crack willow (*Salix fragilis*), introduced by settlers as windbreaks, degrade black stilt habitat by contributing to an overgrowth in formerly open areas (Caruso 2006; Collar *et al.* 1994a; Maloney & Murray 2002: Pierce 1996; Reed *et al.* 1993).

Summary of Factor A

The black stilt's primary habitat and only known nesting ground within the UWB is a globally rare ecosystem that is being altered by water diversion, wetland conversion, invasive species, and recreation. Lack of suitable habitat for feeding and nesting increases the species' risk of extinction. The species does not tolerate human disturbance, and recreational activities within the species' riverside nesting grounds has the potential to disrupt the species' breeding success. Reduction in habitat quality is likely to increase the vulnerability of black stilt to predation (see Factor C). We find that the black stilt population is at significant risk throughout all of its range by the present or threatened destruction, modification, or curtailment of its habitat.

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

the species from use for commercial, recreational, scientific, or educational purposes. The species has not been formally considered for listing in the Appendices of CITES (http://www.cites.org).

C. Disease or Predation

There are currently no known diseases affecting the black stilt in the wild. Jakob-Hoff (2001) of the Auckland Zoo Wildlife Health and Research Centre, New Zealand, conducted a risk assessment for disease transmission caused by the translocation of captive black stilt to the wild population. The assessment considered a number of ‘‘diseases of concern’’ that may potentially threaten the wild population, including salmonellosis, yersiniosis, campylobacteriosis, pasteurellosis (fowl cholera), capillariasis, cestodiasis, trematodiasis, avian malaria, and coccidiosis. The assessment found no reported major die-offs of wild black stilts resulting from infectious diseases carried by birds translocated from captivity to the wild. Most of the illnesses and deaths that occurred among captive-reared birds were related to husbandry and could be controlled with improved husbandry methods, such as improved diet and parasite screening. Finally, the assessment suggested the establishment of a surveillance program to determine the prevalence of significant disease outbreaks in wild black stilts and facilitate development of pre-release quarantine and health-screening protocols regarding captive-reared birds (Jakob-Hoff 2001). A screening program for potential pathogens and improved husbandry methods specific to the black stilt captive population were outlined in the 1998 management plan for captive black stilts (Reed 1998). In 2005, a review of the records since 1995 for captive-held birds showed that infection, along with trauma, was a major cause of death among all age classes in captivity, especially chicks within the first two weeks after hatching (Van Heezik *et al.* 2005). Van Heezik *et al.* (2005) reported that protocols that monitor birds, intervene at the first signs of illness, and minimize the introduction of pathogens into the breeding unit were strictly adhered to. This has prevented the spread of these infectious diseases among captive-held birds or transmission into the wild populations (Van Heezik *et al.* 2005).

Predation by introduced mammalian predators and by unnaturally high numbers of avian predators is a primary threat to the black stilt (R. Maloney in litt. February 2007). Non-native predators introduced since the late 19th century include feral cats (*Felis catus*), ferrets (*Mustela furo*), stoats (*M. erminea*), hedgehogs (*Erinaceus europaeus*), and brown rats (*Rattus norvegicus*) (Maloney & Murray 2002; R. Maloney in litt. February 2007; Pierce 1996; Sanders & Maloney 2002). In addition, population numbers of avian predators, such as the non-native Australian harrier (*Circus approximans*) and the native kelp gull (*Larus dominicanus*), are unnaturally high because of human-induced changes, such as the introduction of rabbits, agricultural development, and the presence of rubbish dumps (Dowding & Murphy 2001; Maloney & Murray 2002). New Zealand is home to only one native mammal, a species of bat, and introduced mammalian predators pose a great risk to native bird species of New Zealand, including the black stilt, because these species evolved in the absence of these predators (Caruso 2006).

Several aspects of the black stilt's life history and nesting behavior contribute to heavy predation losses (Dowding & Murphy 2001). Solitary ground-nesting birds, the black stilt's preference for open nesting sites and feeding areas, such as dry, stable riverbanks, may increase their susceptibility to predation by mammalian predators, such as feral cats and ferrets, which use the banks as pathways (Maloney & Murray 2002; Pierce 1982; Pierce 1986b; Reed *et al.* 1993a). Nesting as early as August, when other prey sources are less available, adds to the black stilts' vulnerability (Reed *et al.* 1993a). Both sexes share nesting responsibility during the lengthy fledging period and are equally vulnerable to predation during the breeding season (Maloney & Murray 2002; Pierce 1986b; Pierce 1996; Sanders & Maloney 2002). Black stilts exhibit ineffective anti-predator behavior, contributing to significant mortality of nestlings and fledglings (Maloney & Murray 2002). For instance, black stilts do not perform distraction displays until late in incubation (Reed *et al.* 1993a). They will also re-nest in the same site if a clutch is lost to predation (Pierce 1986b; Sanders & Maloney 2002).

To test the effects of predation on the black stilt, Pierce (1986a) undertook a predator control study in a portion of the species' range during three breeding seasons, from 1977 to 1979, monitoring a total of 50 nests. Traps were placed around 23 randomly selected nests; these nests were ''protected.'' These and the remaining 27 nests, designated as ''unprotected,'' were monitored. Pierce (1986a) determined that 64 percent of black stilt breeding failures were attributed to predation and found that success in fledging and breeding increased at protected nests to 32.5 percent and 10.8 percent, respectively (R. Maloney in litt. February 2007). Most predation was caused by brown rats (14 nests), ferrets (13 nests), and cats (11 nests).

In a review of 499 eggs placed in the wild from 1979 to 1999, mortality was attributed to predation (45 percent); unknown causes (43 percent); flooding (10 percent); and human disturbance, disease, cold weather, poor parenting, and starvation (2 percent) (Maloney and Murray 2002). However, direct observation of predation events is difficult (R. Maloney in litt. February 2007), and, of all these deaths, only 11 were known conclusively (5 of which were directly observed predation events).

In an unpublished report by Saunders *et al.* (1996, as cited in Dowding & Murphy 2001), predation may have accounted for nearly 77 percent of black stilt chick losses between 1982 and 1995. Using video cameras, Sanders and Maloney (2002) studied the causes of mortality on ground-nesting birds in the UWB. The study monitored 23 black stilt nests and recorded 5 lethal events attributed primarily to cats and harriers. Cats were observed eating eggs, killing an adult nesting bird, and stalking nests. One black stilt nest containing ceramic eggs was visited by cats nine times over a 32-day period. A harrier ate a chick and a hatching egg in another nest. Unlike other bird species being observed in the same study, black stilts continued to nest upon dummy eggs even after being visited by cats, revealing that the use of dummy eggs increased their risk of mortality and further confirming that the species is ill-adapted to this predation pressure (Sanders & Maloney 2002).

Despite 20 years of predator trapping undertaken by the New Zealand Department of Conservation to protect black stilt nesting and fledging attempts, predator control efforts have met with mixed success. Fledging success (the number of chicks fledged versus the number of chicks hatched) was increased in some but not all years (Keedwell *et al.* 2002). In a review of predator trapping activities conducted between 1981 and 2000, Keedwell *et al.* (2002) found that efforts were inconsistent, resulting in highly variable results each season. For instance, predator control was sometimes undertaken for the entire breeding season but other times began well after the start of the breeding season. Keedwell *et al.* (2002) calculated that over the 20-year management period, the effort expended in predator control was equivalent to roughly 9.8 ''person years.'' According to Dr. Maloney (in litt. March 2007), the intensity and scale of control need to be significantly expanded to be effective in increasing fledgling survival and recruitment.

Summary of Factor C

For the reasons outlined above, we believe that disease is not currently a contributory threat factor for the black stilt. Predation by introduced mammalian and avian predators causes black stilt mortality at all life stages. Despite evidence that predator control significantly increased the species' breeding success, predator control efforts have been limited and inconsistent. We consider predation to be a significant contributory factor currently threatening this species and one that is projected to continue in the future.

D. The Inadequacy of Existing Regulatory Mechanisms

Four aspects are considered under this factor: National protection, habitat protection, the black stilt's status as a culturally significant species, and the species' management plans.

National protection: The black stilt is an ''absolutely protected'' species under the New Zealand's Wildlife Act of 1953 (1953 Act No. 31 1953). Under this Act, it is illegal to (a) hunt or kill; (b) buy, sell, or otherwise dispose of, or have possession of any absolutely protected wildlife or any skin, feathers, or other portion, or any egg of any absolutely protected wildlife; or (c) rob, disturb, or destroy, or have possession of the nest of any absolutely protected species (Part 5, 63(1)). Violations of this law by individuals can result in imprisonment for a term not exceeding 6 months; or a fine not exceeding $100,000 plus a further fine not exceeding $5,000 for each head of wildlife and egg of wildlife in respect of which the offence is committed (Part 5, 67(A)(1)(a)). Violations by corporations can result in a fine not exceeding $200,000 plus a further fine not exceeding $10,000 for each head of wildlife and egg of wildlife in respect of which the offence is committed (Part 5, 67(A)(1)(a)). Given that take by humans is not a threat to the black stilt, this law does not reduce any threats to the species.

Habitat protection: New Zealand protects more than 30 percent of its total land area as reserve land (Craig *et al.* 2000; Green & Clarkson 2006). However, except for a few small and scattered wetland reserves, most black stilt habitat is unprotected by the government (Maloney & Murray 2002). Habitat modification, including diversion or use of water for electrical generation, agriculture, and recreational activities (as discussed under Factor A), is a primary threat to this species.

The Waitaki Catchment Water Allocation Plan addresses water allocation for activities that involve the take, use, damming, and diversion of water in relation to the Waitaki Catchment. The most recent plan was approved in 2004 by the New Zealand Ministry for the Environment, in accordance with the Resource Management Act of 1991 and the Resource Management (Waitaki Catchment) Amendment Act of 2004 (NZ MFE 2005). The objectives of the

Waitaki Catchment Regional Plan were to balance electrical generation with conservation and other human uses of the Catchment, including an evaluation of minimum lake levels required to achieve these objectives. The evaluation gave specific consideration to the effect of water flow changes on the feeding, roosting, and breeding habitat of the black stilt (and other wetland birds), and it was determined that the established water levels were suitable for these wetland species (NZ MFE 2005). However, the Waitaki Catchment Regional Plan provided exemptions for other activities that also adversely affect black stilt and its habitat, including certain agricultural uses and recreational activities (See Factor A). Policy 35 of the Waitaki Catchment Water Allocation Plan exempts certain activities from allocation limits, including ''tourism and recreational facilities from the lakes [Tekapo, Pukaki and Ohau] and from the canals leading from them'' (NZ MFE 2004). Rule 2(2) of the Waitaki Catchment Water Allocation Plan exempts ''stock drinking-water * * * and processing and storage of perishable produce'' from consideration under the allocation limits (NZ MFE 2005). Thus, while the Waitaki Catchment Water Allocation Plan addresses regulation on water levels associated with hydroelectric power generation, it did not address or reduce threats to black stilt habitat from water diversion for certain agricultural and recreational activities, which is adversely affecting the black stilt (Factor A).

Status as a culturally significant species: The UWB is considered a ''*taonga,*'' and the black stilt a ''*taonga*'' species for the Ngai tahū, the native tribal population inhabiting most of the South Island, New Zealand (Schedule 97 1998; NZ MFE 2005). ''*Taonga*'' is a Maori word for any item, object or thing that has special significance to the culture, including birds and plants (Auckland Museum 1997). Under the Ngai tahū Claims Settlement Act of 1998, the New Zealand Department of Conservation must consult with, and have particular regard to, the views of the Ngai tahū when making management decisions concerning ''*taonga*'' species (1998 Act No. 97. 1998; Maloney & Murray 2002). An Ngai tahū representative is a member of the Kakï Recovery Group (Maloney in litt. February 2007), which implements the management plan for the black stilt (Maloney & Murray 2002). Including the tribes in resource decision-making is an important conservation strategy undertaken by the New Zealand government (NZ MFE 2001). New Zealand's Resource Management Act of 1991 is based on sustainably managing resources, while encouraging community and individual involvement in the planning for conservation (NZ MFE 1991). We believe that local involvement is important for resource conservation and may help to reduce threats to the species by increasing awareness of the conservation risks.

Management plans: According to the New Zealand Ministry of Environment, high priority is afforded to the black stilt recovery plan (NZ MFE 1997). Beginning in 1981, the New Zealand Department of Conservation undertook management of the wild black stilt population to increase fledging success and recruitment of juveniles in the declining populations in Mackenzie basin (R. Maloney in litt. March 2007; Reed *et al.* 1993b). Since 1993, black stilt management has been guided by two consecutive recovery plans, the first published in 1993 (Reed *et al.* 1993a) and a second, updated plan approved in 2002 (Maloney & Murray 2002), that covers the period 2001–2011.

The goals of the current recovery plan (effective from 2001 to 2011) are to increase the black stilt population within the next 10 years to more than 250 breeding individuals, with a mean annual recruitment rate that exceeds the mean annual adult mortality rate (Maloney & Murray 2002). There are two overlapping phases. Phase 1 of the program involves a series of objectives aimed at increasing the number of black stilts in the wild by maximizing recruitment rate both in the wild (for instance, by ensuring that all female black stilts are mated with a male each season) and by captive-rearing black stilts and releasing large numbers of captive-born young to the wild. A review of captive breeding records from two breeding seasons (1981 to 1982 and 2001 to 2002) found that the survival rate of captive-bred stilts that were reintroduced to the wild was 88 percent at 2 months and 82 percent at 10 months (Van Heezik *et al.* 2005). Between 1992 and 1999, researchers determined that the recruitment rate of chicks that had been artificially incubated in captivity and then hatched and raised in the wild was only 4 percent, with only 8 of the 189 chicks surviving to 2 years of age. However, birds that were hatched and raised in captivity and then released into the wild achieved a minimum recruitment rate of 22 percent (Maloney & Murray 2002). Thus, wild losses of eggs, chicks, and fledglings are largely avoided by artificially incubating and captive-rearing young to 3 or 9 months of age before releasing them back to the wild. This technique has been used for most eggs since 1998, and has resulted in approximately 30 percent recruitment rate (Van Heezik *et al.* 2005).

A second concurrent phase seeks to increase black stilt breeding success and adult survival in the wild by continuing research on the primary causes of mortality and developing mitigation measures to prevent excess mortality. Attempts to monitor all forms of mortality via direct observation began in 1998 and are ongoing. Goals under this phase include obtaining a better understanding of the causes of chick and adult mortality, developing multi-species predator control methods, and understanding mate choice decisions at different population densities. As an example, because monitoring birds between post-flight to adulthood is difficult, researchers are monitoring adults using transmitters (Maloney & Murray 2002). In September 2007, researchers released 38 adult black stilts fitted with transmitters (Timaru Herald 2007). These transmitters help researchers locate wild birds that have died (Maloney & Murray 2002).

The management of the captive black stilt population is addressed in both recovery plans (Reed *et al.* 1993; Maloney & Murray 2002), and also in a separate Department of Conservation management plan published in 1998 (Reed 1998). According to Reed (1998), the goals of the captive management plan are to provide young birds for release into the wild and develop a self-sustaining captive population. Five objectives were established to achieve these goals: (1) Establish a captive population capable of being self-sustaining, (2) provide juveniles for release and eggs for fostering to the wild, (3) undertake research to increase productivity and survival, (4) establish health monitoring of the captive population, and (5) advocate conservation of black stilts to the general public. This management plan outlines the expansion of the captive breeding program and formalizes the protocols for captive release, health screening, and monitoring.

Experts consider that, despite only incremental success in increasing wild population numbers, the captive-breeding program, along with predator control, have prevented the species from going extinct in the wild (BLI 2007e; Maloney & Murray 2002: Reed *et al.* 1993; Van Heezik *et al.* 2005). The management plans are addressing several aspects to facilitate the species' recovery, including research into survival, production of offspring for release into the wild, and continued research into the causes of mortality in the wild, including predation. However, the relative success of the captive breeding program is hindered by the inadequacy of regulatory mechanisms, combined with limited or inconsistent efforts to control predators (Factor C) and conserve and provide suitable habitat for the species (Factor A).

Summary of Factor D

Regulatory mechanisms exist to protect the black stilt from take. However, take is not a primary threat to the species. Government-sponsored measures are in place to facilitate the species' recovery (as discussed under this factor), including mitigating threats from predation (as discussed under Factor C). However, the inadequacy of regulatory mechanisms to protect or curb habitat destruction in the species' only known breeding ground (Factor A), combined with inconsistent predator control (Factor C), results in failure to reduce or remove threats from the species' habitat. As such, we believe that the inadequacy of regulatory mechanisms is a contributory risk factor currently and in the future for this species.

E. Other Natural or Manmade Factors Affecting the Continued Existence of the Species

Three additional factors are considered herein: Genetic risks associated with small population sizes, hybridization, and threats from stochastic events (random natural occurrences).

Genetic risks associated with small population sizes: The small size of the black stilt population, estimated in 2007 as 87 adults consisting of 17 breeding pairs (DOC 2007b), makes this species vulnerable to any of several risks, including inbreeding depression, loss of genetic variation, and accumulation of new mutations. Inbreeding can have individual or population-level consequences either by increasing the phenotypic expression (the outward appearance or observable structure, function or behavior of a living organism) of recessive, deleterious alleles or by reducing the overall fitness of individuals in the population (Charlesworth & Charlesworth 1987; Shaffer 1981). Small, isolated populations of wildlife species are also susceptible to demographic problems (Shaffer 1981), which may include reduced reproductive success of individuals and chance disequilibrium of sex ratios. Research has shown that the long-term survival of the black stilt as a species requires gene flow to be at least 5 percent, and that the present gene flow is approximately 15 percent (Maloney & Murray 2002). However, the relatedness of the entire black stilt population has not been determined, and inbreeding depression is a possible threat (Maloney & Murray 2002).

A general approximation of minimum viable population size is the 50 / 500 rule (Soulé 1980; Hunter 1996). This rule states that an effective population (N_e) of 50 individuals is the minimum size required to avoid imminent risks from inbreeding. N_e represents the number of animals in a population that actually contribute to reproduction, and is often much smaller than the census, or total number of individuals in the population (N). Furthermore, the rule states that the long-term fitness of a population requires an N_e of at least 500 individuals, so that it will not lose its genetic diversity over time and will maintain an enhanced capacity to adapt to changing conditions.

The available information for 2007 indicates that the breeding population of the black stilt (based on the number of wild and captive-held breeding pairs) is 46 individuals (DOC 2007b); 46 is just below the minimum effective population size required to avoid risks from inbreeding (N_e = 50 individuals). Moreover, the upper limit of the population is 102 adults (DOC 2007b). This represents the maximum potential number of reproducing members in the wild black stilt population and is less than one-fifth of the upper threshold (N_e = 500 individuals) required for long-term fitness of a population that will not lose its genetic diversity over time and will maintain an enhanced capacity to adapt to changing conditions. As such, we currently consider the species to be at risk due to lack of near- and long-term viability.

Hybridization: Black stilt males and pied stilt females can produce fertile offspring (BLI 2007e; DOC 2007a; Maloney & Murray 2002; Reed *et al.* 1993a). However, hybrid offspring exhibit distinct differences in survival rate and behavior that may be deleterious to the species' long-term survival (Reed *et al.* 1993a). Hybrid survival to adulthood is about 50 percent that of the offspring of pure black stilt pairs. In addition, researchers noted changes in behavioral patterns in chicks fostered to pied stilt parents between 1981 and 1987. Due to the limited number of wild black stilt breeding pairs, part of the species' management plan at that time was to cross-foster black stilt eggs to pied stilt parents. Cross-fostered black stilts were half as likely to be re-sighted in the UWB and mixed pairs were more likely to participate in migratory behavior with the pied stilt population rather than remain in their natal range, as pure black stilts would. As a result, cross-fostering of black stilt eggs with pied stilt parents was discontinued. More importantly, this research revealed that hybridization was detrimental to the long-term survival of the black stilt, as mixed pairs were effectively ''lost'' from the population (Reed *et al.* 1993b).

Hybrid management (such as breaking up mixed-pair bonds prior to mating) is part of the conservation strategy identified in the black stilt recovery plan, and researchers believe black stilts possess several inherent qualities that reduce gene flow, such as the black stilt's strong positive assortative mating (selecting black stilt over pied stilt when given the choice) and the low fitness of hybrid offspring (Maloney & Murray 2002). However, black stilts live in relative isolation from each other, and nesting pairs are often located miles (kilometers) apart (BLI 2007e; DOC 2007a; Pierce 1984a; Reed *et al.* 1993a). Sex ratios are an important indicator of the species' tendency to pair with pied stilts (Maloney & Murray 2002), and experts note that black stilts pair with the pied stilt when ''suitable'' mates within the species are not available (DOC 2007a; Greene 1999; NZ CMaG 2007; Reed *et al.* 1993a). Given the species' dispersed nature, the likelihood for hybridization with the growing population of pied stilts increases as black stilt population numbers decrease and black stilt males are less able to find females (Greene 1999; Pierce 1996).

Threats from stochastic events: With a wild adult population of 87 adults (DOC 2007b), experts consider the risk of a single catastrophic event to be a serious threat that could destroy most of the population (Maloney & Murray 2002). New Zealand's South Island is subject to tsunamis and earthquakes. According to the New Zealand Institute of Geological and Nuclear Sciences (NZ GNS) (2007), since 1840, when tsunami recordkeeping began, 10 tsunamis measuring 16.4 ft (5 m) or higher have hit New Zealand. New Zealand is vulnerable to tsunamis because of the high amount of seismic activity in the region. Approximately 10,000 to 15,000 earthquakes occur in New Zealand annually, most of low magnitude (Quake Trackers 2007). New Zealand is expected to experience earthquakes of magnitude of 7 on the Richter scale only about once a decade (Walsh 2003). However, since 2003, the southern region of the South Island has been rocked by at least three earthquakes near or above that magnitude. Centered in or near Fiordland, 266 mi (429 km) south of the heart of black stilt territory (The

New Zealand (NZ) Herald 2004, 2007; Walsh 2003), the years and magnitudes of each of these high-magnitude earthquakes were: 2003, 7.2 magnitude; 2004: 7.2 magnitude; 2007: 6.7 magnitude (NZ Herald 2004, 2007; Walsh 2003). The 2003 earthquake was the first on-land earthquake of this magnitude since 1968 (Walsh 2003). The main quake triggered a small tsunami that brought flooding as far north as Haast (Jackson Bay), less than 100 mi (161 km) from the UWB, where the majority of the black stilt population lives year-round and the only known breeding ground for the species (McGinty & Hancox 2004; Walsh 2003). At least 5,000 aftershocks were recorded from the 2003 earthquake, one registering 6.1 on the Richter scale (McGinty & Hancox 2004; NZ Herald 2007). More than 400 landslides were triggered, the largest of which sent 262,000 cubic yards (yd^3) (200,000 cubic meters (m^3)) of soil crashing down the fiord at Charles Sound, triggering a 3 to 6 ft (1 to 2 m) high tsunami that inundated surrounding vegetation 13 to 16 ft (4 to 5 m) above sea level (McGinty & Hancox 2004). According to Maloney and Murray (2002), flooding was the second leading cause of egg mortality in a study conducted between 1977 and 1979. Stochastic events, such as earthquakes and tsunamis, could result in extensive mortalities from which the population may be unable to recover, leading to extinction (Caughley 1994; Charlesworth & Charlesworth 1987; Maloney & Murray 2002).

Summary of Factor E

The black stilt is subject to genetic dilution, including changes in survival and behavior, due to demographic problems and hybridization with the pied stilt, and is also susceptible to other genetic risks, such as inbreeding, due to its small population size. The species is vulnerable due to stochastic event, such as a tsunamis or earthquakes, which are known to occur in the region. We consider the species' extremely small population size, along with the associated risks of genetic dilution, demographic shifts, and vulnerability to stochastic events, to be significant risks factors throughout the black stilt's range currently and in the future.

Conclusion and Determination for the Black Stilt

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the black stilt. We have determined that the species is in danger of extinction throughout all of its known range primarily due to ongoing threats to its habitat (Factor A); predation (Factor C); and genetic dilution from hybridization, lack of near- and long-term genetic viability, and susceptibility to stochastic events due to risks associated small population sizes (Factor E). Furthermore, we have determined that the inadequacy of existing regulatory mechanisms is a contributory risk factor that endangers the species' continued existence (Factor D). Therefore, we are determining endangered status for the black stilt under the Act. Because we find that the black stilt is endangered throughout all of its range, there is no reason to consider its status in any significant portion of its range.

II. Caerulean Paradise-Flycatcher (*Eutrichomyias Rowleyi*)

Species Description

The caerulean paradise-flycatcher is a member of the Monarchidiae family, locally known as ''*burung niu*'' (Whitten 2006). It is native to Indonesia, and adults are about 5 in (18 cm) in height, with a long tail and long rictal bristles (stiff hairs around the base of the bill) (Riley & Wardill 2001; Whitten *et al.* 1987). There is scant biometric data for this species, because, other than the type specimen, only one additional specimen was captured, measured, and released in 1998 (Riley & Wardill 2001). The species is described as a bright cerulean blue (which can be likened to a deep blue sky) with gray undertones on the belly, legs, upper wing coverts (feathers) and down the sides of the neck to the breast (BLI 2007d; Riley & Wardill 2001; Whitten *et al.* 1987). The type specimen, which was described as a male, is slightly larger and duskier in appearance than the specimen measured in 1998, leading researchers to believe that the former specimen was a juvenile and the latter, a female (Riley & Wardill 2001).

Taxonomy

The first specimen of caerulean paradise-flycatcher was collected by Meyer in 1873. The species has always been placed in the Monarchidiae family, but within three different genera. When described in 1878, Meyer placed the species in the genus *Zeocephus*; later it was placed in the genus *Hypothymis* (Riley & Wardill 2001; Whitten *et al.* 1987). In 1939, it was placed into the monotypic genus *Eutrichomyias*, also of the Monarchidae family, and distinguished from *Hypothymis* by its abundant rictal bristles (Riley & Wardill 2001). Riley and Wardill (2001) suggest that the species may be more related to *Hypothermis*, but insufficient information impedes a conclusive decision. Therefore, we accept the species as *Eutrichomyias rowleyi*, which follows the Integrated Taxonomic Information System (ITIS 2007).

Habitat and Life History

The caerulean paradise-flycatcher was known only from its type specimen until 1998. Current knowledge of its ecology and behavior are based on 33 sightings between 1998 and 1999 (Riley & Wardill 2001; Whitten *et al.* 1987). Riley and Wardill (2001) point out that the basic lack of ecological information on this species impedes its conservation. Information about the species' range, behavior, reproduction, and population size is quite limited.

The species has been observed mostly in the steep-sloped, closed canopies of low-elevation broadleaf primary forest, between 1,394 and 2,133 ft (425 and 650 m). A few birds were observed foraging on a scrub forest ridge top or in secondary forest, but only when those areas were bordered by primary forest. The caerulean paradise-flycatcher prefers primary forest habitat, but can forage in secondary scrub that is bordered by primary forest; however, the species is absent from disturbed habitat away from primary forest (www.rdb.or.id; BLI 2001a, 2007d; Riley & Wardill 2001).

The species is often observed foraging in association with other bird species and a particular squirrel species, believed to be the Celebes dwarf squirrel (*Prosciurillus murinius*) (Riley & Wardill 2001). Adept at catching flies in the air, this insectivore feeds primarily in the canopy and sub-canopy, but is known to descend to the understory (*http://www.rdb.or.id*; BLI 2001a, 2007d; Riley & Wardill 2001).

Experts believe that the species is sedentary, as individuals do not appear to move between the valleys in which they are observed (www.rdb.or.id; BLI 2001a, 2007d; Riley & Wardill 2001). The largest recorded flock size has been five birds (Riley & Wardill 2001). Based on two sightings of young, in October and in December, researchers presume that nesting and fledging occur in that time period (www.rdb.or.id; BLI 2001a; Riley & Wardill 2001). Researchers believe the bird builds nests of palm leaves (likely *Arenga* spp.) in the branches of understory trees (including *Szygium* spp.) from 7 to 8 ft (2 to 2.5 m) off the ground (www.rdb.or.id; BLI 2001a; Riley & Wardill 2001). Both sexes appear to care for the young (Riley & Wardill 2001).

Historical Range and Distribution

The only known range of the caerulean paradise-flycatcher is on Sangihe Island, north of Sulawesi, Indonesia (Riley & Wardill 2001; Whitten *et al.* 1987). Sangihe Island, also known as Great Sangihe, Great Sangir, or Sangir Besar Island, is part of the Sangihe-Talaud archipelago (Whitten *et al.* 1987) in the waters between Sulawesi (northern Indonesia) and the Philippines (Brodjonegoro *et al.* 2004). The archipelago consists of two island groups, the Sangihe group and the Talaud group, and until 2002, the entire island group was administered as one unit. Thus, most available information on the archipelago concerns both island groups.

The Sangihe-Talaud archipelago includes 77 islands; 56 are inhabited, including Sangihe (Brodjonegoro *et al.* 2004). The total land mass of the Sangihe-Talaud archipelago is 314 mi^2 (813 km^2) (Mous & DeVantier 2001), of which Sangihe Island includes 270 mi^2 (700 km^2) (Riley 2002), making it the largest island in the archipelago. The Island became part of the Dutch East India Company in the 17th century, and remained primarily under Dutch control for the next 300 years (Simkin and Siebert 1994). In some of the earliest accounts, Sangihe Island was already known for its coconut and nutmeg plantations (New York Times Archives 1892). Most of Sangihe Island was deforested by 1920, having been logged for timber and paper production or converted to cash crop plantations (Riley 2002; Riley & Wardill 2001; Whitten *et al.* 1987).

The extent of the caerulean paradise-flycatcher's historic distribution is not well known because there have been so few sightings of this species. Following the initial discovery of the species in 1873, there were only two reported sightings; both unconfirmed (Riley & Wardill 2001). By the 1980s, with no confirmed sightings of live caerulean paradise-flycatchers for over 100 years, the species was presumed extinct due to loss of habitat (Riley & Wardill 2001; Thompson 1996; Whitten *et al.* 1987).

Current Range and Distribution

The caerulean paradise-flycatcher was rediscovered in 1998 (Riley & Wardill 2001), occupying the forested valleys around the base of Mount Sahendaruman, on the southern part of Sangihe Island (www.rdb.or.id; BLI 2001a; BLI 2005; Riley & Wardill 2001). An extinct volcano, Mt. Sahendaruman is variously referred to as: Gunungan Sahendaruman and Gunungan Sahengbalira (the latter of which is actually the name of a mountain peak) (*http://www.rdb.or.id*; BLI 2001a) and Pegunungan Sahendaruman (BLI 2004b). Mt. Sahendaruman supports the only extensive remaining primary forest on the island (*http://www.rdb.or.id*; BLI 2001a, 2007d; Riley & Wardill 2001) and is home to three critically-threatened species of birds, including the caerulean paradise-flycatcher; no other area in Indonesia supports more than one critically threatened bird species (BLI 2001a).

Mt. Sahendaruman extends to an altitude of approximately 3,382 ft (1,031 m) (Riley 2002). The entire forest covers an area of less than 3 mi^2 (8 km^2). However, because of the species' preference for riverine habitat at elevations from 1,394 to 2,133 ft (425 to 650 m), the actual range available to the flycatcher is estimated to be an area of 0.8 mi^2 (2 km^2) on the lower valleys near the fringe of the forest (www.rdb.or.id; BLI 2001a, 2007d; Riley & Wardill 2001). Moreover, because the species is rarely seen at higher elevations, experts believe that this species has reached its upper elevational limit (Riley & Wardill 2001).

Population Estimates

The population is estimated to be between 19 and 135 individuals. This estimate is based on inferences made from 33 sightings between 1998 and 1999 (www.rdb.or.id; BLI 2001a, 2007d; Riley & Wardill 2001). The basis for this estimate is well explained by Riley and Wardill (2001, p. 49), who note the possibility that the total population may consist of only those 19 observed birds. More recent census data is not available.

Conservation Status

The caerulean paradise-flycatcher is a protected species in Indonesia (J.C. Wardill in litt. 1999, as cited in BLI 2001a). The IUCN considers this species to be ''Critically Endangered'' due to its low estimated population size and restricted range (BLI 2004a).

Summary of Factors Affecting the Caerulean Paradise-Flycatcher

A. The Present or Threatened Destruction, Modification, or Curtailment of the Caerulean Paradise-flycatcher's Habitat or Range

Today, much of Sangihe Island is covered by plantations or secondary forests and the caerulean paradise-flycatcher's habitat on Mt. Sahendaruman provides the only remaining extensive primary forest on the island (Riley & Wardill 2001; Whitten *et al.* 1987). Land use patterns on Sangihe Island have been fairly stable (Vidaeus 2001), and there have been no significant forest losses on Sangihe Island (Whitten 2006) because the Sangihe Island economy is not driven by timber harvest as in other parts of Indonesia. The inaccessibility of Mt. Sahendaruman forest made timber extraction uneconomical (Vidaeus 2001). However, Riley & Wardill (2001) noted that the caerulean paradise-flycatcher likely only existed on Mt. Sahendaruman because of the steep, fairly inaccessible terrain.

Most threats to the caerulean paradise-flycatcher habitat have been locally derived (Vidaeus 2001), caused by smaller scale activities on the lower fringes of the primary forest on Mt. Sahendaruman (Riley & Wardill 2001), including within the boundaries of the Mt. Sahendaruman Protection Forest (see Factor D). Forest clearing by farmers is generally small scale, between 53,820 to 161,459 square ft (ft^2) (5,000 to 15,000 m^2), and occurs along the fringes of the primary forest, which is adjacent to the species' preferred habitat. BirdLife International (2006c) reported that shifting cultivation has caused the gradual erosion of the lower fringes of the primary forest on Mt. Sahendaruman. Encroachment for forest product extraction on the fringes of the forest also disrupts the flycatcher's habitat (www.rdb.or.id; BLI 2001a, 2007d Kirby 2003a; Riley & Wardill 2001). Forest is also cleared for wood, paper production, conversion to cash crops, shifting cultivation, and settlements (Riley & Wardill 2001; Whitten *et al.* 1987). Researchers believe that the species has reached its upper elevational limit and that human pressures on the lower fringes of its habitat have boxed the species into its current range (www.rdb.or.id; BLI 2001a; Riley & Wardill 2001).

Summary of Factor A

The caerulean paradise-flycatcher is currently limited to an area of suitable habitat that may be as small as 0.8 m^2 (2 km^2) on Mt. Sahendaruman. Preferring lower elevations, the species appears to have reached its upper elevational limit for suitable habitat. Encroachment on the fringes at the base of the mountain threatens the species to the lower extent of its range. Given the caerulean paradise-flycatcher's limited range and preference for closed-canopy primary forest, habitat modification even at a small scale can have a profound effect on the species. Based on the above information, we believe that the present and future threatened destruction, modification, or curtailment of the caerulean paradise-flycatcher's habitat or range threatens the species throughout its range.

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

While there is no documented evidence that the species is a specific target of hunting, researchers familiar with the area and the species consider indiscriminate hunting to be a risk factor for this species (Riley & Wardill 2001; www.rdb.or.id; BLI 2001a). Sangihe Island locals are known for hunting birds indiscriminately with air rifles as a hobby in and around the forests of Mt. Sahendaruman (BLI 2001a; Riley & Wardill 2001). BirdLife International (2006c) describes hunting pressures on small passerines, to which group of birds the caerulean paradise-flycatcher belongs, as ''intensive.'' Riley and Wardill (2001) noted that while conducting fieldwork in Mt. Sahendaruman forest in 1998, a group of three hunters were observed carrying 20 to 30 birds of all sizes that had been shot.

Indiscriminate hunting has resulted in declines of more accessible bird species on the island (www.rdb.or.id; BLI 2001a) and locals have identified hunting as a key cause for the decline in bird species in the Mt. Sahendaruman area (BLI 2001a). The practice is so pervasive that BirdLife International—Indonesia Programme (Vidaeus 2001) has focused on creating educational materials aimed at school children to encourage them to find alternative hobbies to hunting. Given the species' extremely small population size, between 19 and 135 individuals, indiscriminate hunting of even a few individuals would have a detrimental effect on the population (See Factor E).

Riley (2002) conducted research on mammal hunting on Sangihe Island, finding that, after habitat loss, hunting pressure was the biggest threat on the island. In interviews with local farmers, 77 percent of the farmers admitted to hunting mammals variously using air rifles, snares and mist nets. Furthermore, hunting pressure was particularly high for the bear cuscus (*Ailurops ursinus melanotis*), a small marsupial found only in the primary forests of Mt. Sahendaruman, the same habitat as the caerulean paradise-flycatcher. Riley and Wardill (2001) characterize the flycatcher as adverse to human disturbance, and hunting pressures in the same habitat as the flycatcher contribute to disturbance activities that are disruptive to the species (as described under Factor A).

The species is not known to be in international trade and has not been formally considered for listing under CITES (www.cites.org).

Summary of Factor B

Indiscriminate bird hunting and hunting-related disturbances are widespread within the species' range (Mt. Sahendaruman forest). The species has an extremely small population size and is adverse to human disturbance. We consider incidental hunting and hunting disturbances to be factors that threaten this species throughout its range.

C. Disease or Predation

There is no available evidence indicating that disease or predation have led to decline in caerulean paradise-flycatcher populations or contribute to the species' risk of extinction.

D. The Inadequacy of Existing Regulatory Mechanisms

The caerulean paradise-flycatcher was declared a protected species by the Indonesian government in January 1999 (J. C. Wardill in litt. 1999 as cited in BLI 2001a). Protected species are regulated under the Act of the Republic of Indonesia No. 5 of 1990 Concerning Conservation of Living Resources and Their Ecosystems (Act No. 5 1990). Under this Act, hunting, capturing, killing, possession, or trade in protected species or their parts is prohibited, except as permitted for research, science, or conservation purposes (Article 21–22). Despite this law, an analysis conducted by the IUCN (World Conservation Union) in 2003 found that this species remained insufficiently protected (Conservation International 2003). Lee *et al.* (2005) noted that Indonesia has over ''150 existing national laws and regulations to protect its wildlife species and area * * * however, Indonesia lacks an integrated system of law enforcement'' (p. 478). Problems include lack of awareness of wildlife laws and inadequate monitoring capability among law enforcement officials (Lee *et al.* 2005). Evidence of continued indiscriminate hunting within the species' habitat indicates that the caerulean paradise-flycatcher's listing as protected in 1999 has not reduced the threat of hunting (Factor B).

The caerulean paradise-flycatcher's habitat lies within an approximately 16 mi^2 (43 km^2) area centered on Mt. Sahendaruman that has been designated as Protection Forest since 1994, under the jurisdiction of the Department of Forestry (Riley & Wardill 2001). However, Whitten (2006) noted that protection forests do not confer specific protections on the wildlife found therein; for example, hunting is not prohibited (Whitten 2006). Thus, the species is not adequately protected from hunting due to its presence within the Mt. Sahendaruman Protection Forest.

Plans that began in 2001 to have the Mt. Sahendaruman Protection Forest designated a wildlife preserve, with core areas as a strict nature reserve (www.rdb.or.id; BLI 2001a, 2007d; Riley & Wardill 2001), have not been implemented (Whitten 2006). However, such a designation might not benefit the species. According to experts, designating this habitat as a nature reserve would shift management of the area from the local government to the central government. This centralization of enforcement and administration might be unresponsive or ineffective in protecting the species and may not produce the most viable options for long-term conservation of the species (Vidaeus 2001; Whitten 2006). Because this designation has not been enacted, we are unable to evaluate whether this regulatory mechanism might effectively address the issues of habitat destruction (Factor A) and hunting (Factor B).

The species' habitat is also inadequately protected (BLI 2003a, 2004b; Conservation International 2003; Whitten 2006). There are no strictly protected areas on the island (Riley & Wardill 2001; Whitten 2006). The Mt. Sahendaruman Protection Forest is managed for its watershed value (Riley 2002; Riley & Wardill 2001). Although the Mt. Sahendaruman Protection Forest contains the only remaining primary forest on the island that is suitable for the caerulean paradise-flycatcher (Riley & Wardill 2001), small-scale forest conversion for agricultural purposes and non-timber forest product extraction occurs on the fringes of the forest (see Factor A). Local rights to manage cultivation and settlement areas within the Protection Forest are among the key disputes between locals and the forestry department (BLI 2001a). Thus, the habitat's status as a Protection Forest does not protect the species from threats of habitat modification.

The caerulean paradise-flycatcher has been included in a biodiversity project, Action Sampiri. Members of the Action Sampiri research team, Riley and Wardill, rediscovered this species in 1998 (Riley & Wardill 2001; Whitten 2006). Present-day members of Action Sampiri (now known as Yayasan Sampiri) were contracted to develop a public awareness program on the merits of enhancing forest protection as part of a comprehensive conservation project for the Sangihe-Talaud islands being implemented by BirdLife International and the World Bank, with funding from the Global Environment Facility

(Whitten 2006). Conservation efforts that focus on people's awareness of the forest and its value, including potential for ecotourism with the prospect for local employment opportunities, are considered important to the species' long-term conservation (BLI Indonesia Program 2001; Riley & Wardill 2001; Whitten 2006). For instance, the caerulean paradise-flycatcher is among the endemic birds designated as island mascots, which has promoted greater awareness of the species among locals and has led to a general reduction in indiscriminate hunting (www.rdb.or.id; BLI 2001a).

Summary of Factor D

Based on the above information, existing regulatory mechanisms are not adequate to reduce or remove threats from habitat destruction (Factor A) and hunting (Factor B). Encroachment and destruction along the fringes of the species' habitat are significant current and future threats for this species, yet the species' habitat is insufficiently protected. Further, the lack of enforcement of protections against take and inadequate protection within its habitat does not adequately reduce or remove the threat of hunting. We believe that the inadequacy of regulatory mechanisms and their enforcement are contributory risk factors that threaten the species now and in the future.

E. Other Natural or Manmade Factors Affecting the Continued Existence of the Species

The caerulean paradise-flycatcher's small estimated population size, between 19 and 135 individuals (BLI 2007d; Riley & Wardill 2001), makes this species vulnerable to any of several risks, including inbreeding depression, loss of genetic variation, and accumulation of new mutations. Inbreeding can have individual or population-level consequences by either increasing the phenotypic expression of recessive, deleterious alleles or by reducing the overall fitness of individuals in the population (Charlesworth & Charlesworth 1987). Small, isolated populations of wildlife species are also susceptible to demographic problems (Shaffer 1981), which may include reduced reproductive success of individuals and chance disequilibrium of sex ratios. In the absence of more species-specific life history data, a general approximation of minimum viable population sizes is referred to as the 50/500 rule (Soulé 1980; Hunter 1996), as described under Factor E of the black stilt. The available information indicates that the population of the caerulean paradise-flycatcher may be as small as 19 birds (www.rdb.or.id; BLI 2001a, 2007d; Riley & Wardill 2001); this is clearly below the minimum effective population size (N_e = 50 individuals) required to avoid risks from inbreeding. Moreover the upper limit of the population estimate of no more than 135 birds (www.rdb.or.id; BLI 2001a, 2007d; Riley & Wardill 2001) is a quarter of the upper threshold (N_e = 500) required for long-term fitness of a population that will not lose its genetic diversity over time and will maintain an enhanced capacity to adapt to changing conditions. As such, we currently consider the species to be at significant risk of potential demographic shifts and lack of near- and long-term viability.

Summary of Factor E

Demographic shifts and lack of near- and long-term viability associated with the extant population's small size are major risks to the caerulean paradise-flycatcher. Therefore, we consider the species' extremely small population size and the risks associated with loss of genetic diversity and demographic shifts to be significant factors that threaten the caerulean paradise-flycatcher throughout its range currently and in the future.

Conclusion and Determination for the Caerulean Paradise-Flycatcher

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the caerulean paradise-flycatcher. We have determined that the species is in danger of extinction throughout all of its known range primarily due to disturbance and encroachment of its habitat (Factor A), threats from hunting and hunting-related disturbances (Factor B), and lack of near- and long-term genetic viability associated with the species' small population size (Factor E). Furthermore, we have determined that the inadequacy of existing regulatory mechanisms to reduce or remove these threats is a contributory factor to the risks that endanger this species' continued existence (Factor D). Therefore, we are determining endangered status for the caerulean paradise-flycatcher under the Act. Because we find that the caerulean paradise-flycatcher is endangered throughout all of its range, there is no reason to consider its status in any significant portion of its range.

III. Giant Ibis (*Pseudibis Gigantea*)

Species Description

The giant ibis is a waterbird in the family Threskiornithidae. It is native to Cambodia, Lao People's Democratic Republic (hereafter, Lao PDR), and Vietnam. Adults stand approximately 3 ft (1 m) tall, and have dark grey-brown plumage, with a dark hindcrown and nape. Wing-coverts are pale gray, with darker tips. They have light red legs, a long downward curving bill, and red eyes. Juveniles have short, black feathers on their hindcrown and hindneck, a shorter bill, and brown eyes (BLI 2007h).

Taxonomy

The species was first taxonomically described by Oustalet in 1877 and named Pseudibis gigantea, in the Threskiornithidae family. That same year, Elliot placed the species in its own monotypic genus Thaumatibis, in the same family, on the basis that the giant ibis is much larger and less colorful than all other ibises (BLI 2007h). We accept the species as Pseudibis gigantea, which follows the Integrated Taxonomic Information System (ITIS 2007).

Habitat and Life History

The giant ibis requires large areas of undisturbed habitat in deciduous dipterocarp forest and associated wetlands (Tom Clements, Wildlife Conservation Society—Cambodia Program, Phnom Penh, Cambodia, in litt. December 2007). It is found in open habitats (open wooded plains, humid clearings) and deciduous forested wetlands (pools in deep forest, lakes, swamps, seasonally flooded marshes, paddy fields) (BLI 2007h; Collar *et al.* 1994b; Matheu & del Hoyo 1992). The mix of dry forest and freshwater swamp ecosystems is found only in this region (WWF 2001, 2005). Freshwater swamp habitat is flooded at least 6 months of the year and consists of shrubland (dominated by a nearly continuous canopy of deciduous species, including spurges (Euphorbiaceae family) and legumes (Fabaceae family)) and of forestland (dominated by mangroves (Rhizophoraceae family) and melaleucas (*Melaleuca* spp.)). The freshwater swamp ecosystem is found only in Cambodia and Vietnam (WWF 2001). Lower Mekong dry forests, found only in Cambodia, Lao PDR, and Vietnam, also provide habitat to the giant ibis. These forests are characterized by deciduous tropical hardwoods (Dipterocarpaceae family) and semi-evergreen forest (containing a mix of deciduous and evergreen trees) interspersed with meadows, ponds, and other wetlands. Semi-evergreen forests are unique to mainland Southeast Asia (WWF 2006b).

Although considered nonmigratory, the giant ibis will travel to seek out permanent pools of water during the dry season (Bird *et al.* 2006; Matheu & del Hoyo 1992). The giant ibis may forage alone, in pairs or in small groups (BLI 2007h). Preferring mudflats, they use their bills to probe in the mud for a variety of seeds and small animals, including invertebrates, small amphibians, and reptiles (Clements *et al.* 2007; Davidson *et al.* 2002). Although considered a wetland species, the giant ibis will also forage in dry areas; it is believed that this is an adaptation to the lengthy dry season within its range (www.rdb.or.id; BLI 2001b, 2007h; Davidson *et al.* 2002).

Until recently, little was known about giant ibis breeding biology, except that the species was believed to nest in trees as other ibises do (BLI 2007h). A nesting survey was conducted in Preah Vihear Protected Forest (PVPF) and Kulen Promtep Wildlife Sanctuary (KPWS) between 2004 and 2007 (Clements *et al.* 2007). The majority of giant ibises bred in remote areas, sing wetlands that have a minimal human presence (T. Clements in litt. December 2007). The number of nests remained fairly stable over the four years of the surveys, although their locations changed. Researchers found an average of 19 nests in the 534-mi^2 (1,383-km^2) area surveyed in PVPF and 7 nests in the 726-mi^2 (1,881-km^2) KPWS. Fledging success was estimated at around 50 percent, suggesting that the population was not increasing. Researchers determined that weather and predation were the primary limiting factors (Clements *et al.* 2007). See Factor C.

The giant ibis is characterized as highly sensitive to human disturbance (Bird *et al.* 2006; www.rdb.or.id; BLI 2001b, 2007h; T. Clements in litt. December 2007; Clements *et al.* 2007; Dudley 2007; Eames *et al.* 2004). Clements (in litt. December 2007) postulated that the species' sensitivity to human populations is due to disturbance (e.g., at feeding ponds) and incidental persecution through hunting and poisoning of water sources (see Factors A and B).

Historical Range and Distribution

The giant ibis's historical range extended from central and peninsular Thailand; through northern, central, and coastal regions of Cambodia; southern and central Lao PDR; and southern Vietnam (www.rdb.or.id; BLI 2001b).

A comparison of recorded observations of this species maintained by BirdLife International (2001b) paints an erratic picture of the ''appearance'' and ''disappearance'' of the giant ibis in each range country during the 20th century. The species has been suspected or considered extinct in each of its range countries at least once since it was first described in 1877. In the early part of the century, the species was observed most often in Thailand. In the mid-1920s, the species was seen only in Lao PDR, Cambodia, and Vietnam (www.rdb.or.id; BLI 2001b). By 1992, the species was considered extant only in Vietnam and possibly in Cambodia (Matheu & del Hoyo 1992). By the end of the 20th century, the species was considered extinct in Vietnam and Thailand, and extant primarily in Cambodia and in Lao PDR to a lesser extent (www.rdb.or.id; BLI 2001b, 2007h). Today, the species is considered extinct only in Thailand (www.rdb.or.id; BLI 2001b; Matheu & del Hoyo 1992).

Experts have noted several factors unrelated to the species' actual status that have contributed to this erratic record: (1) The records may not be complete because sightings may go unreported or unconfirmed for several years (BLI 2001b; Matheu & del Hoyo 1992) (e.g., in Vietnam, there were several unconfirmed sightings in the 1980s); (2) nearly continuous war in the last half of the 20th century in one or all of the range countries may have impeded expeditions to locate the species (Matheu & del Hoyo 1992) (e.g., Cambodia experienced a nearly 50-year period of war, during which time there were only four sightings of the species); and, (3) the habitat may be remote or the terrain difficult to access, which might also impede opportunities to observe the species (Duckworth *et al.* 1998). For these reasons, recorded sightings (or the lack thereof) cannot be used as a basis for concluding extinction (Butchart *et al.* 2006).

Specific information for each range country follows.

Cambodia: The first specimen of giant ibis was obtained in Cambodia in 1876, but no additional sightings were reported until 1918. Historically, the species' range spanned from the north through central region and into the eastern portions of the country. The giant ibis was observed several times in the 1920s and 1930s, but only four times between 1939 and 1989 (www.rdb.or.id; BLI 2001b). In 1992, experts believed the species might be extant in Cambodia, but indicated that the recent reports had been unconfirmed (Matheu & del Hoyo 1992). The species was observed again in 2000 (see Current Range, below). Disturbance and hunting are two factors attributed to the species' decline (Wildlife Conservation Society (WCS) 2007a, 2007b, 2007c).

Lao PDR: The giant ibis was not reported from Lao PDR until 1926. Thereafter, it was observed only once each decade in the 1930s and the 1940s. Based on the paucity of sightings, it was never believed to be common in Lao PDR (www.rdb.or.id; BLI 2001b). By 1992, the species was no longer considered extant in Lao PDR (www.rdb.or.id; BLI 2001b; Matheu & del Hoyo 1992), although the species was observed again the next year (see Current range, below). Historical declines are attributed to hunting and wetland draining or other human disturbances (www.rdb.or.id; BLI 2001b).

Thailand: This species was observed in Thailand several times between 1896 and 1913, at a time when it was not being reported in any of the other range countries, except for one sighting in Cambodia. All sightings were made in the southern regions of Thailand and there have been no confirmed sightings of this species in Thailand since 1913 (www.rdb.or.id; BLI 2001b). From the scant sightings of this species, researchers are uncertain whether the giant ibis was ever resident to Thailand, or just a visitor (www.rdb.or.id; BLI 2001b). Since 1992, the species has been considered extinct in Thailand, primarily due to loss of habitat from wetland draining (www.rdb.or.id; BLI 2001b; Matheu & del Hoyo 1992).

Vietnam: The species was observed once late in the 19th century and not seen again until the mid-1920s, when it was observed several times until 1931. By the turn of the 21st century, the giant ibis was believed extirpated from Vietnam, with no confirmed sightings between 1931 and 2003 (www.rdb.or.id; BLI 2001b; Eames *et al.* 2004). The species was rediscovered in 2003. Hunting is considered the primary cause of the historical decline, and land conversion to agriculture is a secondary cause (www.rdb.or.id; BLI 2001b).

Current Range and Distribution

The giant ibis' current range is the mix of dry forest and freshwater swamp forest ecosystems of Cambodia, Lao PDR, and Vietnam; it is considered extirpated from Thailand (BLI 2000a, 2001b; www.rdb.or.id; BirdLife International—Indochina Programme (BLI–IP) & Vietnam's Ministry of Agriculture and Rural Development (MARD) 2004; Eames *et al.* 2004; World Wide Fund for Nature (WWF) 2001, 2005). Each range country is discussed below.

Cambodia: Between 1992 and 2002, there were no confirmed giant ibis sightings in Cambodia. However, since 2002, the species has been observed at several sites throughout Cambodia. Observations in 2002 and 2003 suggest that the species continues to inhabit its historic range in the north, central, and eastern provinces. In the Northern Plains, the giant ibis has been observed in Stung Treng and Preah Vihar Provinces (bordering Lao PDR), and Kratie Province (Bird *et al.* 2006; www.rdb.or.id; BLI 2001b; Clements *et al.* 2007). The Northern Plains are considered the largest remaining contiguous tract of seasonally inundated meadows and permanent pools within a deciduous dipterocarp forest (Davidson *et al.* 2002). In central Cambodia, the species has been observed in the Tonle Sap floodplains (Kompong Thom and Siem Reap) (www.rdb.or.id; BLI 2001b; Clements *et al.* 2007). The Tonle Sap floodplain and associated rivers is considered one of the few remaining remnants of freshwater swamp forest type in the region. Approximately 2,120 mi² (5,490 km²) of the freshwater swamp forest ecoregion is protected in Cambodia. Of this amount, the Tonle Sap Great Lake Protected Area (which includes the Tonle Sap floodplain) makes up 2,092 mi² (5,420 km²) of that protected habitat (WWF 2001). In eastern Cambodia, the species has been located in the Lomphat Wildlife Sanctuary (Mondulkiri and Rattanakiri Provinces) (Bird *et al.* 2006; www.rdb.or.id; BLI 2001b; Clements *et al.* 2007; Davidson *et al.* 2002). The Lomphat Wildlife Sanctuary spans a 965 mi² (2,500 km²) area in northeastern Cambodia (in Mondulkiri and Rattanakiri Provinces) near the Vietnam border (WildAid 2003, 2005). The Lomphat Sanctuary is considered to be one of the most important areas for wildlife in Cambodia (WildAid 2005).

More recent sightings suggest that the giant ibis' range may extend further south and east than previously understood (Bird *et al.* 2006). The species has been observed in Kampot Province (the southernmost Province in Cambodia) (www.rdb.or.id; BLI 2001b) and in the buffer zone of Seima Biodiversity Conservation Area (SBCA) (Kratie and Mondulkiri Provinces, eastern Cambodia) (Bird *et al.* 2006; Clements *et al.* 2007). The SBCA was designated in 2002 and encompasses a 540 mi² (1,400 km²) area (WCS 2007b).

Lao PDR: The giant ibis was believed extinct in Lao PDR in 1992 (Matheu & del Hoyo 1992). The following year, an observation was confirmed and it has since been observed in Lao PDR several times. Based on surveys conducted in 1998, no giant ibises were found in central Lao PDR (Duckworth *et al.* 1998), indicating that the giant ibis may no longer be present in central Lao PDR, as it was historically (www.rdb.or.id; BLI 2001b). Previously suspected to be nonresident (www.rdb.or.id; BLI 2001b), however in 2007 it is being reported as a resident (BLI 2007b).

The giant ibis has been found in the open deciduous forest of two areas in extreme southern Lao PDR: Xe Pian National Biodiversity Conservation Area (NBCA) (Champasak and Attapeu Provinces) and Dong Khanthung proposed NBCA (Champasak Province) (www.rdb.or.id; BLI 2001b, 2007b; Clements *et al.* 2007; Poole 2002) and giant ibis may only be a frequent visitor to Lao PDR there from Cambodia. The Xe Pain NBCA is 927 mi² (2,400 km²) (www.rdb.or.id; BLI 2001c). The Dong Khanthung proposed NBCA has not yet been defined or approved (BLI 2007b).

Thailand: The species has not been observed in Thailand since 1913 (www.rdb.or.id; BLI 2001b).

Vietnam: At the turn of the 21st century, giant ibis was believed extirpated from Vietnam, with no confirmed sightings since 1931 (www.rdb.or.id; BLI 2001b; Eames *et al.* 2004). However, in 2003, several giant ibises were observed during surveys in Yok Don National Park (BLI–IP & MARD 2004; Eames *et al.* 2004; World Wide Fund for Nature (WWF) 2005). Located in Dok Lok Province in central Vietnam, the Park shares a western border with Cambodia. There is some speculation that the birds flew over the border from Cambodia (Mondulkiri Province) (WWF 2005), but this has not been confirmed or refuted.

Population Estimates

Population estimates are provided for the global population of giant ibis as well as for each range country. The range country estimates should not be considered distinct subpopulations. Very little is known about the species' ecology and dispersal, and all known areas where giant ibis have been observed are contiguous. There may be some interchange between populations and researchers have been unable to identify discrete subpopulations of this species (T. Clements in litt. December 2007).

Global population estimates: The giant ibis is characterized as uncommon and local throughout its range (Matheu & del Hoyo 1992; BLI 2000a). It occurs at relatively low densities and requires large areas of undisturbed habitat (deciduous dipterocarp forest and associated wetlands) (T. Clements in litt. December 2007). The majority of the giant ibis population today is located in Cambodia, with a small number in southern Lao PDR, even fewer in Vietnam, and no known individuals in Thailand (BLI 2000a, 2001b; www.rdb.or.id; Clements *et al.* 2007). The population has been conservatively estimated at a minimum of 100 pairs, with no more than 250 total individuals (Clements *et al.* 2007).

Cambodia: Population surveys have been conducted in several areas since the giant ibis' rediscovery in Cambodia in 2000. Aerial surveys between 2000 and 2001 indicated that between 50 birds and 90 were located in the Northern Plains (BLI–IP & MARD 2004). Based on the nest surveys conducted between 2004 and 2007 in Preah Vihear Protected Forest (PVPF) and Kulen Promtep Wildlife Sanctuary (KPWS), also in the Northern Plains, there was evidence of 28 nesting pairs of birds (Clements *et al.* 2007). Extrapolating to the available suitable habitat within the Northern Plains (including the Tonle Sap Lake), researchers estimated the population in the Northern Plains at 30 to 40 pairs. In the Eastern Plains (including the Siema Biodiversity Conservation Area (SBCA) and the Lomphat Wildlife Sanctuary), the population has been estimated at no more than 10 to 20 pairs. In northeastern Cambodia, Siem Pang (Stung Treng Province) surveys suggest that an excess of 14 pairs may exist. The total giant ibis population in Cambodia, based on available suitable habitat, is 82 to 100 pairs (Clements *et al.* 2007).

Lao PDR: The giant ibis Laotian population is estimated to include no more than 5 to 10 pairs of birds (Clements *et al.* 2007).

Vietnam: In 2003 and 2004, several giant ibises were observed during surveys in Yok Don National Park (Don Lok Province), the only known location within Vietnam (BLI–IP & MARD 2004; Eames *et al.* 2004; World Wide Fund for Nature (WWF) 2005). Yok Don National Park, which occupies a 446-mi² (1,155-km²) area, became a protected area in 1986 and a national park in 1991. The forest has three use areas: A 312-mi² (809-km²) strict protection area, a 117-mi² (3,043-km²) forest rehabilitation area, and a 16-mi² (42-km²) administration and services area. In addition, a 517-mi² (1,339-km²) buffer zone has been defined (Eames *et al.* 2004). However, these protections are ineffective at reducing or removing threats directed at the species (see Factor D).

Eames *et al.* (2004) postulated that the species is either very rare or a visitor in Vietnam. The Yok Don area is contiguous with sites in Cambodia (such as Eastern Mondulkiri) that are known to support resident breeding birds of giant ibises (T. Clements in litt. December 2007). During the re-evaluation of the species' status, experts concluded that Yok Don National Park is unlikely to support any breeding pairs (Clements *et al.* 2007). They considered that the birds observed within the Park were likely to be foraging or dispersing birds and that it was unlikely that the Park ''supported resident breeding birds due to the high level of disturbance and hunting'' (T. Clements in litt. December 2007).

Conservation Status

Global conservation status: Using the IUCN categories, the global population of giant ibis falls within the range of 50 to 250 individuals (BLI 2007h). The recent rediscovery of giant ibis in Vietnam and additional populations in Cambodia prompted BirdLife to re-evaluate the species' status in 2007 (Jez Bird, Global Species Programme Assistant, BirdLife International, in litt. November 2007; BirdLife Globally Threatened Species Forum 2007). They concluded that, despite recent new sightings of giant ibis in Vietnam and Cambodia, there was insufficient evidence to confirm that the giant ibis population exceeds 250 individuals (Clements *et al.* 2007; J. Bird in litt. November 2007).

The giant ibis has been categorized by the IUCN as a ''Critically Endangered'' since 1994 (BLI 2004c). BirdLife International, which serves as the IUCN Red List authority for birds, re-evaluated the status of the species in 2007 and decided to retain its critically endangered status for the 2008 Red List (J. Bird in litt. November 2007; Clements *et al.* 2007).

Cambodia: In 2005, the giant ibis was declared the national symbolic bird in Cambodia (Chheang Dany, Deputy Director, Wildlife Protection Office, Phnom Penh, Cambodia, in litt. January 2007) and, as of 2007, the species had been proposed as endangered in the draft wildlife list in Cambodia, the highest protected species category by the Forestry Law of 2002. However, this regulatory mechanism is ineffective at reducing or removing threats directed at the species (see Factor D).

Lao PDR: In Lao PDR, the giant ibis is legally protected and receives some habitat protection in the Xe Pian National Biodiversity Conservation Area (NBCA) (www.rdb.or.id; BLI 2001b). However, these regulatory mechanisms are ineffective at reducing or removing threats directed at the species (see Factor D).

Vietnam: In Vietnam, the species is listed as endangered (Eames *et al.* 2004). However, this regulatory mechanism is ineffective at reducing or removing threats directed at the species (see Factor D).

Summary of Factors Affecting the Giant Ibis

Where applicable in the sections below, factors affecting the survival of the giant ibises are discussed in two parts: (1) Regional factors (affecting or including two or more range countries), and (2) Factors within individual range countries.

A. The Present or Threatened Destruction, Modification, or Curtailment of the Species Habitat or Range

Giant ibis is affected throughout its range by (1) habitat modification from dam construction, (2) deforestation caused by war, (3) illegal logging and wood fuel collection, (4), and continued human encroachment (Bird *et al.* 2006; BLI 2007h; T. Clements in litt. December 2007; Clements *et al.* 2007; Poole 2002; WWF 2001, 2005).

(1) Habitat modification from dam construction: Dam construction along the Mekong River Basin (MRB) has altered giant ibis habitat throughout its range. The MRB begins as a system of tributaries and streams originating in the Tibetan Plateau and flowing eventually into the Mekong River Delta, 2,000 mi (4,800 km) from start to finish. Including parts of China, Myanmar and Vietnam, nearly one-third the land area of Thailand, and most of Cambodia and Lao PDR, the MRB encompasses a 307,000 mi^2 (795,000 km^2) area. The Lower Mekong River Basin (LMRB) includes Cambodia, Lao PDR, Thailand, and Vietnam (Mekong River Commission (MRC) 2007). According to the Asian Development Bank (ADB 2005), 13 dams are built, being built, or proposed to be built along the Mekong River Subregion. This important regional resource has a profound influence on each of the diverse ecosystems through which it flows, including giant ibis habitat. Two examples are discussed.

Construction of Yali Falls hydroelectric dam began in Vietnam in 1993 and was completed in 1999. The 226-ft (69-m) high dam was constructed at Yali Falls, on a tributary of the Sesan River. Part of the LMRB, the Sesan River originates in Vietnam and flows through Cambodia, where it meets the Mekong River. The Mekong River, in turn, flows into the Tonle Sap floodplain (Center for Natural Resources and Environmental Studies (CRES) 2001). The Tonle Sap floodplain, currently the southernmost extreme of the giant ibis' range in Cambodia, and freshwater swamp forest ecosystem rely on the Mekong River as part of its seasonal cycle of flooding (WWF 2001). A study of the impact of this dam on downstream communities in 2001 found that the effect of the dam on humans (including resettlement, drowning in unexpected floods, and livelihood changes especially for fishermen) would be ''significant but manageable,'' by relocating communities inland, for instance. The report also noted no anticipated impacts on waterbirds (CRES 2001). However, the study did not look beyond Vietnam and the effects of water flow disruption further downstream, including Tonle Sap floodplain in Cambodia. Within the first year of the dam's completion, massive devastating floods were reported downstream (CRES 2001).

Dam construction along the Srepok River, which flows through giant ibis habitat in Vietnam and Cambodia, has also altered the species' habitat. Construction of the Buon Koup Dam began in 2003 (San *et al.* 2007), altering the natural water and vegetation patterns along the Srepok River, affecting Yok Don National Park (Eames *et al.* 2004). A draft environmental impact analysis (EIA) identified several impacts to people living along the Cambodian side of the river, including daily irregular water fluctuations, erosion of riverbanks, and water pollution, as well as impacts on paddy production, fish migration, fishing livelihoods, and species diversity (San *et al.* 2007). In response to unpredictable water levels and flash flooding caused by dams, people began moving inland (ADB 2005).

Dam construction along the MRB has diverted water from critical ecosystems and has altered or threatens to alter the natural water and vegetation along waterways within the Mekong River Delta, a vital water source throughout the species' range. Impacts include drastic water level fluctuations, frequent flooding, and reduced water levels during the dry season, as well as the potential for riverbank erosion and increased water pollution. As populations move further inland to escape the unpredictable changes caused by dam construction, they encroach upon inland forested areas, including freshwater swamp ecosystems and semi-evergreen forests, which serve as giant ibis habitat (See (4) Continued human encroachment, below). The giant ibis is adverse to human disturbance (Bird *et al.*, 2006; www.rdb.or.id; BLI 2001b, 2007h; Dudley 2007; Eames *et* *al.*, 2004), and increased human disturbance exacerbates the impact of habitat modification caused by dam construction. See also (4) Continued human encroachment, below.

(2) Deforestation from war: The entire range of the giant ibis was severely affected by deforestation resulting from the Vietnam War (1959 to 1975). Bombing, herbicide spraying, and land-clearing activities were undertaken during the War. According to Westing (2002), 13.8 million U.S. tons (14 million metric tons) of high-explosive munitions were dropped by the United States throughout the region, including 5 percent in Cambodia, 16 percent in Lao PDR, 8 percent in northern Vietnam, and 71 percent in southern Vietnam, targeting primarily rural areas. Between 18 to 19 million gallons (gal) (68 to 72 million liters (l)) of herbicides (including Agent Orange contaminated with dioxin (see Factor E)) were sprayed on the region (Schechter *et al.*, 2001; Westing 2002). Of this amount, less than 0.1 percent was sprayed in Cambodia, 2 percent in Lao PDR, negligible amounts in northern Vietnam, and over 98 percent in southern Vietnam. Finally, 3 percent (1,255 mi^2 (3,250 km^2)) of the total forested area in South Vietnam was plowed over with tractors (Westing 2002). Inland forested areas, including freshwater swamp ecosystems and semi-evergreen forests, which serve as giant ibis habitat, were especially affected by herbicide applications during the war, where up to 77 percent of the total spraying occurred (Boi 2002). The most affected areas of bombing, spraying, and bulldozing correspond with the historic range of the giant ibis, where the species went unobserved until 1993, and the figures for southern Vietnam are particularly informative, where the species remains unobserved to this day (www.rdb.or.id; BLI 2001c).

(3) Illegal logging and wood fuel collection: The open and deciduous forested wetland habitats preferred by the giant ibis species have diminished over much of Indochina, and only Cambodia retains significant portions of this habitat (WWF 2005). Deforestation from illegal logging and wood fuel collection has reduced the number of nesting sites available to the species (BLI 2007h; Poole 2002). In addition, it led to increased habitat disturbance (see (4), Continued human encroachment).

Cambodia: Poole (2002) reported that large nesting trees around Cambodia's Tonle Sap floodplain, particularly crucial to ibises for nesting, are under increasing pressure by felling for firewood and building material. Illegal logging has been reported in Trapeang Boeung (Global Witness 2007), where the giant ibis was observed in 2003 (www.rdb.or.id; BLI 2001b), and in the SBCA, where the species was observed in 2006 (Bird *et al.*, 2006).

Lao PDR: Logging has been reported in the Xe Pian National Biodiversity Conservation Area (NBCA), where the giant ibis has been observed, perhaps as a seasonal visitor (Robichaud *et al.*, 2001).

In Vietnam: Deforestation in Vietnam has been significant throughout the 20th century. In 1943, approximately 43 percent of the total land area in Vietnam was covered by natural forest. This corresponded to 54,054 mi^2 (140,000 km^2). By 1945, 22,007 mi^2 (57,000 km^2) of natural forest had been cleared (Brown *et al.*, 2001). By 1990, the total forested area had been reduced to 27 percent, nearly half the amount of 1943 (Boi 2002).

Logging bans in Vietnam became progressively more pervasive in the 1990s. In 1992, logging in watershed and special-use forests was banned. In 1999, all commercial logging in natural forests in the northern highlands and midlands, the southeast, and in the Mekong River and Red River Delta Provinces was banned. As of 2001, 58 percent of Vietnam's natural forests were covered by the ban (Brown *et al.*, 2001). (See Factor D.)

The government planned to obtain its wood needs from plantation forests (Brown *et al.*, 2001). In 1999, the total forested area had increased to 33 percent, corresponding to 36,464 mi^2 (94,440 km^2). This figure included 5,680 mi^2 (14,710 km^2) of plantation forest, only 1 percent of which represented deciduous forest (Boi 2002). The increase in plantations forests led to changes in species composition.

Changes in species composition led to changes in the amount of forest cover. Following the Food and Agriculture Organization's (FAO) classifications for forest cover, Cuong (1999) determined from remote sensing data that, between 1943 and 1995, forest cover in Vietnam transformed from 43 percent cover (which considered to be medium forest cover by FAO), to 28 percent (which FAO considers to be open forest).

(4) Continued human encroachment: Habitat alteration from dam construction and destruction caused by war are compounded by human encroachment throughout the species' range (see also (2), Factors within individual range countries, below).

Cambodia: In Cambodia's Tonle Sap floodplain, the effects of dam construction are exacerbated by agricultural conversion (Eames *et al.* 2004). Tonle Sap floodplain is considered "prime rice-growing habitat" (WWF 2001, p. 1). Extensive cultivation during the dry season and the impacts from fishing communities along the delta, disrupt the natural water cycle, resulting in drastic water level fluctuations within the Mekong River Delta, with frequent flooding and lower water levels during the dry season (WWF 2001).

The buffer zone of Cambodia's Seima Biodiversity Conservation Area (SBCA) (Kratie and Mondulkiri Province), where giant ibis was observed in 2006 (Bird *et al.* 2006), is threatened by a variety of human activities, including road building, increased subsistence activities, and collection of non-timber forest products (Bird *et al.* 2006; WCS 2007b). Resin tapping is common throughout the SBCA, and the concomitant increase in the number of people entering the SBCA to undertake this and other extractive activities poses an additional threat to the giant ibis (Bird *et al.* 2006), which is highly sensitive to human disturbance (Bird *et al.* 2006; www.rdb.or.id; BLI 2001b, 2007h; T. Clements in litt. December 2007; Clements *et al.* 2007; Dudley 2007; Eames *et al.* 2004).

Lao PDR: Robichaud *et al.* (2001) identified the following ongoing internal and external threats to giant ibis habitat in the Xe Pian National Biodiversity Conservation Area (NBCA): (1) Subsistence agriculture, (2) subsistence hunting, (3) trade hunting, (4) subsistence fishing, (5) trade fishing, (6) free-ranging livestock, (7) road construction, and (8) infrastructure development.

Vietnam: Giant ibis habitat in Vietnam's Yok Don National Park is threatened by road building, road improvements, and artificial waterhole creation on sites of natural "*trapeangs*" (seasonal and permanent waterholes). Giant mimosa (*Mimosa pigra*) has spread rapidly along the Srepok River since the 1980s (Eames *et al.* 2004). Giant mimosa is an aggressively invasive plant that forms dense thickets, closing formerly open habitats and outcompeting native species (WWF 2001).

The giant ibis requires large areas of undisturbed habitat and is known to be highly sensitive to human disturbance (Bird *et al.* 2006; www.rdb.or.id; BLI 2001b, 2007h; T. Clements in litt. December 2007; Clements *et al.* 2007; Dudley 2007; Eames *et al.* 2004). In the nesting surveys conducted between 2004 and 2007, researchers found that the most nests were located more than 3 mi (5 km) from villages (Clements *et al.* 2007). Bird *et al.* (2006) studied the effect of habitat disturbance on several large waterbirds, including the giant ibis. They found that the giant ibis was significantly less likely to visit watering holes that were frequented by humans. The majority of the species breeds in remote areas and uses wetlands that have minimal human presence (T. Clements in litt. December 2007).

Habitat fragmentation caused by loss of habitat is compounded by human disturbance and is likely to have a disproportionate effect on the remaining individuals (Clements *et al.* 2007). According to Clements (in litt. December 2007), continuing expansion of human settlements and wetland manipulation are likely to cause strong declines over time, even if deforestation rates are low.

Summary of Factor A

Giant ibis habitat has been destroyed and degraded throughout the core of its range, and habitat reduction or modification continues to be a significant factor endangering the species. The giant ibis is a waterbird that seeks out permanent sources of water, and the impacts from habitat destruction and alteration are exacerbated by its aversion to human disturbance. Dam construction has contributed to habitat alteration on a regional scale along waterways within the Mekong River Delta (a vital water source throughout the species' core range) and contributes to unpredictable water fluctuations and changes in human activity along the waterways. The effects of flooding are exacerbated by extensive cultivation during the dry season and the impacts from fishing communities along the delta. Habitat loss through wetland drainage for agricultural purposes has reduced foraging and roosting areas. Logging has been reported in giant ibis territory in each range country, and deforestation reduces the number of trees available to the species as nesting sites. Expansion of human settlements and conversion of wetland areas to agriculture continue throughout the species' known range. The encroachment of nesting sites and foraging areas is compounded by human disturbance and may disproportionately promote fragmentation of remaining individuals. Based on the above information, we find that the present or threatened destruction, modification, or curtailment of the giant ibis' habitat or range is a significant on-going and future risk to the species.

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

(1) Overutilization within the region: The giant ibis is susceptible to hunting for consumption and disturbance caused by hunting other species throughout its range (Bird *et al.* 2006; www.rdb.or.id; BLI 2001b, 2007h; T. Clements in litt. December 2007; Desai & Luthy 1996; Eames *et al.* 2004; Poole 2002; WCS 2007a, 2007b, 2007c). There have been reports of severe hunting pressures on large mammals and waterbirds, including giant the ibis, throughout the species' range (ADB 2005; T. Clements in litt. December 2007; Desai & Luthy 1996; Poole 2002; United Nations Environment Programme-Strategic Environment Framework (UNEP–SEF) 2005; WCS 2007a, 2007b, 2007c). In 2005, the United Nations Environment Programme-Strategic Environment Framework (UNEP–SEF 2005) reviewed major threats to biodiversity, including giant ibis, within the Greater Mekong Sub-region (including Cambodia, Lao PDR, Myanmar, Thailand, and Vietnam). They found that, after habitat loss, the second greatest threat to endangered wildlife in the region was hunting and gathering. Giant ibises are particularly vulnerable to hunting during the dry season, when they seek out permanent water sources and are more likely to encounter people seeking out these same water resources (BLI 2007h).

Given the species' small estimated global population size (a minimum of 100 pairs, but no more than 250 total individuals (Clements *et al.* 2007)), any hunting would be detrimental to the species' continued existence. Highly sensitive to human disturbance, giant ibises are negatively affected by disturbance from hunting-related activities, even when they are not directly targeted (T. Clements in litt. December 2007).

(2) Overutilization within individual range countries:

Cambodia: Cambodia is the core of the species' range, where the total Cambodian giant ibis population is estimated to be 82 to 100 pairs (Clements *et al.* 2007). Subsistence hunting is a challenge to wildlife protection in Cambodia, where the average annual income is US$268 and ''95 percent of the country lives from tree cutting and wildlife hunting'' (WildAid 2002, p. 1). According to Clements (in litt. December 2007), in surveys conducted over the past eight years, there have been occasional reports of giant ibis being hunted for personal or commercial use in Cambodia, but ''it [giant ibis] appears to have little value wildlife trade.'' In the past 5 years, Clements (in litt. December 2007) is aware of two instances of giant ibis hunting, both for personal consumption. In addition, locals poison waterholes, using commonly available herbicides, fertilizers, or insecticides, to hunt fish and sometimes to poison large waterbirds for consumption (T. Clements in litt. December 2007).

Poole (2002) noted that bird species in Cambodia are generally susceptible to indiscriminate hunting and egg collection. A 1996 wildlife survey of three sites within Mondulkiri and Rattanakiri Provinces, where Lomphat Wildlife Sanctuary is located and wherein the giant ibises have been observed, revealed that hunting was extensive and intense (Desai & Vuthy 1996). The Wildlife Conservation Society reported hunting as the single largest threat to wildlife in the Northern Plains (WCS 2007a). Subsistence and commercial hunting of a variety of animals has been reported in within the SBCA as recently as February 2006 (Bird *et al.* 2006; WCS 2007b), and collection of eggs and chicks from nests threaten large waterbirds in the Tonle Sap floodplain (Clements *et al.* 2007; WCS 2007a, 2007b, 2007c). See also Factor D.

Lao PDR: BirdLife International (2006a) reports that hunting in Lao PDR has severely impacted most large waterbirds. While we have no information that the giant ibis is specifically targeted, this practice would severely threaten the species in Lao PDR, where the giant ibis population is unlikely to exceed 5 to 10 pairs (Clements *et al.* 2007).

Vietnam: Large mammals and waterbirds are particularly vulnerable to hunting within Yok Don National Park, the only location within Vietnam where giant ibis has been observed (Eames *et al.* 2004), and wildlife hunting continued to be a problem within the Yok Don National Park in 2005 (Eames *et al.* 2005) (see also Factor D). The U.S. Department of State (DOS) reported that Vietnam's wildlife, including endangered birds, is threatened by illegal export to China (DOS Cable 2007). However, we have no specific information that the giant ibis is part of such trade. The species is not known to be in international trade and has not been formally considered for listing under CITES (www.cites.org).

Summary of Factor B

Indiscriminate hunting threatens giant ibis throughout its range. Giant ibises are especially accessible and more vulnerable to hunting at the height of the dry season when they are concentrated around available waterholes. The species' aversion to human disturbance makes it more vulnerable to disruption from hunting-related activities. Given their small population numbers (estimated to be

100 pairs at minimum, but no more than 250 individuals) and the apparent inadequacies in enforcement (Factor D), we consider incidental killing from hunting and hunting disturbances to be factors that threaten this species throughout its range.

C. Disease or Predation

According to the Deputy Director of the Wildlife Protection Office in Cambodia (C. Dany in litt. January 2007), highly pathogenic avian influenza (HPAI) H5N1 continues to be a serious problem. This strain of avian influenza first appeared in Asia in 1996 and spread from country to country with rapid succession (Peterson *et al.* 2007). By 2006, the virus was detected across most of Europe and in several African countries. Influenza A viruses, to which group strain H5N1 belongs, infect domestic animals and humans, but wildfowl and shorebirds are considered the primary source of this virus in nature (Olsen *et al.* 2006), particularly wild birds of wetland and aquatic environments (Peterson *et al.* 2007). Although the Wildlife Protection Office noted that the U.S. Department of Agriculture Animal and Plant Health Inspection Service were helping train field staff on surveillance techniques, Cambodia lacks an avian influenza wild bird surveillance program (C. Dany in litt. January 2007). According to Dany (in litt. January, November 2007), scientists are not sure how many wild bird species carry or are infected by AI, and it is possible that giant ibis may be a carrier. However, a comprehensive study has not yet been undertaken. Lack of an avian influenza wild bird surveillance program in Cambodia will make it difficult to resolve whether giant ibis is a carrier.

Until recently, there was no information on predation affecting the giant ibis, and there is still very little known about giant ibis breeding ecology and dispersal (T. Clements in litt. December 2007). However, recent research suggests that predation impacts the largest known concentration of giant ibises in Cambodia's Northern Plains (estimated to be 30 to 40 pairs of birds), representing between one-third to one-fourth of the total known population (Clements *et al.* 2007). Nesting surveys were conducted between 2004 and 2007, and the giant ibis' fledging success was estimated at 50 percent. Researchers determined that predation had negatively impacted the giant ibis' fledging success. Predation by crows (*Corvus macrorhynchos*), macaques (Macaca sp.), hawks (species unknown), civets (Cynogale sp), and martins (species unknown) was identified as a major contributor to the species' low fledging success (Clements *et al.* 2007). Given the species' small global population size and that the Northern Plains species may represent up to one-fourth of the known giant ibis population, we consider this level of predation to be a significant factor that threatens the species' continued existence.

Summary of Factor C

While the avian flu may be a threat to giant ibises, there is no evidence that known populations are currently infected. Potential for disease outbreaks warrants monitoring (see Factor D) and may become a more significant threat factor in the future. However, we find that disease is not a risk to the giant ibis at this time.

Predation by crows, macaques, hawks, civets, and martins threatens the largest known concentration of giant ibises and contributes to the species' low fledging success (estimated to be only 50 percent). Given the risks associated with small population sizes, further reductions in population numbers jeopardizes the species' viability and resiliency to adapt to changing conditions (see Factor E). We consider predation to be a factor that endangers the species.

D. The Inadequacy of Existing Regulatory Mechanisms

(1) Regional regulatory mechanisms:

The Mekong River Commission (MRC) was formed between the governments of Cambodia, Lao PDR, Thailand, and Vietnam in 1995 as part of The Agreement on the Cooperation for the Sustainable Development of the Mekong River Basin. The signatories agreed to jointly manage their shared water resources and economic development of the river (MRC 2007). In 2003, the governments of Cambodia, China, Lao PDR, Myanmar, Thailand, and Vietnam committed to cooperate on developing a regional power grid (via hydroelectric dams), among other things, under the Asian Development Bank's Greater Mekong Subregion Program (International Rivers Network. 2004). However, according to the International Rivers Network (2004), the master plan to create the regional power grid did not thoroughly assess the impacts to communities, fisheries, Forests or nature reserves. The cooperative efforts have had little impact on the dams being built in the Mekong River Region or on broader decision-making processes within the Region (CRES 2001). According to the Asian Development Bank, 13 dams have been built, are being built, or are proposed to be built along the Mekong River Subregion (ADB 2005). The continued modification of giant ibis habitat has been identified as a primary threat to this species (Factor A), and this regional regulatory mechanism is not effective at reducing that threat.

(2) Regulatory mechanisms within individual range countries:

Cambodia: Several laws exist in Cambodia to protect the giant ibis from two of the primary threats to the species, habitat destruction (Factor A) and hunting (Factor B). However, they are ineffective at reducing those threats. In Cambodia, Declaration No. 359, issued by the Ministry of Agriculture, Forestry and Fisheries in 1994, prohibited the hunting of giant ibis. However, reports of severe hunting pressure within the giant ibis' habitat and illegal poaching of wildlife in Cambodia continue (Bird *et al.* 2006; Desai & Luthy 1996; FFI 2000; Poole 2002; UNEP–SEF 2005; WCS 2007a, 2007b, 2007c).

Joint Declaration No. 1563, On the Suppression of Wildlife Destruction in the Kingdom of Cambodia, was issued by the Ministry of Agriculture, Forestry and Fisheries in 1996. However, JICA (1999) reported that this regulatory measure was ineffectively enforced. In 2000, survey work conducted by Fauna and Flora International in collaboration with the Government of Cambodia, Ministry of Environment and Wildlife Protection Office, found evidence of illegal hunting of a variety of animals and noted a flagrant disregard for the illegality of this activity: ''Hunters and dealers freely displayed the illegal materials and readily provided any details requested,'' indicating a lack of wildlife laws awareness or inadequate law enforcement (FFI 2000).

The Forestry Law of 2002 strictly prohibited hunting, harming, or harassing wildlife (Article 49) (Law on Forestry 2003). This law further prohibited the possession, trapping, transport, or trade in rare and endangered wildlife (Article 49). As of 2007, Dany (in litt. January 2007) noted that the species had been proposed as endangered in the draft wildlife list in Cambodia, the highest protected species category by Forestry Law 2002 (Law on Forestry 2003). However, to our knowledge, Cambodia has not yet published a list of endangered or rare species. Thus, this law is not currently effective at protecting the giant ibis from threats by hunting (Factor B).

The Creation and Designation of Protected Areas regulation (November 1993) established a national system of protected areas. In 1994, through Declaration No. 1033 on the Protection of Natural Areas, the following activities were banned in all protected areas: (1) Construction of saw mills, charcoal ovens, brick kilns, tile kilns, limestone ovens, tobacco ovens; (2) hunting or placement of traps for tusks, bones, feathers, horns, leather, or blood; (3) deforestation; (4) mining minerals or use of explosives; (5) use of domestic animals, such as dogs; (6) dumping of pollutants; (7) the use of machines or heavy cars which may cause smoke pollution; (8) noise pollution; and (9) unpermitted research and experiments. In addition, the Law on Environmental Protection and Natural Resource Management of 1996 (Law on Environmental Protection and Natural Resource Management 1996) sets forth general provisions for environmental protection. Under Article 8 of this law, Cambodia declares that its natural resources (including wildlife) shall be conserved, developed, and managed and used in a rational and sustainable manner. Several protected areas have been established within the range of the giant ibis, including the Tonle Sap Great Lake Protected Area, Seima Biodiversity Conservation Area, and Lomphat Wildlife Sanctuary.

The Tonle Sap Great Lake protected area was designated a Multiple Use Management Area in 1993 through the Creation and Designation of Protected Areas Decree (Creation and Designation of Protected Areas 1993). Under this decree, Multiple Use Management Areas are those areas which provide for the sustainable use of water resources, timber, wildlife, fish, pasture and recreation with the conservation of nature primarily oriented to support these economic activities. In 1997, the Tonle Sap region was designated a UNESCO ''Man and Biosphere'' site. To echo the United Nations designation, the Cambodian government developed a National Environmental Action Plan (NEAP) in 1997, supporting the UNESCO site goals. Among the priority areas of intervention are fisheries and floodplain agriculture at Tonle Sap Lake, biodiversity and protected areas, and environmental education. NEAP was followed by the adoption of the Strategy and Action Plan for the Protection of Tonle Sap (SAPPTS) in February 1998, and the issuance of a Royal Decree officially making Tonle Sap Lake a Biosphere Reserve on April 10, 2001 (Tonle Sap Biosphere Reserve Secretariat 2007). In 2006, the Cambodian government created Integrated Farming and Biodiversity Areas (IFBA), including 115 mi^2 (300 km^2) near Tonle Sap Lake, to protect the distinctive flora in that region (WWF 2006a). The above measures have focused attention on the conservation situation at Tonle Sap and have begun to improve the conservation situation there, but several management challenges remain, including overexploitation of flooded forests and fisheries; negative impacts from invasive species; lack of monitoring and enforcement; low level of public awareness of biodiversity values; and uncoordinated research, monitoring, and evaluation of species' populations (Matsui *et al.* 2006; Tonle Sap Biosphere Reserve Secretariat 2007).

The Seima Biodiversity Conservation Area was established through Declaration 260.12–08–2002 (On the Establishment of Seima Biodiversity Conservation Area in Samling Forest Concession in Mondul Kiri and Kratie Provinces). However, threats at this site remain. Lack of clear land and resource tenure within the buffer zone of Seima Biodiversity Conservation Area (SBCA) (Kratie and Mondulkiri Province), where giant ibises were observed in 2006 (Bird *et al.* 2006), has resulted in influxes of squatters interested in claiming, cutting, or clearing the land (WCS 2007b). In early 2006, during surveys of the Seima Biodiversity Conservation Area (SBCA), where giant ibis is located, researchers encountered hunters ''with no law enforcement in operation'' (Bird *et al.* 2006, p. v).

The Lomphat Wildlife Sanctuary, where the giant ibis is also found, was established in 1993 through the Creation and Designation of Protected Areas Decree (Creation and Designation of Protected Areas 1993) and is considered to be one of the most important areas for wildlife in Cambodia (WildAid 2005). Under this decree wildlife sanctuaries are considered natural areas where nationally significant species of flora and fauna, natural communities, or physical features require specific intervention for their perpetuation (Creation and Designation of Protected Areas 1993). In 2003 and 2004, the Service's Rhino and Tiger Conservation Fund supported the Lomphat Conservation Project (LCP), which has a long-term goal of assisting rangers and field staff in the conservation of the Sanctuary's living resources, including giant ibis. Six teams of rangers were trained during the duration of the LCP, and the Sanctuary began instituting patrols on at least 15 days per month. The rangers have been extremely efficient in locating poachers, illegal loggers, and entire camps set aside for poachers. Educational materials were developed and tailored to the villagers' patterns of use of the local resources (WildAid 2003), and villagers have demonstrated a keen interest in offering information to protect their resources and assist the rangers. Extensive public outreach has improved conservation awareness throughout the Sanctuary and around its borders (WildAid 2005). Project leaders for the Lomphat Conservation Project indicated that great strides have been made in training rangers and combating poaching, although community outreach required more effort (WildAid 2005). In 2005, the giant ibis was declared the national symbolic bird in Cambodia (C. Dany in litt. January 2007), which may help to raise public awareness as to the need to conserve the species and its habitat.

Giant ibis habitat within Cambodian protected areas faces several challenges. The legal framework governing wetlands management is institutionally complex, resting upon legislation vested in government agencies responsible for resource use (Fishery Law 1987), land use planning (Land Law 2001), and environmental conservation (Environmental Law 1996, Royal Decree on the Designation and Creation of National Protected Areas System 1993) (Bonheur *et al.* 2005). Furthermore, the country's wildlife protection office lacks the staff, technical ability and monetary support to conduct systematic surveys on the giant ibis (C. Dany in litt. January 2007). This, in turn, leads to ineffective monitoring and enforcement, and, consequently, resource use goes largely unregulated (Bonheur *et al.* 2005). Thus, the protected areas system in Cambodia is ineffective in removing or reducing the threats of habitat modification (Factor A) and hunting (Factor B) faced by the giant ibis.

Lao PDR: Giant ibis is legally protected in Lao PDR (Eames *et al.* 2004). In Lao PDR, the giant ibis is found in one protected area, the Xe Pian National Biodiversity Conservation Areas (NBCA). Regulation No. 0524/MAF.2001, on NBCAs and wildlife management, was issued by the Ministry of Agriculture and Forestry on June 7, 2001 (Robichaud *et al.* 2001). This regulation is a comprehensive code of wildlife protection. Penalties for violation of the existing decrees and instructions are outlined in the Penal Code of the Lao PDR (October 23, 1989) and refined in the Instructions for the Implementation of Decree No. 118 and in the Forestry Law of 1996.

Xe Pian NBCA was established in 1993 as part of the system of National Protected Areas. Long-term biodiversity conservation is the primary objective of NBCAs, according to PM Decree 164 and the 1996 Forestry Law. While the establishment of this protected area represents a positive step toward conserving habitat in Xe Pian, the protection afforded giant ibis in the Xe Pian NBCA is marginal to ineffective due to confusion over management authority and lack of enforcement (www.rdb.or.id; BLI 2001c, 2001d; Rauchibauld *et al.* 2001). Furthermore, the existence of an NBCA does not rule out construction of hydroelectric dams, or commercial activities such as logging (www.rdb.or.id; BLI 2001d), identified as threats to this species (Factor A).

Thailand: The species is currently considered extirpated from Thailand. However, giant ibis is protected by the Wildlife Animal Reservation and Protection Act (WARPA) (B.E. 2535 1992; Eames *et al.* 2004). Under WARPA, hunting is prohibited (section 16), as is possession of carcasses (section 19), trade (section 20), and collection, harm or possession of nests (section 21). Violations of sections 16, 19, or 20 of WARPS may result in imprisonment not exceeding four years or fines nor exceeding 40,000 baht (Thai dollars), or both. Violations of section 21 of WARPA may result in imprisonment not exceeding one year or fines not exceeding 6,000 baht. This protection may help to remove the threat of hunting, which affect the species throughout its existing current range (Factor B), but does nothing to remove or reduce the threat to habitat reduction (Factor A), which was attributed as the primary cause for the species' extinction in Thailand (www.rdb.or.id; BLI 2001b; Matheu & del Hoyo 1992).

Vietnam: Decree No. 32/2006/ND–CP of March 30, 2006, on Management of Endangered, Precious, and Rare Forest Plants and Animals, establishes a list of endangered species and protections afforded to those species (Decree No. 32 2006). However, the giant ibis is not on that list (Official Dispatch No. 3399 2002) and therefore is not afforded any legal protection under this Decree.

Vietnam banned hunting without a permit in 1975 (Zeller 2006). However, the Department of State (DOS Cable 2007) reports that Vietnam's wildlife, including birds, continues to be susceptible to domestic consumption.

Yok Don National Park was established by Decree in 2002 (International Centre for Environmental Management (ICEM) 2003). Under Vietnam's Law on Forest Protection and Development of 2004 (No. 25 2004), National Parks are considered special use forests, which are used mainly for conservation of nature, preservation of national forest ecosystems, and biological gene resources; scientific research; protection of historical and cultural relics as well as landscapes; in service of recreation and tourism. The Law on Forest Protection and Development prohibits, among other things: (1) Unpermitted logging; (2) unpermitted hunting, shooting, capture, caging, or slaughter of forest animals; (3) illegally destroying forest resources or ecosystems; (4) violating regulation on forest fire prevention; (5) violating regulations on prevention and elimination of organisms harmful to forests; (6) illegal encroachment; (7) illegal possession, transport, or trade in forest plants and animals; (8) illegally grazing cattle in strictly-protected zones of special use forests; (9) illegally exerting adverse impacts on wildlife; and (10) illegally bringing toxic chemicals or explosives into forests (Article 12). However, the Yok Don National Park apparently lacks specific regulations governing activities within the Park (Eames *et al.* 2004), and it is unclear what tangible protections, if any, are afforded the species in this area. Furthermore, there are continued external threats to the biological resources in the park (*e.g.*, the proposed Ea Tung dam) (ICEM 2003) (Factor A) and hunting (Factor B). Eames *et al.* (2005) reported that hunting was a problem for wildlife within the Yok Don National Park. Thus, the measures in place are ineffective at reducing the threats to this species.

Summary of Factor D

Existing regulatory mechanisms throughout the giant ibis' range are ineffective at reducing or removing threats directed at the species, including habitat modification (Factor A) and hunting (Factor B). We believe that the inadequacy of regulatory mechanisms, especially with regard to lack of law enforcement and habitat protection, is a contributory risk factor for the giant ibis.

E. Other Natural or Manmade Factors Affecting the Continued Existence of the Species

Other factors which affect the giant ibis' continued existence are: its small population size and environmental toxins.

Small population size: Small, isolated populations of wildlife species are susceptible to demographic shifts and genetic problems (Shaffer 1981). These threat factors, which may act in concert, include natural variation in survival and reproductive success of individuals, chance disequilibrium of sex ratios, changes in gene frequencies due to genetic drift, and diminished genetic diversity and associated effects due to inbreeding. Demographic problems may include reduced reproductive success of individuals and chance disequilibrium of sex ratios.

We are unaware of any genetic studies for the giant ibis. However, threats to near- and long-term genetic viability can be estimated. In the absence of more species-specific life history data, the 50/500 rule (as explained under Factor E for the black stilt) (Soulé 1980; Hunter 1996) may be used to approximate minimum viable population sizes, as described under Factor E for the black stilt. The available information indicates that the largest concentration of giant ibis consists of 30 to 40 pairs (Clements *et al.* 2007). This would equate to 60 to 80 individuals, which just meets the minimum effective population size (N_e = 50 individuals) required to avoid risks from inbreeding. The current maximum estimate of no more than 250 individuals for the entire population (Clements *et al.* 2007) is only half of the upper threshold (N_e = 500) required for long-term fitness of a population that will not lose its genetic diversity over time and that will maintain an enhanced capacity to adapt to changing conditions. As such, we currently consider the species to be at risk of long-term genetic viability and associated demographic problems.

Environmental toxins: Environmental toxins likely pose a threat to the giant ibis, given its foraging habit and diet. Agent Orange was one of the primary defoliants sprayed during the Vietnam War (Westing 2002). One of the formulations (2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD)) released dioxin as a byproduct as it broke down. Dioxin is a known human carcinogen. Studies conducted following the war through the mid-1990s found that residents of southern Vietnam contained extremely high levels of dioxin found in fluid or tissue samples, including mother's milk and food fish. Sediment studies in the 1980s indicated that dioxin can move through soil into lakes or rivers, where it attaches to organic material in the sediment. In 1995, tissue sample studies revealed that even residents in areas that were not sprayed by Agent Orange (in northern Vietnam) contained low levels of TCDD contamination. In 2001, high levels of dioxin were still being detected in residents in southern Vietnam 30 years after TCDD was sprayed. Residents born subsequent to spraying and newly arrived residents had similarly high levels of dioxin in their systems. The authors concluded that it is highly probable that current dioxin contamination detected in humans is the result of past and current exposure to dioxin that has moved from the soil into river sediments, into fish, and subsequently into people from fish consumption (Schechter *et al.* 2001). The giant ibis forages in mud flats, probing the mud with their bills. With evidence that dioxin contamination in soils persists more than 30 years after the Vietnam War, it is likely that the giant ibis is being exposed to this contaminant.

According to Gatehouse (2004), when fish, birds, or mammals are exposed from conception through postnatal or post hatching stages, dioxins may disrupt development of several major organ systems (including the endocrine, reproductive, immune and nervous systems). Dioxins are potent developmental toxicants even at low concentrations, and effects of dioxin poisoning in birds include poor breeding success, embryo lethality, and developmental deformities (Gatehouse 2004). Although we are unaware of any studies of the effect of environmental contaminants on the giant ibis, this may be a factor in the species' low fledging success (estimated to be 50 percent (Clements *et al.* 2007)).

Birds may be exposed to dioxins in their food or by foraging in contaminated soil (Gatehouse 2004). Animals vary in their sensitivity to dioxin (Karchner *et al.* 2006) and levels of contamination vary relative to their trophic level (position in the food chain) (Gatehouse 2004). Giant ibis consumes primarily invertebrates, small reptiles, and amphibians (www.rdb.or.id; BLI 2001b, 2007h; Davidson *et al.* 2002). According to Gatehouse (2004), other bird species at this mid-trophic level accumulate dioxin contamination at a low to midrange (where birds of prey have the highest levels of contamination). Dioxin poisoning is known to affect reptiles, resulting in development abnormalities (Shirose *et al.* 1995). Residual contamination in the tissues of prey species may remain long after contaminant concentrations are reduced (Gatehouse 2004). Given that giant ibis is a mid-trophic level species, which are known to accumulate dioxin at low-to mid-range levels, and that reptiles, a food source for giant ibis, are known to retain residual dioxin within their tissues, it is likely that the giant ibis is being exposed to dioxin through its prey species as well.

Summary of Factor E

The giant ibis' small population, estimated to be at least 100 pairs, but no more than 250 total individuals, poses a risk to the species throughout its range with regard to lack of near-term long-term genetic viability and to potential demographic shifts. We consider the species' extremely small population size and associated lack of genetic viability and threats of demographic shifts to be significant risks to the giant ibis throughout its range.

Dioxin contamination likely poses a threat to the giant ibis, given its foraging habits of eating along mud flats and probing the mud with its bill and the fact that dioxin contamination remains in the soil more than 30 years later. Diet may also expose giant ibises to dioxin accumulated in the tissue of prey species. Although we believe that dioxin contamination could be a factor contributing to the decline of the giant ibis, there has been no direct research into the effects of dioxin on giant ibis. As such, insufficient information precludes our ability to determine whether dioxin contamination endangers the species.

Conclusion and Determination for the Giant ibis

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the giant ibis. We have determined that the species is in danger of extinction throughout all of its known range primarily due to ongoing threats to its habitat (Factor A), unregulated hunting (Factor B), and genetic and demographic risks associated with the species' small population size and habitat fragmentation (Factor E). Predation threatens the largest known concentration of giant ibis in the Northern Plains of Cambodia (Factor C). Furthermore, we have determined that the inadequacy of regulatory mechanisms to reduce or remove these threats is a contributory factor to the risks that endanger this species' continued existence (Factor D). Therefore, we are determining endangered status for the giant ibis under the Act. Because we find that the giant ibis is endangered throughout all of its range, there is no reason to consider its status in any significant portion of its range.

IV. Gurney's pitta (*Pitta gurneyi*)

Species Description

The Gurney's pitta is a member of the Pittidae family and is native to Myanmar and Thailand. The species is also known commonly as the black-breasted pitta (www.rdb.or.id; BLI 2001c) and the jewel-thrush (BLI-IP & Biodiversity and Nature Conservation Association (BANCA) Darwin Project Office 2004). Adults are between 7 and 8 in (18 and 20 cm) tall. The male has a blue crown and a turquoise-tinged tail. Black plumage covers the breast, with brown on the upper side, and black and yellow bands along the sides of the underbelly. The female has a brown crown and paler light-brown and buff (or black and yellow) banding on the underparts. The juvenile is draped in brown plumage on the crown, nape, and breast, with pale streaks on the upper belly and white speckles on the wings (BLI 2007g; Gould 1969; Thailand Scientific Authority 1990).

Taxonomy

Gurney's pitta, in the family Pittidae, was described by Hume as *Pitta gurneyi* in 1875 (BLI 2005) from a specimen obtained in Myanmar.

Habitat and Life History

This species' habitat requirements of this species were poorly understood until surveys were conducted in the 1980s (see Population Estimates, below). Gurney's pitta inhabits lowland, semi-evergreen secondary rainforest, at elevations from 260 to 460 ft (80 to 140 m). They are especially found at elevations less than 328 ft (100 m), in areas with little to no undergrowth (BLI 2000b, 2001c; Gould 1969). Access to permanent sources of water is a central feature of Gurney's pitta habitat, such that populations are often located near gully systems where moist conditions remain year-round (BLI 2000b, 2001c).

Gurney's pitta has been described as a ''relatively silent species'' (Rose 2003, p. 142); although more audible during mating season, and the species occurs more often in the mornings and evenings (www.rdb.or.id; BLI 2001c; Gould 1969). The species rarely ventures into open areas (www.rdb.or.id; BLI 2001c) and does not live in groups (Thai Society for the Conservation of Wild Animals (TSCWA) no date (n.d.)). A terrestrial bird, Gurney's pitta hops around the forest floor on its strong hind legs to forage on insects, snails, and especially earthworms (www.rdb.or.id; BLI 2001c; Kekule 2005; TSCWA n.d.).

Apparently monogamous (www.rdb.or.id; BLI 2001c), the species breeds during the monsoon season from April to October (www.rdb.or.id; BLI 2001c, 2007g). Dome-shaped nests with a single opening are built approximately 3.3 to 8.2 ft (1 to 2.5 m) off the ground in spiny understory palms, including rakum (*Salacca rumphii* or *Salacca wallichiana*), rattan (*Daemonorops* or *Calamu longisetus*), and licuala palms (*Licuala* spp.) (BLI 2001c, 2003b; Kekule 2005; Rose 2003; TSCWA n.d.). Eggs are cream-colored with brown flecks, the typical clutch size is 3 to 4, and eggs are incubated by both males and females for as few as 10 and up to 20 days (www.rdb.or.id; BLI 2001c; Rose 2003; TSCWA n.d.). In captivity, pairs nested twice in 1 year (www.rdb.or.id; BLI 2001c). Gurney's pitta apparently has a low rate of breeding success, with an average production of one (Lambert 1996 as cited in BLI 2001c), two, or, at most, three chicks (Kekule 2005) fledged per clutch. In the only nest monitoring study, three giant ibis nests achieved an overall fledging rate of 27.3 percent (www.rdb.or.id; BLI 2001c; Rose 2003). Thus, the species has low fledging success.

Historical Range and Distribution

Gurney's pitta is native to Myanmar and Thailand, and the species was historically observed throughout the Thai-Malay peninsula (peninsular Thailand and adjacent southern Myanmar) (www.rdb.or.id; BLI 2001c, 2007g). The species has been characterized as formerly common across much of this range (BLI 2000b; Kekule 2005). However, BirdLife International (2001c) pointed out that the Gurney's pitta will not be found in absence of its preferred habitat and characterized the species as *locally* abundant *within its preferred habitat* (lowland, semi-evergreen secondary rainforest in areas with little-to-no undergrowth) (BLI 2000b, 2001c; Gould 1969).

A comparison of the confirmed observations of Gurney's pitta maintained by BirdLife International (2001c) since the species was first described reveals that there have often been large gaps in observations in the past. In Myanmar, the species was not observed for the nearly 30-year period between 1877 and 1904, and went unobserved again in Myanmar between 1914 and 2003. In Thailand, the species was historically observed with greater frequency (www.rdb.or.id; BLI 2001c). However, there were long periods during which the species was not observed in Thailand, including a 50-year period, from 1936 to 1986, during which there was only one confirmed observation of the species in 1952. Gould noted in 1969 that the species ''moves about quite a lot'' (Gould 1969, p. 154), which may be a reference to the species' ''disappearance'' and ''reappearance'' across its range (see also Population Estimates, below).

These occurrence records are likely incomplete for several reasons other than the species' rarity, including: (1) The relative silence of the species, making it difficult to detect when surveying suitable habitat (for instance, Rose (2003) noted that during a 39-hour period observing one nest, only nine calls were heard); (2) long periods of war within the region (Kekule 2005) (for instance, Thailand was involved in or affected by war from 1965–1988); (3) the inaccessible habitat and danger from landmines (in Myanmar, for example (Kekule 2005)); and (4) government regulations restricting access to researchers (Kekule 2005, regarding Myanmar). For these reasons, experts caution against claims of extinction until thorough surveys have been completed (Butchart *et al.*, 2006).

The distribution of Gurney's pitta appears to have steadily contracted in a southerly direction (BLI 2001c). Prior to 1950, the species was observed in several locations within Myanmar's Tanintharyi Division (referred to historically as ''Tenasserim'') and in the central (Prachuap Khiri Khan) and southern (Chumphon, Ranong, Nakhonsrithammarat, Phuket, Phatthatumg, and Trang) Provinces of Thailand. Between 1950 and 1979, the species was only observed once, in the southernmost Province of Thailand's central region, Prachuap Khiri Khan. Between 1980 and 2000, the species was observed only in southern peninsular Thailand (in Phangnga, Krabi, and Suratthani Provinces) (www.rdb.or.id; BLI 2001c). Until its rediscovery in Myanmar in 2003, the species was believed to have a range limited to a 20 mi^2 (50 km^2) area in Thailand (BLI 2000b). Experts believe that steady habitat loss since the 1920s has been a main driver in the species' historical decline (BLI 2000b, 2001c; Rose 2003).

Current Range and Distribution

BirdLife International (2000b) estimated the range of Gurney's pitta to be 942 mi^2 (2,440 km^2). However, range estimates are based on the ''Extent of Occurrence'' for the species, which is defined by the authors as ''the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred, or projected sites of present occurrence of a species, excluding cases of vagrancy'' (BLI 2000b, p. 22). Therefore, this estimate likely includes areas that are unsuitable for the pitta, such that its range is probably smaller than this estimate.

Today, the Gurney's pitta is found in two areas, one within each range country. Details for each range country will be discussed below, starting with Thailand, because much of what we know about the Gurney's pitta is based on this population.

Thailand: In Thailand, Gurney's pitta was rediscovered in 1986 in at least five localities within its historical range, including Prachuap Khiri Khan, Suratthani, Phangnga, Krabi, and Trang Provinces. Although two territories may still exist in Trang Province (in an area called Tambon Aw Tong) (Rose 2003), the only remaining viable population occupies a 2-mi^2 (5.2-km^2) area in Krabi Province, near Mount Khao Nur Chuchi (BLI 2007g; Round & Gretton 1989). Its range is described as extremely small and declining (Rose 2003).

The Mt. Khao Nur Chuchi area may be referred to by any of several names, including Khao Nur Chuchi Reserve, Khlong Pra-Bang Khram Non-Hunting Area, Khlong Pra-Bang Khram Wildlife Sanctuary (Rose 2003, Kekule 2005), and Kao Phra Bang Khram Forest Reserve, which describes an area adjacent to the wildlife sanctuary (www.rdb.or.id; BLI 2001c; TSCWA n.d.). Following the rediscovery of Gurney's pitta near Mt. Khao Nor Chuchi in 1986, a non-hunting area was established in 1987. This area was upgraded to a wildlife sanctuary in 1993; however, crucial areas of pitta habitat were not included in the sanctuary (www.rdb.or.id; BLI 2001c; Round 1999). Rather, the remaining territories remain part of the Kao Phra Bang Khram Forest Reserve (see Factors A and D). Hereafter, this population will be referred to as the Khao Nur Chuchi population.

Myanmar: In Myanmar, Gurney's pitta was rediscovered in 2003 at four sites in the Ngawun Reserve Forest, within its historic range of Tanintharyi Division, in southern Myanmar. All sightings were within 1.2 mi (2 km) of the trans-Tanintharyi highway and within the 193 mi^2 (500 km^2) Ngawun Forest Reserve (BLI–IP & BANCA Darwin Project Office 2004). The species also apparently occurs in neighboring Lenya forest, site of the proposed Lenya National Park, also in Tanintharyi Division (BLI–IP & BANCA Darwin Project Office 2006).

Researchers believe that Myanmar has the largest remaining suitable habitat for the species (BLI–IP & BANCA Darwin Project Office 2004; Eames *et al.* 2005). In 2004, using satellite imagery, the remaining habitat available to the pitta was estimated to be 1,349 mi^2 (3,496 km^2). Most of this habitat is fragmented, but the five largest patches total an area of 553 mi^2 (1,431 km^2) and range in size from 53 to 180 mi^2 (137 to 467 km^2) (BLI–IP & BANCA Darwin Project Office 2004), significantly larger than the entire estimated range of the Gurney's pitta (of 20 mi^2 (50 km^2)) prior to its rediscovery in Myanmar (Eames *et al.* 2005). As of 2005, experts also believed that suitable habitat existed in a neighboring Lenya forest to support Gurney's pitta (BLI–IP & BANCA Darwin Project Office 2006; Eames *et al.* 2005).

Population Estimates

Population estimates are provided for the global population of Gurney's pitta, as well as for each range country. Thailand is discussed before Myanmar, as most information on Gurney's pitta is based on the population in Thailand, which was the only known population of Gurney's pitta until 2003 when it was rediscovered in Myanmar.

Global population estimate: The relative silence of this species has made it difficult to census (David Olson, Irvine Ranch Land Reserve Trust, in litt. February 2007; Rose 2003). Until the recent rediscovery of Gurney's pitta in Myanmar in 2003 (BLI 2003b), the global population estimate for Gurney's pitta was based solely on the Thai population, which stood between 24 and 30 individuals (www.rdb.or.id; BLI 2001c; Rose 2003). With the discovery of the Myanmar population, the global population may be between 175 to 185 individuals. The IUCN has not undertaken a formal re-evaluation of the global population of Gurney's pitta since its rediscovery in Myanmar.

Thailand: The Khao Nur Chuchi population is considered the last remaining viable population in Thailand (Round & Gretton 1989). Censuses undertaken following its rediscovery in the late 1980s aimed to identify additional localities and the number of individuals extant within the area. The species reportedly declined from 44 to 45 pairs in 1986 (BLI 2000b) to 17 pairs in 1987 (Rose 2003) and to 9 pairs in 1997 (BLI 2000b) and then increased to 11 breeding pairs in 2000 (www.rdb.or.id; BLI 2001c). As of 2003, the population stood between 24 and 30 individuals (www.rdb.or.id; BLI 2001c; Rose 2003).

Myanmar: BirdLife International—Indochina Program has been conducting site surveys on the rediscovered populations within the Ngawun Forest Reserve (BLI 2003b). In 2003, at least 10 to 12 pairs were observed (BLI 2003b; Eames *et al.* 2005). In 2004, researchers determined that the Myanmar population was sizable, having made approximately 150 pitta sightings (BLI–IP & BANCA Darwin Project Office 2004).

Extrapolating on the availability of suitable habitat, researchers estimated that the Myanmar population might include up to 8,000 pairs (Eames *et al.* 2005; Grimmitt 2006). However, we believe that this population estimate, based on the availability of suitable habitat, may be an overestimate for this species for two reasons: (1) The Myanmar population may not be randomly distributed in suitable habitat as assumed by these researchers, and (2) the extrapolation does not take into account human-induced threats, such as trapping. Therefore, until the predictions have been ground-truthed, we are unable to consider the 8,000 pair estimate as a reliable reflection of the current population size. We consider the 150 pitta sightings made in 2004 to be the most accurate current estimate of the Gurney's pitta population size in Myanmar.

Conservation Status

The conservation status of the Gurney's pitta is provided both on a global level and according to individual range countries. Thailand is again discussed before Myanmar.

Global population status: The Gurney's pitta has been classified as ''Critically Endangered'' by the IUCN since 1994 (BLI 2005).

Thailand: Gurney's pitta is protected by the Wildlife Animal Reservation and Protection Act (WARPA) in Thailand (B.E. 2535 1992; Eames *et al.* 2005). However, this regulatory mechanism is ineffective at reducing or removing threats directed at the species (see Factor D).

Myanmar: The species is protected in Myanmar by the Wildlife Act of 1994 (www.rdb.or.id; BLI 2001c). However, this regulatory mechanism is ineffective at reducing or removing threats directed at the species (see Factor D).

Summary of Factors Affecting the Gurney's pitta

Where applicable in the sections below, factors affecting the survival of Gurney's pitta are discussed in two parts: (1) Regional factors (affecting or including both range countries), and (2) Factors within individual range countries.

A. The Present or Threatened Destruction, Modification, or Curtailment of the Gurney's Pitta's Habitat or Range

(1) Regional factors

Experts believe that steady habitat loss since the 1920s contributed to the species' historical decline (BLI 2000b, 2001c; Rose 2003). Large-scale conversion of habitat for agriculture (such as rice planting) in Southeast Asia, including Thailand and Myanmar, began in the 1800s. This was followed by forest clearing for cash crops, such as rubber (*Hevea brasiliensis*) and oil palm (*Elaeis guineensis*). The 1950s saw the advent of a commercial logging industry to satisfy an increasing demand for Asian timber (Sodhi *et al.* 2004). Despite a complete logging ban implemented in Thailand in 1989, illegal logging and forest conversion for agriculture continued.

(2) Factors Within Individual Range Countries

Thailand: Thailand has lost an average of 1,274 mi^2 (3,300 km^2) of natural forest since 1960, with deforestation rates in the last three decades often exceeding 3 percent per year (Brown *et al.* 2001). By 1987, only 20 to 50 km^2 of forest below 328 ft (100 m) (habitat preferred by Gurney's pitta) remained in peninsular Thailand (BLI 2000b, 2001c). A portion of the last remaining viable population of Gurney's pitta, the Khao Nur Chuchi population, was included within the Khlong Pra-Bang Khram Wildlife Sanctuary in 1993. However, encroachment for settlements and clearing for crops were continuous problems through the 1990s, as summarized by BirdLife International (2001c). The other, more extensive, portion of the population was included in the Kao Phra Bang Khram Forest Reserve (www.rdb.or.id; BLI 2001c).

There has been a substantial conservation effort to foster sustainable agricultural practices around the Khao Nor Chuchi protected area. In 1990, the Khao Nor Chuchi Lowland Forest Project was established to engage the local community in management, education programs, and ecotourism, to reduce pressure on the remaining forest habitat. This project met with only limited success (BLI 2007g), and illegal forest clearance has persisted into the 21st century (www.rdb.or.id; BLI 2001c; Rose 2003). Moreover, the more recent practice of planting oil palms, which are more profitable than rubber plantations, on illegally cleared forest patches, removes the natural ground cover used for foraging and concealment by the ground-dwelling pitta (Rose 2003).

Myanmar: Gurney's pitta is found within the 193 mi^2 (500 km^2) Ngawun Reserve Forest, described as the largest remaining contiguous lowland forest in southern Myanmar (BLI 2003b, 2005), and also within neighboring Lenya forest, site of a proposed National Park (BLI–IP & BANCA Darwin Project Office 2006), located within Tanintharyi Division. Recent surveys indicated that Myanmar's Tanintharyi Division contains substantial suitable habitat for pittas (estimated to be 1,349 mi^2 (3494 km^2), but much of it was fragmented (BLI 2005) and deforestation for oil palm plantations was ongoing (Eames *et al.* 2005). Between 1990 and 1995, Myanmar lost 1,494 mi^2 (3,870 km^2) of forest per year, averaging a 1.4 percent reduction in forests per year (FAO 1999). In southern Tanintharyi Division, logging reduced one large patch of lowland forest from 163 mi^2 (423 km^2) in 1990 to 102 mi^2 (265 km^2) in 2000 (Eames *et al.* 2005).

Summary of Factor A

Although the known range of the Gurney's pitta has expanded considerably with the rediscovery of the species in Myanmar, habitat conversion, destruction, and encroachment continues to be a significant factor throughout the species' range. Illegal logging and conversion for cash crops continue throughout the species' range. Based on the above information, we find that the Gurney's pitta is at significant risk throughout its range due to the present or threatened destruction, modification, or curtailment of its habitat or range.

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

Gurney's pitta was popular in the pet trade in the 1980s and was overutilized for this purpose by local snare-trappers (BLI 2007g; Rose 2003; Thailand Scientific Authority 1990). Illegal trade in the species was occurring even when experts were not reporting sightings of the species. For instance, the species was reportedly on the price list of an illicit Thai-based animal dealer in 1985, one year before the population was rediscovered in Thailand (Thailand Scientific Authority 1990). Ironically, the rediscovery of the pitta in Thailand can be credited to a wildlife smuggler in Bangkok, who helped rediscover the species. After the smuggler was found with a bird in his possession, he led researchers to a small forest patch in southern Thailand, where the species was subsequently observed (Round & Gretton 1989). The species was listed in Appendix III of CITES by Thailand in 1987 (UNEP–WCMC 2007a), requiring that a certificate of origin or export permit from Thailand accompany international exports of the species. In 1990, Gurney's pitta was uplisted to CITES Appendix I, which prohibited international trade for commercial purposes. According to the WCMC database, there has been no CITES-reported trade in this species since its listing in 1987 (UNEP–WCMC 2007b).

Trapping for the caged-bird trade continued to threaten the species through the late 20th into the early 21st century (www.rdb.or.id; BLI 2001c; Rose 2003), including evidence of non-specific poaching at Khao Nur Chuchi Non-Hunting Area (WorldTwitch Thailand 2000). Although Rose (2003) believed that trapping had ceased, Kekule (2005) found bird-nets surrounding an abandoned pitta nest within the Khao Nur Chuchi population in Thailand; the nets were placed there by villagers to capture the birds (see also Factor D).

We are not aware of any specific information regarding trapping or illegal trade in Myanmar, and there is no specific information indicating that scientific or educational uses of the species are a threat.

Summary of Factor B

Trapping has impacted the species in the past and may be ongoing. Given the species' small population size in Thailand, estimated at 24 to 30 individuals, reports of ongoing trapping and hunting activities within the species' only known range in Thailand is a significant concern. As such, we consider the trapping or hunting to be factors that threaten the species in Thailand.

C. Disease or Predation

There is no information about diseases affecting Gurney's pitta. Regarding predation, dog-tooth cat snake (*Boiga cynodon*) is a natural predator of the Gurney's pitta. The dog-tooth cat snake is a member of the night tree adder family that can reach lengths up to 9 ft (2.75 m). A tree dweller, this snake is native to several southeast Asian countries. In Thailand, the snake has been found in Prachuap Khiri Khan (the location of the largest known pitta population in Thailand) and it shares many similarities with Gurney's pitta, including living mainly in lowland rain forests, rarely entering cultivated areas or human settlements, and principally feeding on birds and their eggs (Thiesen n.d). Gretton (1988) reported that a dog-tooth cat snake killed near a Gurney's pitta nest contained a chick that it had apparently taken from the nest the previous day. Given the small remaining population size in Thailand (estimated to be 11 breeding pairs in 2000 (BLI 2000b)), predation by the dog-tooth cat snake would present a threat to the pitta, but no further information on this threat is available to us.

Summary of Factor C

Predation may affect Gurney's pittas, but there is insufficient information for us to consider this a significant factor currently impacting the Gurney's pitta.

D. The Inadequacy of Existing Regulatory Mechanisms

Thailand: Gurney's pitta is protected by the Wildlife Animal Reservation and Protection Act (WARPA) (B.E. 2535 1992; Eames *et al.* 2005). Under this act, hunting is prohibited (section 16), as is possession of carcasses (section 19), trade (section 20), and collection, harm, or possession of nests (section 21). Violations of sections 16, 19, or 20 may result in imprisonment not exceeding four years or fines not exceeding 40,000 baht, or both. Violations of section 21 may result in imprisonment not exceeding 1 year or fines not exceeding 6,000 baht. However, while Thai law does not allow capture or sale of the Gurney's pitta, the law does allow for possession of the species and bird-nets have recently been found near empty Gurney's pitta nests within the range of Thailand's only remaining viable population of the species (the Khao Nur Chuchi population) (Kekule 2005). This suggests that this regulation is inadequate to protect the few remaining individuals of this species from hunting (Factor B).

Protection of the species' habitat has not been effective in addressing forest clearance and poaching (Factor A). When the Khlong Pra-Bang Khram Wildlife Sanctuary was established in 1993, it provided incomplete protection for pitta territories, as only 5 of the 21 known pitta territories were encompassed within the Sanctuary. The most important and extensive areas of pitta habitat and territories were not included, including a crucial 12 mi^2 (30 km^2) area considered to be core to the pitta habitat (Round 1999; BLI 2001c). Sanctuaries are reportedly rarely patrolled by staff (WorldTwitch Thailand 2000) and a survey in 2001 confirmed that protection and law enforcement at Khao Nor Chuchi was essentially nonexistent (Rose 2003). While the Sanctuary receives funds for its management from the central government, authority to address problems within the Reserve is given to the provincial officials. This provides neither the authority nor the responsibility for Reserve staff to focus on problems within the reserve (BLI 2001c). As habitat destruction is ongoing within giant ibis habitat (BLI 2001c; Kekule 2005; Rose 2003), this regulatory mechanism is ineffective at addressing the threat of habitat destruction (Factor A).

Myanmar: This species is considered a ''completely protected'' species of wildlife under section 15(a) of Myanmar's Protection of Wildlife and Wild Plants and Conservation of Natural Areas Law of 1994 (Forest Department Notification No. 583/94; Protection of Wild Life and Wild Plants and Conservation of Natural Areas Law 1994). This law made it is illegal to kill, hunt, wound, possess, sell, transport, or transfer a completely protected species without permission (section 37).

Violators of this law are subject to imprisonment for up to 7 years or a fine up to kyats 50,000, or both (section 37). We have no information that the species is being trapped, hunted, or sold in Myanmar. Therefore, this regulation is not currently removing or reducing the primary threat to this species within Myanmar, habitat destruction (Factor A).

There are currently no protected areas in the peninsular region where the Gurney's pitta is found (Hirschfeld 2008). Within the Ngawun Forest Reserve, the habitat of the Gurney's pitta is protected under the provisions of the Burma Forest Act of 1902, as amended (Conservation Monitoring Centre 1992). Prohibited activities in reserved forests include trespassing, pasturing, damaging trees, setting fires, mining, cultivation, poisoning or dynamiting, hunting, shooting, fishing, or setting traps or snares. According to BirdLife International—Indochina Program (BLI–IP & BANCA Darwin Project Office 2005), the Ngawun Forest Reserve is the largest block of lowland forest in southern Myanmar, but it remains inadequately protected due to ineffective enforcement. Therefore, this regulation is not removing or reducing the primary threat to this species within Myanmar, habitat destruction (Factor A).

The species is also apparently extant in neighboring Lenya forest, site of the proposed Lenya National Park (BLI–IP & BANCA Darwin Project Office 2006). However, it appears that the Park has yet to be established and, as currently drawn, its boundaries would not encompass critical pitta territories within the Lenya Forest or the Ngawun Forest Reserve (BLI–IP & BANCA Darwin Project Office 2006; Grimmitt 2006). Therefore, because that establishment of the Park as currently drawn would exclude pitta territory, this mechanism would not likely remove or reduce the primary threat to this species within Myanmar, habitat destruction (Factor A).

Summary of Factor D

Although regulatory mechanisms are in place that could reduce or remove threats to the species, implementation of these mechanisms appears to be slow (such as the delay in establishing the proposed National Park), ineffective (such as the inability to quell poaching threats to the species), or inadequate. For instance, in Thailand, there is evidence of trapping within Gurney's pitta territory. Despite indications that poaching is ongoing, the law allows for possession of the species, although it does not allow capture or sale. Therefore, we believe the inadequacy and ineffective implementation of regulatory mechanisms are contributory risk factors that endanger the Gurney's pitta.

E. Other Natural or Manmade Factors Affecting the Continued Existence of the Species

Collection of forest products may constitute a disturbance to Gurney's pitta in Thailand during their breeding season. The edible fruits of the rakum palm, one of the palms in which the Gurney's pitta nests, are sought after in Thailand (BLI 2007g). Peak harvest occurs in June and July (World Agroforestry Center (WAC) n.d.), coinciding with the Gurney's pitta breeding season (www.rdb.or.id; BLI 2001c, 2007g). However, forest-collected fruit is considered inferior to the cultivated variety, harvest has never been tracked (WAC n.d.), and we are unaware of any research concerning this type of disturbance in relation to the Gurney's pitta. Thus, we are unable to conclude that this activity threatens the species' survival, due to insufficient information.

Small, isolated populations of wildlife species are susceptible to demographic and genetic problems (Shaffer 1981). These threat factors, which may act in concert, include natural variation in survival and reproductive success of individuals, chance disequilibrium of sex ratios, changes in gene frequencies due to genetic drift, and diminished genetic diversity and associated effects due to inbreeding. Demographic problems may include reduced reproductive success of individuals and chance disequilibrium of sex ratios (Charlesworth & Charlesworth 1987; Shaffer 1981). Using the 50 / 500 rule (as described under Factor E for the black stilt) (Soulé 1980; Hunter 1996) and given the two population estimates (24 to 30 in Thailand (www.rdb.or.id; BLI 2001c; Rose 2003), and 150 in Myanmar (BLI–IP & BANCA Darwin Project Office 2005)), the population in Thailand has likely undergone inbreeding. In addition, both the Thai and the Myanmar populations exist at numbers well below the minimum (of at least 500 individuals in order to prevent the loss of genetic diversity over time and maintain an enhanced capacity to adapt to changing conditions. As such, we currently consider the species to be at significant risk due to lack of near- and long-term genetic viability.

Summary of Factor E

The Gurney's pitta may be adversely affected by collection of the rakum fruit in Thailand, which grows in a tree in which the pitta nests and which ripens coincident with the Gurney's pitta's breeding season. However, no specific data exist to indicate that disturbance from fruit collection may be an actual threat. Therefore, we do not consider fruit collection to be a factor impacting the Gurney's pitta at this time.

The small population size of the Gurney's pitta, estimated at 24 to 30 in Thailand and 150 in Myanmar, poses a risk to this species throughout its range with regard to lack of near-term long-term genetic viability and to potential demographic shifts. Therefore, we consider the species' extremely small population size and associated genetic and demographic risks to be significant factors that endanger the Gurney's pitta throughout its range.

Conclusion and Determination for the Gurney's Pitta

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the Gurney's pitta. We have determined that the species is in danger of extinction throughout all of its known range primarily due to habitat loss (Factor A), trapping, or hunting in Thailand (Factor B), and genetic and demographic risks associated with the species' small population size (Factor E). Furthermore, we have determined that the inadequacy of existing regulatory mechanisms to reduce or remove these threats is a contributory factor to the risks that endanger this species' continued existence (Factor D). Therefore, we are determining endangered status for the species under the Act. Because we find that the Gurney's pitta is endangered throughout all of its range, there is no reason to consider its status in any significant portion of its range.

V. Long-Legged Thicketbird (*Trichocichla rufa*)

Species Description

The long-legged thicketbird is an Old World warbler belonging to the Sylvidae family, and native to the Fiji Islands. The species is also commonly known as the long-legged warbler (BLI 2007i). Local residents named the secretive thicketbird ''Manu Kalou,'' or ''Spirit Bird,'' during the 19th century because of its ethereal voice (BLI 2000c; Dutson & Masibalavu 2004). Adults stand 6 in (17 cm) tall, with long blue legs, a short black bill, and a long tail. Upperparts of the body are warm brown with a long supercilium (head plumage). The throat is white and the flanks are a pale, rufous color (BLI 2007i).

Taxonomy

The long-legged thicketbird was described by Reichenow as *Trichocichla rufa* in 1890, and placed in the Sylvidae family as a monospecific genus. Two specimens discovered on the island of Vanua Levu in 1974 were described as a distinct subspecies (*Trichocichla rufa clunei*) (BLI 2003c; Kirby 2003b; Helen Pippard, Director of Environment, Suva, Fiji, in litt. February 2007). However, ITIS and BirdLife recognize the long-legged thicketbird only to the species level, and we accept this taxonomy.

Habitat and Life History

The long-legged thicketbird requires intact mid- to high-elevation forest associated with riverine habitat and dense vegetation (H. Pippard in litt. February 2007). Its habitat is dominated by old-growth montane forest (BLI 2007i), and the species is found at altitudes ranging from 2,625 to 3,281 ft (800 to 1000 m) (Dutson & Masibalavu 2004).

Because this species was known only from four voucher specimens until 2002, very little is known about its life history (BLI 2007i). It is characterized as a secretive ground-warbler that is easily overlooked unless it is singing (BLI 2007i). Its call is distinctive, and recognizing its song is considered key to identifying it in the wild (Dutson & Masibalavu 2004).

Historical Range and Distribution

The long-legged thicketbird is endemic to the Fijian Islands. The Fijian Archipelago comprises over 320 islands, over an area approximating 502,000 mi² (1.3 million km²) (Chand 2002). Historically the species was found on two Fijian islands: Viti Levu and Vanua Levu. Viti Levu, meaning ''Big Fiji,'' is the largest island, with an area of 4,011 mi² (10,390 km²). Vanua Levu, meaning ''Big Land,'' is little more than half as large at 2,135 mi² (5,530 km²) (Chand 2002).

The long-legged thicketbird was long considered extinct, with no confirmed observations since 1894 (BLI 2003c; Kirby 2003b) and several unconfirmed sightings in 1967, 1973, and 1991 (BLI 2000c). The first confirmed sighting in recent time was that of two individuals in 1974, found on the island of Vanua Levu (BLI 2003c; Kirby 2003b). There was no evidence of its continued existence until 2002, when it was rediscovered on Viti Levu (BLI 2003c). The Fijian government considers the species to be extinct on Vanua Levu, where forests are less intact and there have been greater impacts from forest loss, including invasive species (H. Pippard in litt. February 2007).

Current Range and Distribution

The long-legged thicketbird was rediscovered in 2002, although confirmation of the sighting took nearly a year (BLI 2003c; Kirby 2003b). It was located at several sites on Viti Levu, found only in dense undergrowth of the Fijian mountains (BLI 2003c; Kirby 2003b; H. Pippard in litt. February 2007). However, a researcher who spent 5 years working in Fiji on conservation projects indicated that the species is ''commonly found if you know where to look for it in mid-elevation rocky streams with dense overstories'' (D. Olson in litt. February 2007). The largest known concentration of the long-legged thicketbird is found within the approximately 2 mi² (5 km²) area known as the Wabu National Forest Reserve (BLI 2007i). Little is known about the species' current range, necessitating additional surveys in suitable habitat (BLI 2007i).

Population Estimates

There is insufficient information to determine the historic population levels of this species (BLI 2007i). Today, researchers believe that the species is locally common in ideal habitat (unlogged forest at elevations between 2,625 and 3,281 ft (800 and 1000 m)), but that it is patchy in distribution and absent from most forest (BLI 2003c, 2007i; D. Olson in litt. February 2007; Kirby 2003b). The current population is estimated to be between 50 to 249 individuals. However, this estimation is a categorical one, used by BirdLife International to conform to the IUCN criteria. The actual number of individuals may be much smaller (or larger) than this range suggests. In surveys conducted from 2002 to 2005, 12 pairs were discovered in Wabu (BLI 2003c, 2007i; Kirby 2003b). Nine pairs were found along a 1.24-mi (2-km) length of stream in dense undergrowth thickets; two of these pairs were accompanied by recently fledged juveniles. Using the data from the 2005 field surveys, only 30 individuals were observed during field surveys in 2005 (BLI 2003c; Kirby 2003b).

Conservation Status

The Fiji Department of Environment considers the extant long-legged thicketbird on Viti Levu to be vulnerable to further decline or extinction. Conservation priorities for this species include: protection of forest and research on the species' habitat requirements and impacts of invasive species on the species (H. Pippard in litt. February 2007). As of 2007, the species was classified by the IUCN as endangered, where it was previously classified as data deficient (BLI 2006b, 2007i; H. Pippard in litt. February 2007).

Summary of Factors Affecting the Long-Legged Thicketbird

A. The Present or Threatened Destruction, Modification, or Curtailment of the Long-Legged Thicketbird's Habitat or Range

Habitat destruction from logging, conversion to agriculture, and invasive species threatens the long-legged thicketbird habitat. The most recent estimates of forest cover on the islands of Vanua Levu and Viti Levu are from 1995. In 1995, the total forested area, including mangrove forest, pine plantation, hardwood plantation, scattered natural forest, medium dense natural forest, and dense natural forest, on the Fiji Islands was 3,293 mi² (9933 km²) (Lal & Touvou 2003). This equated to just under half of Fiji's total land area and included an excess of 490 mi² (1,270 km²) of the dense forest, preferred by the long-legged thicketbird (on Viti Levu, and 463 mi² (1,200 km²) on Vanua Levu) (Chand 2002). Although there is more forested area on Vanua Levu than on Viti Levu, Fiji considers that the degree of habitat degradation on Vanua Levu has resulted in the species' extirpation from that island (H. Pippard in litt. February 2007).

Logging: According to the Fijian government, logging of virgin forests is the primary threat to this species, which prefers intact forest habitat (H. Pippard in litt. February 2007). Eighty-three percent of the total land area, including most of the natural forest cover, is privately owned (McKenzie *et al.* 2005). The forestry sector contributes 2.5 percent to Fiji's gross domestic product (GDP) and about F$50 million (US$27.6 million) in foreign exchange export earnings annually (McKenzie *et al.* 2005).

The Fijian government began large-scale planting of pine and hardwoods in the 1960s, such that today 13 percent of Fiji's forests are planted. In 2003, there were approximately 204 mi² (529 km²) of hardwood plantations, mainly big-leaf mahogany (*Swietenia macrophylla*), and 179 mi² (463 km²) of pine (*Pinus caribea*) plantations (ITTO 2005). Habitat conversion for timber plantations, including pine and big-leaf mahogany, in long-legged thicketbird habitat renders the habitat unsuitable for the bird (BLI 2003c), as it prefers intact forest (Pippard in litt. February 2007). See also Factor D.

Conversion to agriculture: The economy is dominated by the sugar industry and food crops, including taro, cassava, sweet potatoes or kumala, and a wide variety of fruits and vegetables. An estimated 67 percent of the labor force is employed in agriculture, and this sector of the economy accounts for almost 21 percent of Fiji's GDP (Chand 2002). In 2007, Fiji released census data that estimated the population on the islands to be 827,900 inhabitants. This represents an increase of 53,000 people since the 1996 census (Fiji Government Online 2007). Most of these people inhabit the two main islands of Viti Levu and Vanua Levu (Dutson & Masibalavu 2004). As the population increases, the production area of these and other major food crops continues to increase each year. In Fiji, all preferred arable lands are fully utilized or unavailable for land tenure reasons. Thus, agriculture has expanded onto steeper marginal land to the interior of the island (Chand 2002). Agricultural conversion produces unsuitable conditions for the long-legged thicketbird, which prefers intact forests with dense vegetation, and the continuing expansion of agriculture into steeper lands to the interior jeopardizes the long-legged thicketbird, which prefers mid- to high-elevation forest (H. Pippard in litt. February 2007).

Invasive species: Although BirdLife International (2007i) noted that the influx of invasive species has not been shown to have deleterious effects on the suitability of the habitat for the long-legged thicketbird, it is unclear what factors were considered to arrive at this determination, including whether they referred to invasive animals or plants. The long-legged thicketbird prefers intact forest, and the Fijian government considers invasive species to be a factor that contributed to the species' extirpation from Vanua Levu (H. Pippard in litt. February 2007). Invasive plants and animals are problematic on Viti Levu (See Factor C for further discussion on invasive animals). African tulip tree (*Spathodea campanulata*) is invasive in forests and open areas of Viti Levu (McKenzie *et al.* 2005).

No longer facing the natural enemies or competition from other species that they faced in their place of origin, invasive plants are capable of spreading and outcompeting native species. Invasive plants can spread and reproduce prolifically, causing significant changes to ecosystems and upsetting their ecological balance.

Human disturbance, such as logging activities and agricultural conversion, is considered a major vector for introducing invasive plants. Once an invasive plant is introduced to an area, it has the potential to invade larger areas (USGS 2006). Thus, in the face of increasing habitat disturbance, invasive plants could pose a threat to the long-legged thicketbird, which prefers intact primary forest (H. Pippard in litt. February 2007). However, we are unaware of specific information regarding the effect of invasive plants on the long-legged thicketbird or its habitat. As, such we are unable to make a determination as to the threat this factor might cause, if any, to the species.

Summary of Factor A

Habitat destruction from logging and habitat conversion to agricultural purposes produce unsuitable conditions for the long-legged thicketbird, which prefers intact forest with dense vegetation. We consider habitat destruction to be a significant threat to the long-legged thicketbird that endangers the species throughout its range.

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

According to the Fijian government, there is no trade, collection, or captive breeding of the long-legged thicketbird at this time, nor is any likely in the future (H. Pippard in litt. February 2007). There is no known threat to the species from use for commercial, recreational, scientific, or educational purposes. The species has not been formally considered for listing in the Appendices of CITES (www.cites.org).

C. Disease or Predation

We have no information to indicate that the long-legged thicketbird is threatened by disease.

Predation by invasive animals, namely rats (*Rattus* spp.) and mongooses (*Rallus phillopensis*), is considered by Fiji to be a highly significant threat to the species (H. Pippard in litt. February 2007). Mongooses were introduced in 1883 to Fiji to kill rats, but both these species could potentially be serious predatory threats to the long-legged thicketbird (BLI 2000c). According to BirdLife International (2007i), however, the long-legged thicketbird has been found successfully nesting alongside these predators in Wabu, indicating that mongooses may not be predators after all. The first sighting of this species in 2002 was of a long-legged thicketbird warding off a mongoose from its nearby nest, which would indicate that the species exhibits anti-predatory behavior (Dutson & Masibalavu 2004). Given the species' small population size, between 50 to 249 individuals, predation could pose a significant risk to the long-legged thicketbird. However, there is insufficient information to determine that predation is ongoing or has the potential to negatively affect this species.

Summary of Factor C

More information is needed in order to determine the role of predation, if any, in this species' decline. Currently, there is insufficient information to determine that threats from predation are contributing to the species' risk of extinction.

D. The Inadequacy of Existing Regulatory Mechanisms

The long-legged thicketbird is a threatened species under Schedule 1, Section 3 of Fiji's Endangered and Protected Species Act of 2002 (No. 29 of 2002). This law and its implementing regulations (Endangered and Protected Species Regulations (Act No. 29 2002; Legal Notice No. 64) prohibit trade in the thicketbird, unless permitted. As trade is not known to be a threat to the thicketbird, this law and its implementing regulations do not address the conservation needs of the species.

The thicketbird is also a ''protected bird'' under Fiji's Birds and Game Protection Act of 1923 (Rev. 1985), as amended. Under this Act it is illegal to willfully kill, wound, or take any protected bird, or attempt to sell, possess, or export a protected bird, or their parts, nests or eggs (Part II, § 3). The penalty for violating this Act is a fine not to exceed $50, or, if this amount cannot be paid, imprisonment for up to 3 months (Part IV, § 15) (Birds and Game Protection Act 1985). As hunting and trapping are not known to be threats to the thicketbird, this law and its regulations do not address the conservation needs of the species.

Some of the forest habitat of the long-legged thicketbird is within the Wabu National Forest Reserve and is protected under Fijian law (BLI 2007i). However, the protections within the reserve are not absolute and the Forestry Act has a number of serious weaknesses. For example, legal loopholes permit clearcutting of forests over which the Forestry Department has no control, and all protected areas established under the provisions of the Forestry Act are subject to dereservation at the ministerial level; and reserve forests have frequently been dereserved (World Conservation Monitoring Centre 1992). In addition, forest reserves are managed as long-term production forests, with extraction being allowed by permit (Forest Decree 1992, Part III). In 2003, experts considered that insufficient protection of long-legged thicketbird habitat would lead to a high probability of habitat conversion or destruction (BLI 2003c; Kirby 2003b). According to Dutson and Masibalavu (2004), BirdLife Fiji is working with the Department of Forestry to focus on long-term protection within the Wabu and with local communities to focus on forest conservation and alternatives to forest destruction, such as ecotourism, which may help to moderate habitat destruction. However, we consider this regulatory mechanism to be inadequate in removing or reducing the primary threat to this species, habitat destruction.

Summary of Factor D

While some of the forest habitat of the long-legged thicketbird is within the 2-mi^2 (5-km^2) Wabu Forest Reserve (Wabu) and is protected under Fijian law, the regulatory mechanisms in place to protect the species do not adequately reduce or remove the primary manmade threat to this species, habitat destruction (Factor A). We conclude that the inadequacy of existing regulatory mechanisms is a contributory risk factor that endangers the long-legged thicketbird.

E. Other Natural or Manmade Factors Affecting the Continued Existence of the Species

Two additional factors are considered herein, genetic risks associated with small population sizes and threats from stochastic events.

Effect of small population sizes: Small, isolated populations of wildlife species are susceptible to demographic and genetic problems (Shaffer 1981). These threat factors, which may act in concert, include natural variation in survival and reproductive success of individuals, chance disequilibrium of sex ratios, changes in gene frequencies due to genetic drift, and diminished genetic diversity and associated effects due to inbreeding, loss of genetic variation, and accumulation of new mutations. Inbreeding can have individual and population consequences by either increasing the phenotypic expression of recessive, deleterious alleles or by reducing the overall fitness of individuals in the population (Charlesworth & Charlesworth 1987; Shaffer 1981). In the absence of more species-specific life history data, a general approximation of minimum viable population size is referred to as the 50/500 rule (Soulé 1980; Hunter 1996), described under Factor E for the black stilt. The available information indicates that, with an N_e of approximately 50 (BLI 2007i), the long-legged thicketbird teeters on the edge of the minimum number of individuals required to avoid imminent risks from inbreeding (N_e = 50). The current maximum estimate of 249 individuals for the entire population (BLI 2007i) is only half of the upper threshold (N_e = 500) required to maintain genetic diversity over time and to maintain an enhanced capacity to adapt to changing conditions. As such, we currently consider the species to be at risk due to its lack of near- and long-term genetic viability.

Threats from stochastic events: Small populations of wildlife species also susceptible to stochastic environmental events (for example, severe storms, prolonged drought, extreme cold spells, wildfire). Stochastic events could result in extensive mortalities from which the population may be unable to recover, leading to extinction (Caughley 1994; Charlesworth & Charlesworth 1987). Fiji is susceptible to damage from tropical storms and cyclones. Tropical storms, which can sustain winds up to 130 miles per hour (mph) (209 kilometers per hour (kph)), are common in the South Pacific from November to April (Ligaiula 2007). Cyclones, also known as typhoons, are storms that typically form at sea and move inland, generating high winds exceeding 130 mph (209 kph) up to 200 mph (322 kph). Thirteen tropical storms have hit Fiji in the past 10 years (Associated Press 2007). In December 2007, Cyclone Daman made landfall on Viti Levu, with winds up to 155 mph (250 kph). Trees were destroyed, and heavy rains caused landslides and flooding in low-lying areas (Ligaiula 2007). The extant long-legged thicketbird population is extremely small and highly localized (BLI 2003c, 2007i; Kirby 2003b). Therefore, any additional stress to the population due to stochastic events, such as cyclones, represents a risk to the species and could lead to a further decline in the species' abundance or the extent of its occupied range.

Summary of Factor E

In addition to ongoing threats to the species' habitat (see Factor A), a major risk to the long-legged thicketbird is lack of near- and long-term genetic viability associated with the extant population's extremely small size. In addition, the long-legged thicketbird is vulnerable to reductions in numbers or extinction from stochastic events, such as cyclones. We consider the species' extremely small population size, the associated genetic risks and demographic shifts, and vulnerability to stochastic events to be significant risks that endanger the long-legged thicketbird throughout its range.

Conclusion and Determination for the Long-Legged Thicketbird

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the long-legged thicketbird, above. We have determined that the species is in danger of extinction throughout all of its known range primarily due to ongoing threats to its habitat (Factor A), lack of near- and long-term genetic and associated demographic shifts, and susceptibility to stochastic events due to risks associated small population sizes (Factor E). Furthermore, we have determined that the inadequacy of existing regulatory mechanisms (Factor D) is a contributory risk factor that endangers the species. Therefore, we are determining endangered status for the long-legged thicketbird under the Act. Because we find that the long-legged thicketbird is endangered throughout all of its range, there is no reason to consider its status in any significant portion of its range.

VI. Socorro Mockingbird (Mimus graysoni)

Species Description

The Socorro mockingbird is a member of the Mimidae family, and endemic to Socorro Island, Mexico. This species is also referred to as Socorro thrasher, especially in older literature (e.g., Brattstrom & Howell 1956). Adults stand about 10 in (25 cm) tall and are mostly brown, with whitish underparts, darker wings (except for two narrow bands of white), a dark tail, reddish iris, and dark gape (the soft tissue at the corner of the mouth) (BLI 2007f; Martínez-Gómez & Curry 1998). Male and female Socorro mockingbirds have similar plumage, but males are larger than females. A juvenile (first-year bird) can be distinguished from an adult by its plumage, spotted breast, grayish iris, and yellowish gape (Martínez-Gómez & Curry 1998).

Taxonomy

The Socorro mockingbird was first taxonomically described as *Mimodes graysoni* (Mimidae family), by Lawrence in 1871. Ornithologists recognized that the species' behavioral characteristics were reminiscent of the mockingbird genus, *Mimus*, of the same family (Barber *et al.* 2004). Genetic analysis conducted by Barber *et al.* (2004) demonstrated that the species is most closely related to *Mimus* spp. In our proposed rule, we referred to this species as *Mimodes.* However, we find the appropriate taxonomy for the species is *Mimus graysoni,* which follows the Integrated Taxonomic Information System (ITIS 2007).

Habitat and Life History

The geography of Socorro Island rises from sea level on the coast to a height of nearly 3,445 ft (4,000 m) elevation on the peak of Mount Evermann, in the center of the island (Comisión Nacional de Áreas Naturales Protegidas (CONANP) n.d.). Socorro mockingbirds are found in greatest abundance at elevations above 1,969 ft (600 m) (Martínez-Gómez & Curry 1996). They prefer undisturbed montane areas and primary forests that have a variety of fruit-bearing plants and a high density of tree species. Dominant plant species in the Socorro's preferred habitat include holly (*Ilex socorrensis*), *Guettarda insularis* (no common name), and lion's paw (*Oreopanax xalapensis*), along with the understory *Triumfetta socorrensis* and *Eupatorium pacificum* (Martínez-Gómez *et al.* 2001). Socorro mockingbirds forage on fruits, invertebrates, and small arthropods (Martínez-Gómez *et al.* 2001). They have been observed feeding on blowfly larvae on sheep carcasses (Brattstrom & Howell 1956).

Little is known about the Socorro mockingbird's life history; breeding information is based largely on studies conducted by Martínez-Gómez and Curry (1995) during 1993 and 1994. They found four nests in 1994, which were located about 12 ft (3.7 m) off the ground, each in a different species of tree: Holly, *Bumelia socorrensis* (no common name), *Guettarda insularis* (no common name), and *Meliosma nesites* (no common name). Researchers inferred that nesting likely occurs between November and July, with a clutch size of three. Eggs were incubated by females only (Martínez-Gómez & Curry 1998) for no more than 15 days (Martínez-Gómez & Curry 1995). A large number of subadults recorded during 1994 suggested high breeding success for the species (J. Martínez-Gómez in litt. via Comisión Nacional Para el Conocimiento y Uso de la Biodiversidad (CONABIO) February 2007).

Historical Range and Distribution

The Socorro mockingbird is endemic to Socorro Island, Mexico, in the Revillagigedo archipelago of Mexico. Socorro Island is the largest of four Revillagigedo Islands, with an approximate land area of 54 mi² (140 km²) (Walter 1990). The island is 210 mi (338 km) southwest of Baja California, Mexico. The Socorro mockingbird was widespread and common on the island prior to 1958 (Martínez-Gómez 2002). Brattstrom and Howell (1956) observed the species in coastal locations in the southwest part of the Island, inland at higher elevations, and in canyons on the northern part of the Island. Socorro mockingbird may have inhabited the southwest portions of the island only seasonally (R. Curry in litt. February 2007). By the 1980s, the species was restricted to undegraded fig groves (*Ficus cotinifolia*), habitat which was becoming rare (Jehl & Parkes 1982). Habitat reduction is considered the primary cause of population and range declines of the Socorro mockingbird (BLI 2000d).

Current Range and Distribution

The current range of the Socorro mockingbird is limited to an estimated 6 mi² (15 km²) area. The species is found in forests above 1,640 ft (500 m) (Martínez-Gómez 2002) and is most abundant at elevations above 1,969 ft (600 m) around Mt. Evermann (CONANP n.d.; Martínez-Gómez & Curry 1996; Wehtje *et al.* 1993).

In our proposed rule (71 FR 67530), we noted, ''the species is less common in taller forest patches and fig groves at low and mid elevations.'' Martínez-Gómez (in litt. via CONABIO February 2007) pointed out that this may be misleading. The field study conducted by Martínez-Gómez *et al.* (2001) indicated that the absence of the Socorro mockingbird in the low-elevation fig grove was due to habitat degradation. This is discussed further under Factor A.

In our proposed rule, we noted that the species ''is absent from areas of [croton] *Croton masonii* scrub near sea-level (Martínez-Gómez & Curry 1996).'' Curry (in litt. February 2007) clarified that it is uncertain whether Socorro mockingbird ever inhabited the croton scrub habitat, except as visitors during the nonbreeding season.

Population Estimates

The Socorro mockingbird was once considered the most abundant landbird on Socorro Island (Brattstrom & Howell 1956). The population declined through the 1960s and 1970s, and by 1978 it was feared to be on the verge of extinction (Jehl & Parkes 1982). In our proposed rule, we wrote that ''current estimates of population size for the species range from 50 to 249 individuals (BLI 2000).'' According to Dr. Robert Curry (Associate Professor, Villanova University, Villanova, Pennsylvania, in litt. February 2007), there are two problems with this figure: (1) It does not reflect the most recent field data, but reflects data collected between 1988 and 1990; (2) it is not an ''estimate'' of the Socorro mockingbird population, but rather the ''category'' to which BirdLife International assigned the species, in accordance with the IUCN listing criteria. Based on the most recent surveys, carried out between 1993 and 1994, the estimated population total was 353 individuals, with a calculated uncertainty of 66 (Martínez-Gómez & Curry 1996). Taking the calculated uncertainty of this estimate into account, the estimated total population ranged between 287 and 419 (R. Curry in litt. February 2007). This estimate was reconfirmed in the summer 2006, when Dr. Juan Martínez-Gómez (Island Endemics Foundation, Mexico, in litt. via CONABIO February 2007) inspected previous banding areas on the Island. He encountered a population similar to that studied by Martínez-Gómez and Curry (1996), above, with an estimated population size between 298 and 408 individuals. While Dr. Martínez-Gómez cautions against extrapolating these estimates beyond the banding areas studied, he indicated a likelihood that additional Socorro mockingbirds are on the island (J. Martínez-Gómez in litt. via CONABIO February 2007).

In our proposed rule, we wrote, ''of 215 birds ringed in 1993–1994, 55 percent were subadults.'' However, Martínez-Gómez (in litt. via CONABIO February 2007) noted this estimate was erroneously based on the pooled data from the 1993–1994 banding study conducted by Martínez-Gómez and Curry (1996), which biased our estimate. The banding for the 2-year study took place at different times of the year: The banding in 1993 took place after the breeding season, and the 1994 banding took place during the entire breeding season. Thus, in analyzing the 1994 data, which would be more representative of actual age ratios, it was apparent that sex ratios were not disproportionate and that the population had produced many young. Thus, the 1994 data suggest that the species has a high breeding success and that the population may be successful in recolonizing the area once habitat quality improves (J. Martínez-Gómez in litt. February 2007).

Conservation Status

The IUCN has listed the Socorro mockingbird as ''Critically Endangered'' since 2000, due to loss of habitat and the small remaining number of mature adults (BLI 2007c). The species is categorized as ''*Peligro*'' in Mexico, meaning it is in danger of extinction (Hesiquio Benítez Díaz, Director de Enlace y Asuntos Internacionales, CONABIO, Tlalpan, Mexico, in litt. February 2007).

Summary of Factors Affecting the Socorro Mockingbird

A. The Present or Threatened Destruction, Modification, or Curtailment of Socorro Mockingbird's Habitat or Range

Socorro mockingbird habitat in the southern portions of the island has been severely degraded by construction of a naval base and sheep overgrazing for the past 50 years. In addition, locust swarms (*Schistocerca piceifrons*) have invaded that island since the mid-1990s. These threats to Socorro mockingbird habitat are discussed in turn.

Naval base: The Mexican Navy built a base on Socorro Island in the late 1950s (Martínez-Gómez *et al.* 2001). Built on the southernmost tip, at Bahia Vargas Lozano, the base supports more than 200 personnel and family (Wehtje *et al.* 1993). The Socorro mockingbird prefers undisturbed montane areas, and may have occupied the area seasonally before the base was built (R. Curry in litt. February 2007). During construction, native vegetation was removed from around the base and replaced with non-native grasses (Martínez-Gómez *et al.* 2001). Habitat destruction caused by construction of the naval base contributed to the species' extirpation from the southern third of the island (BLI 2000d), although not to the same extent as sheep overgrazing.

Sheep overgrazing: The greatest impact on the habitat of Socorro Island has been severe degradation due to intensive grazing by introduced mammals (BLI 2000d; Curry in litt. February 2007; Martínez-Gómez in litt. February 2007; Martínez-Gómez & Curry 1995, 1996; Martínez-Gómez *et al.* 2001). Socorro Island has no native mammals (Jehl & Parkes 1982). In our proposed rule, we noted that Cody (2005) reported that Socorro mockingbird habitat is threatened by destruction from introduced rabbits and pigs. However, Curry (in litt. February 2007) pointed out that, while rabbits and pigs are problematic on the nearby island of Clarión, these two exotic mammals were never introduced on Socorro.

Sheep were brought to Socorro Island near the end of the 19th century and, by 1956, there were an estimated 2,000 sheep living in the southern portions of the island (Brattstrom & Howell 1956). Left feral, the sheep overgrazed, creating extensive open areas (2005) and leaving the soil vulnerable to erosion (R. Curry in litt. February 2007; Wehtje *et al.* 1993). The Socorro mockingbird prefers undisturbed montane areas and forests with a dense understory. In the southern fig forests, hop bush (*Dodonaea viscosa*) has replaced the original understory, and these areas are too degraded for the Socorro to inhabit (Martínez-Gómez *et al.* 2001).

Habitat degradation caused by sheep drastically altered habitat on Socorro Island (BLI 2000d; R. Curry in litt. February 2007; Martínez-Gómez 2002), especially low- to mid-elevation fig forests (ranging in altitude from 0 to 1,640 ft (to 500 m)) in the southern portion of the island (Martínez-Gómez in litt. February 2007). By 1990, they had overgrazed the southern third of the island (Martínez-Gómez & Curry 1996), where the Socorro mockingbird was once plentiful (Brattstrom & Howell), although perhaps only seasonally (R. Curry in litt. February 2007). In the northern regions of Socorro Island, low- to mid-elevation fig forests are largely undegraded and serve as important habitat for the Socorro mockingbird (Martínez-Gómez & Curry 1996; Martínez-Gómez *et al.* 2001). Sheep overgrazing extirpated the species from one-third of its former range (BLI 2000d).

Locust swarms: Another factor causing the degradation of Socorro mockingbird habitat was brought to our attention by Martínez-Gómez (in litt. February 2007). According to Martínez-Gómez (2005), permanent locust (*Schistocerca piceifrons*) swarms have invaded the island since 1994. The locusts swarm twice yearly and are capable of reaching all points on the island. The swarms have defoliated trees and shrubs in several regions of the island, which decreases the availability of food from fruit trees and modifies the primary forest habitat which the species prefers. Locusts are especially pronounced in the southern portion of the Island. A larger number of young locusts and locusts in non-swarming stages are found in the degraded habitats in the south (Martínez-Gómez 2005). Martínez-Gómez (2005) concluded that the higher intensity of outbreaks in the southern portion of the island was an indirect result of sheep overgrazing and predation caused by introduced mammals, namely sheep and cats (see Factor C). Sheep overgrazing has created open conditions, providing suitable habitat for locust reproduction, as evidenced by the high number of young and non-swarming stages of locust found primarily in those areas (Martínez-Gómez 2005). In the northern portions of the island habitat is less degraded and bird densities are higher. Less degraded habitat provides less favorable conditions for the locusts and the swarms are less intense. Because birds eat locusts, they are better able to moderate the effects of the swarm, which also drives down the locust population in the north, where birds are found at higher densities. In the south, locusts swarms are more intense, and habitat destruction combined with predation has reduced the number of birds inhabiting the southern portion of the island. The low bird density in the south is insufficient to moderate the effects of the swarms being produced there. Locust swarms have also reduced available food sources, by denuding the fruit trees of bark which serve as part of the Socorro mockingbird diet. Martínez-Gómez (2005) attributed the greater and continued intensity of swarms in the south to the combination of habitat degradation (which created unsuitable habitat for the birds) and predation by cats (which reduced the number of birds). We consider sheep overgrazing to be a factor contributing to the endangerment of this species.

Summary of Factor A

The current range of the Socorro mockingbird is limited to an estimated 6-mi^2 (15-km^2) area. Habitat has been altered by construction of the Naval base, sheep overgrazing and locust swarms, compounded by predation (Factor C). Locust swarms have reduced available food sources by denuding the fruit trees of bark. Preferring undisturbed montane habitat and primary forest, these factors have created unsuitable conditions for the species. Overgrazing and locust swarms continue to threaten the Socorro mockingbird. We believe that the Socorro mockingbird is at significant risk throughout its range due to the present and ongoing destruction and modification of its habitat.

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

There is no information indicating that the Socorro mockingbird is being utilized for commercial, recreational, scientific, or educational purposes. The species is not known to be in international trade and has not been formally considered for listing under CITES (www.cites.org).

C. Disease or Predation

We are not aware of any disease concerns that may have led to the decline of the Socorro mockingbird species.

Predation by native red-tailed hawks (*Buteo jamaicensis soccoroensis*) and introduced feral cats is a factor in the species' decline. The red-tailed hawk is one of two native raptors on the island; the other is the elf owl (*Micrathene whitneyi graysoni*), a small insectivore. On the mainland, red-tailed hawks eat primarily mammals; however, on Socorro Island their prey consists primarily of birds, land crabs, and lizards (Jehl & Parkes 1983; Wehtje *et al.* 1993). In addition, hawks have been known to prey on adults of other species on the island (Martínez-Gómex & Curry 1995). Martínez-Gómez and Curry (1995) concluded that nesting birds and adult Socorro mockingbirds were vulnerable to predation by red-tailed hawks.

Cats: During their banding study in 1994, Martínez-Gómez and Curry (1995) reported that hawks and feral cats were likely predators of this species. Cats were introduced to the island in 1972 (Martínez-Gómez 2002; Martínez-Gómez *et al.* 2001). Cat predation is considered the major factor responsible for extirpation of the Socorro dove (*Zenaida graysoni*) (Jehl & Parkes 1983). Examinations of cat stomach contents and scats found no substantive evidence of Socorro mockingbird remains. However, Curry (in litt. February 2007) and Martínez-Gómez (2002, 2005) consider that, while feral cats are not the primary reason for the Socorro mockingbird's decline, in combination with habitat degradation caused by sheep, predation by cats is contributing to its decline. Socorro mockingbird fledglings, which are unable to fly for several days after leaving the nest, and ground-foraging adults are vulnerable to predation by feral cats (Martínez-Gómez & Curry 1995, 1996).

According to the Center for Tropical Research in Ecology, Agriculture, and Development (CenTREAD) (2007), eradication of feral cats from Socorro Island is listed as a primary goal in the draft management plan for the Biosphere Reserve (CenTREAD 2007). In 2001, Grupo de Ecología y Conservación de Islas, A.C. (GECI), received a North American Wetlands Conservation Act grant to initiate the eradication of introduced mammals (including rabbits, pigs and sheep) from neighboring Clarión Island and to initiate the eradication of cats and sheep from Socorro Island (Sánchez and Tershy 2001). The work on Clarión Island was completed (CenTREAD 2007). However, the work on Socorro Island may prove to be lengthy and daunting. Dr. Bernie Tershy of the Institute for Marine Sciences (University of California, Santa Cruz, California), a primary researcher involved in the eradication programs on Clarión and Socorro Islands, worked with others to review the documented cases of feral cat eradications on islands and found only 48 examples (Nogales *et al.* 2003). Socorro Island has an area of 54 mi^2 (140 km^2) (Walter 1990) and there are few examples of eradications on larger islands. Of the 48 examples reviewed by Nogales *et al.* (2003), most were conducted on islands smaller than 2 mi^2 (5 km^2) and only a few on islands larger than 6 mi^2 (15 km^2). One successful eradication program on a larger island (Marion Island, Republic of South Africa; area: 112 mi^2 (290 km^2)) took place over a 15-year period. The removal process becomes more complicated when humans occupy the island, because preventing reintroduction of invasive species also becomes a factor (Nogales *et al.* 2003).

Other predators: Feral house mice (*Mus musculus*), on the other hand, already present on the island, pose no known threat to the species (R. Curry in litt. February 2007). Curry (in litt. February 2007) considers the potential accidental introduction of feral black rats (*Rattus rattus*) by Naval transport to be a grave potential threat to the Socorro mockingbird, considering this risk as potentially devastating as the threat of genetic erosion. Such an introduction has not yet occurred and, as such, we do not consider predation by rats to be a factor endangering the species.

Summary of Factor C

Predation by native hawks and feral cats does not appear to be the primary factor causing this species' decline at this time. However, in combination with the threat from habitat degradation (Factor A) and the species' small population size (Factor E), predation is contributing to the endangerment of the species.

D. The Inadequacy of Existing Regulatory Mechanisms

The General Law of Ecological Equilibrium and Environmental Protection was enacted on March 1, 1988, and was amended by Decree published December 13, 1996, and another Decree published January 7, 2000 (General Law of Ecological Equilibrium and Environmental Protection 2000). This law and its amendments: (1) Established the authority to designate protected natural areas to safeguard the genetic diversity of wild species and to preserve species that are in danger of extinction, are threatened endemics, or are rare, and those that need special protection (Article 45); (2) prohibit hunting or exploitation of species within core areas of biosphere reserves (Article 70); (3) specify that use of natural resources in habitats for endemic, threatened, or endangered species must be done in a manner that does not alter the conditions necessary for their survival, development, and evolution (Article 83); (4) prohibit the unpermitted use of threatened and endangered species (Article 87); and (5) stipulate penalties for violation, including fines equivalent to 20 to 20,000 days of the general minimum wage effective in the Federal District at the time the sanction is imposed, confiscation of instruments related to violations, suspension or revocation of permits, and administrative arrest for up to 36 hours (Article 171). While this overarching environmental law aims to protect threatened and endangered species, there are no specific provisions in the law that address the threats to the Socorro mockingbird (i.e., habitat degradation from introduced mammals, habitat destruction (Factor A), and predation (Factor C)).

According to the national legislation NOM–059-ECOL–2001, the species is categorized as ''*Peligro*,'' meaning it is in danger of extinction (H. Benítez Díaz in litt. February 2007). Under Mexico's Wildlife Law (Ley General De Vida Silvestre 2002), it is illegal to kill, possess, transport, or trade in species in danger of extinction without a permit (Article 122). As overutilization is not a threat to the viability of the species, this regulation is of little consequence to the viability of the Socorro mockingbird.

On June 4, 1994, the Mexican government established the Revillagigedo Archipelago Biosphere Reserve and declared it to be a Protected Natural Area (Revillagigedo Archipelago Decree 1994). This reserve included the entire island of Socorro and established the following protections: (1) Formulation of a management plan that sets specific objectives for the reserve (Articles 2 and 3), (2) ban on construction inside core areas of the reserve (which includes the entire island of Socorro) (Article 4), (3) requirement of an environmental impact statement for construction in the buffer zones of the reserve, (4) ban on the establishment of new human settlements within the reserve (Article 7), (5) establishment of a ''closed season'' on all plants and animals in the reserve (Article 9), (6) prohibition on the dumping or discharge of contaminants (Article 11), and (7) limit on recreational activities to those identified in the management plan for the reserve (Article 15). According to the Comisión Nacional de Áreas Naturales Protegidas (n.d.), a management plan has been drafted and is in the process of being published. Management recommendations include: Eradicate cats and sheep from the island; restore the soil and vegetation; and establish a research monitoring station, especially to monitor the population before and after eradications (BLI 2007f). If this management plan is finalized and enacted, this regulatory mechanism has the potential to reduce or remove threats to habitat and from predation and could ultimately result in the recovery of the species. However, based on the best available information at this time, we have no assurances that the management plan will be completed, implemented, and effective. Therefore, this regulatory mechanism is inadequate in reducing the threats to this species.

Summary of Factor D

Regulatory mechanisms are inadequate to reduce the threats to the species, habitat destruction (Factor A) and predation (Factor C). As such, we believe that the inadequacy of regulatory mechanisms is a contributory risk factor that endangers the species.

E. Other Natural or Manmade Factors Affecting the Continued Existence of the Species

Three additional factors are considered herein, genetic risks associated with small population sizes, hybridization, and threats from stochastic events.

Genetic risks associated with small population sizes: The small estimated size of the population, between 298 and 408 individuals (Martínez-Gómez & Curry 1996) exposes this species to any of several risks, including inbreeding depression, loss of genetic variation, and accumulation of new mutations. Inbreeding can have individual or population-level consequences either by increasing the phenotypic expression of recessive, deleterious alleles or by reducing the overall fitness of individuals in the population (Charlesworth & Charlesworth 1987). Small, isolated populations of wildlife species are also susceptible to demographic problems (Shaffer 1981), which may include reduced reproductive success of individuals and chance disequilibrium of sex ratios. In the absence of more species-specific life history data, a general approximation of minimum viable population sizes is referred to as the 50 / 500 rule (Soulé 1980; Hunter 1996), as described under Factor E for the black stilt. The available information indicates that the population of the Socorro mockingbird may be as small as 298 birds (J. Martínez-Gómez in litt. via CONABIO February 2007); this is above the minimum effective population size required to avoid risks from inbreeding (N_e = 50). However, the upper limit of the population estimate of no more than 408 birds (J. Martínez-Gómez in litt. via CONABIO February 2007) is near the upper threshold for N_e = 500). Martínez-Gómez (2002) notes that the species currently exhibits a positive reproductive rate, but that demographic problems will ensue for this species within the next 20 to 30 years, should habitat degradation continue. We conclude that, combined with the threats from habitat destruction (Factor A) and predation (Factor C), this population is vulnerable to genetic risks associated with small population sizes that negatively impact the species' long-term viability.

Hybridization: In addition, the potential for the Socorro mockingbird to hybridize with the northern mockingbird (*Mimus polyglottos*) was brought to our attention by Dr. Curry (in litt. February 2007). The northern mockingbird (Mimus polyglottos) arrived on the Island in 1978, either naturally or transported by Naval personnel (Curry in litt. February 2007), and its population has steadily increased (Jehl & Parkes 1983). Jehl and Parkes (1983) showed that the northern mockingbird's habitat requirements are different from those of the Socorro mockingbird and the northern mockingbird, concluding that the northern mockingbird is not competitively excluding the Socorro mockingbird. They found that the northern mockingbird's success on the island was due to its ability to adapt to the island's degraded habitat. However, it was recently determined that the northern mockingbird is genetically most closely related to the Socorro mockingbird (Arbogast *et al.* 2006; Barber *et al.* 2004), which increases the possibility that the two species are capable of hybridizing (R. Curry in litt. February 2007). In addition, Baptista and Martínez-Gómez (2002) noted that song development in Socorro mockingbird may be being influenced by contact with northern mockingbirds. Interspecific mimicry could facilitate hybridization through sexual misimprinting (R. Curry in litt. February 2007).

We recognize that hybridization can lead to genetic dilution and other genetic risks that undermine the genetic integrity of a species. There is currently no evidence that hybridization has occurred between the Socorro mockingbird and the northern mockingbird. As such, we do not consider this a current factor endangering the species.

Threats from stochastic events: Socorro Island is situated in a zone with a high probability of being in the trajectory of cyclones from the Pacific northeast, which form during the months of May to October. Since 1958, 77 hurricanes and eight tropical storms have hit the Island chain (Comisión Nacional de Áreas Naturales Protegidas (CONANP) n.d.). In 1997, Hurricane Linda came within 46 mi (74 km; 40 nautical miles (nm)) of the island, where it reportedly ''wreaked havoc'' (Wirth 1998). At 160 knots, it was the strongest hurricane recorded in the Pacific since recordkeeping began in 1949 (Lawrence 1999).

Socorro Island is a volcanic island. The most recent eruption of Mt. Evermann occurred in 1993, from an underwater vent off the southwest coast. Regular volcanic activity continues throughout the Island from fumaroles and hydrothermal vents (Bulletin of the Global Volcanism Network 1993). The last major volcanic eruption on Socorro Island occurred in 1948 (CONANP n.d.) and, according to Trombley (2007), the next is expected in 2014. An eruption in 1952 on San Benedicto decimated the native flora and fauna on that island (Martínez-Gómez 2002).

Stochastic events, such as hurricanes and volcanic eruptions, could result in extensive mortalities from which the population may be unable to recover, leading to extinction. Increased population fragmentation in combination with these factors increases the likelihood of extinction of the species through a single stochastic event (Caughley 1994; Charlesworth & Charlesworth 1987).

Summary of Factor E

Combined with the population pressures caused by habitat loss (Factor A) and predation (Factor C), the Socorro mockingbird is subject to long-term genetic risks associated with its small population and compounded by the risk of stochastic events, such as cyclones or eruptions, severely reducing population numbers such that the species is unable to recover. We consider the species' small population size and threats from stochastic events threats that contribute to the endangerment of the species.

Conclusion and Determination for the Socorro Mockingbird

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the black stilt, above. We have determined that the species is in danger of extinction throughout all of its known range primarily due to ongoing threats to its habitats (Factor A) and predation (Factor C), compounded by genetic risks to the species' long-term genetic viability and susceptibility to stochastic events due to risks associated small population sizes (Factor E). Furthermore, we have determined that the inadequacy of existing regulatory mechanisms is a contributory risk factor that endangers the species' continued existence (Factor D). Therefore, we are determining endangered status for the Socorro mockingbird under the Act. Because we find that the Socorro mockingbird is endangered throughout all of its range, there is no reason to consider its status in any significant portion of its range.

Required Determinations

Available Conservation Measures

Conservation measures provided to species listed as endangered or threatened under the Act include recognition, recovery actions, requirements for Federal protection, and prohibitions against certain practices. Recognition through listing results in public awareness and encourages and results in conservation actions by Federal governments, private agencies and groups, and individuals.

Section 7(a) of the Act, as amended, and as implemented by regulations at 50 CFR part 402, requires Federal agencies to evaluate their actions within the United States or on the high seas with respect to any species that is proposed or listed as endangered or threatened, and with respect to its critical habitat, if any is being designated. However, given that the black stilt, caerulean paradise-flycatcher, giant ibis, Gurney's pitta, Long-legged thicketbird, and Socorro mockingbird are not native to the United States, no critical habitat is being proposed for designation with this rule.

Section 8(a) of the Act authorizes the provision of limited financial assistance for the development and management of programs that the Secretary of the Interior determines to be necessary or useful for the conservation of endangered species in foreign countries. Sections 8(b) and 8(c) of the Act authorize the Secretary to encourage conservation programs for foreign endangered species and to provide assistance for such programs in the form of personnel and the training of personnel.

The Act and its implementing regulations set forth a series of general prohibitions and exceptions that apply to all endangered wildlife. As such, these prohibitions would be applicable to the black stilt, caerulean paradise-flycatcher, giant ibis, Gurney's pitta, Long-legged thicketbird, and Socorro mockingbird. These prohibitions, pursuant to 50 CFR 17.21, in part, make it illegal for any person subject to U.S. jurisdiction to "take" (includes harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or to attempt any of these) within the United States or upon the high seas; import or export; deliver, receive, carry, transport, or ship in interstate or foreign commerce in the course of commercial activity; or sell or offer for sale in interstate or foreign commerce any endangered wildlife species. It also is illegal to possess, sell, deliver, carry, transport, or ship any such wildlife that has been taken in violation of the Act. Certain exceptions apply to agents of the Service and State conservation agencies.

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

Paperwork Reduction Act

This final rule does not contain any new collections of information that require approval by the Office of Management and Budget (OMB) under 44 U.S.C. 3501 *et seq.* The regulation will not impose new recordkeeping or reporting requirements on State or local governments, individuals, businesses, or organizations. We may not conduct or sponsor and you are not required to respond to a collection of information unless it displays a currently valid OMB control number.

National Environmental Policy Act

We have determined that environmental assessments and environmental impact statements, as defined under the authority of the National Environmental Policy Act of 1969, need not be prepared in connection with regulations adopted pursuant to section 4(a) of the Act. A notice outlining our reasons for this determination was published in the **Federal Register** on October 25, 1983 (48 FR 49244).

References Cited

A list of the references used to develop this final rule is available upon request (see **ADDRESSES** section).

Author

The primary author of this notice is the staff of the Division of Scientific Authority, U.S. Fish and Wildlife Service (see **ADDRESSES** section).

List of Subjects in 50 CFR Part 17

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

Regulation Promulgation

■ Accordingly, we amend part 17, subchapter B of chapter I, title 50 of the Code of Federal Regulations, as follows:

PART 17—[AMENDED]

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

Authority: 16 U.S.C. 1361–1407; 16 U.S.C. 1531–1544; 16 U.S.C. 4201–4245; Pub. L. 99–625, 100 Stat. 3500; unless otherwise noted.

■ 2. Amend 17.11(h) by adding new entries for "Ibis, giant," "Mockingbird, Socorro," "Paradise-flycatcher, caerulean," "Pitta, Gurney's," "Stilt, black," and "Thicketbird, long-legged" in alphabetical order under Birds, to the List of Endangered and Threatened Wildlife as follows:

§ 17.11 Endangered and threatened wildlife.

* * * * *

(h) * * *

Species		Historic range	Vertebrate population where endangered or threatened	Status	When listed	Critical habitat	Special rules
Common name	Scientific name						
*	*	*	*	*	*	*	
BIRDS							

Species		Historic range	Vertebrate population where endangered or threatened	Status	When listed	Critical habitat	Special rules
Common name	Scientific name						
*	*	*	*	*	*	*	
Ibis, giant	*Pseudibis gigantea*	Cambodia, Lao PDR, Thailand, Vietnam.	Entire	E	760	NA	NA
*	*	*	*	*	*	*	
Mockingbird, Socorro	*Mimus Graysoni*	Mexico	Entire	E	760	NA	NA
*	*	*	*	*	*	*	
Paradise-flycatcher, caerulean.	*Eutrichomyias rowleyi*	Indonesia	Entire	E	760	NA	NA
*	*	*	*	*	*	*	
Pitta, Gurney's	*Pitta gurneyi*	*Myanmar, Thailand*	Entire	E	760	NA	NA
*	*	*	*	*	*	*	
Stilt, black	*Himantopus novaezelandiae*.	New Zealand	Entire	E	760	NA	NA
*	*	*	*	*	*	*	
Thicketbird, long-legged.	*Trichocichla rufa*	Fiji	Entire	E	760	NA	NA
*	*	*	*	*	*	*	

Dated: January 7, 2008.

Kenneth Stansell,

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

[FR Doc. E8–492 Filed 1–15–08; 8:45 am]

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