DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

50 CFR Part 17

[96000–1671–0000–B6]

Endangered and Threatened Wildlife and Plants; Annual Notice of Findings on Resubmitted Petitions for Foreign Species; Annual Description of Progress on Listing Actions

AGENCY: Fish and Wildlife Service, Interior.

ACTION: Notice of review.

SUMMARY: In this notice of review, we announce our annual petition findings for foreign species, as required under section 4(b)(3)(C)(i) of the Endangered Species Act of 1973, as amended. When, in response to a petition, we find that listing a species is warranted but precluded, we must complete a new status review each year until we publish a proposed rule or make a determination that listing is not warranted. These subsequent status reviews and the accompanying 12-month findings are referred to as ''resubmitted'' petition findings.

Information contained in this notice describes our status review of 50 foreign taxa that were the subjects of previous warranted-but-precluded findings, most recently summarized in our 2007 Notice of Review (72 FR 20184). Based on our current review, we find that 20 species (see Table 1) continue to warrant listing, but that their listing remains precluded by higher-priority listing actions. For 30 species previously found to be warranted but precluded, the petitioned action is now warranted. We will promptly publish listing proposals for those 30 species (see Table 1).

With this annual notice of review (ANOR), we are requesting additional status information for the 20 taxa that remain warranted but precluded by higher priority listing actions. We will consider this information in preparing listing documents and future resubmitted petition findings for these 20 taxa. This information will also help us to monitor the status of the taxa and in conserving them.

DATES: We will accept comments on these resubmitted petition findings at any time.

ADDRESSES: Submit any comments, information, and questions by mail to the Chief, Division of Scientific Authority, U.S. Fish and Wildlife Service, 4401 N. Fairfax Drive, Room 110, Arlington, Virginia 22203; by fax to 703–358–2276; or by e-mail to *ScientificAuthority@fws.gov*. Comments and supporting information will be available for public inspection, by appointment, Monday through Friday from 8 a.m. to 4 p.m. at the above address.

FOR FURTHER INFORMATION CONTACT: Mary M. Cogliano, PhD, at the above address or by telephone 703–358–1708; fax, 703–358–2276; or e-mail, *ScientificAuthority@fws.gov*.

SUPPLEMENTARY INFORMATION:

Background

The Endangered Species Act of 1973, as amended (Act) (16 U.S.C. 1531 *et seq.*), provides two mechanisms for considering species for listing. First, we can identify and propose for listing those species that are endangered or threatened based on the factors contained in section 4(a)(1). We implement this through the candidate program. Candidate taxa are those taxa for which we have sufficient information on file relating to biological vulnerability and threats to support a proposal to list the taxa as endangered or threatened, but for which preparation and publication of a proposed rule is precluded by higher-priority listing actions. None of the species covered by this notice were assessed through the candidate program; they were the result of public petitions to add species to the Lists of Endangered and Threatened Wildlife and Plants (Lists), which is the other mechanism for considering species for listing.

Under section 4(b)(3)(A) of the Act, when we receive a listing petition, we must determine within 90 days, to the maximum extent practicable, whether the petition presents substantial scientific or commercial information indicating that the petitioned action may be warranted (90-day finding). If we make a positive 90-day finding, we are required to promptly commence a review of the status of the species, whereby, in accordance with section 4(b)(3)(B) of the Act we must make one of three findings within 12 months of the receipt of the petition (12-month finding). The first possible 12-month finding is that listing is not warranted, in which case we need not take any further action on the petition. The second possibility is that we may find that listing is warranted, in which case we must promptly publish a proposed rule to list the species. Once we publish a proposed rule for a species, sections 4(b)(5) and 4(b)(6) govern further procedures, regardless of whether or not we issued the proposal in response to the petition. The third possibility is that we may find that listing is warranted but precluded. A warranted-but-precluded finding on a petition to list means that listing is warranted, but that the immediate proposal and timely promulgation of a final regulation is precluded by higher priority listing actions. In making a warranted-but precluded finding under the Act, the Service must demonstrate that expeditious progress is being made to add and remove species from the lists of endangered and threatened wildlife and plants.

Pursuant to section 4(b)(3)(C)(i) of the Act, when, in response to a petition, we find that listing a species is warranted but precluded, we must make a new 12-month finding annually until we publish a proposed rule or make a determination that listing is not warranted. These subsequent 12-month findings are referred to as ''resubmitted'' petition findings. This notice contains our resubmitted petition findings for all foreign species previously described in the 2007 Notice of Review (72 FR 20184) and that are currently the subject of outstanding petitions.

Previous Notices

The species discussed in this notice were the result of three separate petitions submitted to the U.S. Fish and Wildlife Service (Service) to list a number of foreign bird and butterfly species as threatened or endangered under the Act. We received petitions to list foreign bird species on November 24, 1980, and May 6, 1991 (46 FR 26464 and 56 FR 65207, respectively). On January 10, 1994, we received a petition to list 7 butterfly species as threatened or endangered (59 FR 24117).

We took several actions on these petitions. To notify the public on these actions, we published petition findings, listing rules, status reviews, and petition finding reviews that included foreign species in the **Federal Register** on May 12, 1981 (46 FR 26464); January 20, 1984 (49 FR 2485); May 10, 1985 (50 FR 19761); January 9, 1986 (51 FR 996); July 7, 1988 (53 FR 25511); December 29, 1988 (53 FR 52746); April 25, 1990 (55 FR 17475); September 28, 1990 (55 FR 39858); November 21, 1991 (56 FR 58664); December 16, 1991 (56 FR 65207); March 28, 1994 (59 FR 14496); May 10, 1994 (59 FR 24117); January 12, 1995 (60 FR 2899); and May 21, 2004 (69 FR 29354). Our most recent review of petition findings was published on April 23, 2007 (72 FR 20184).

Since our last review of petition findings, we have taken two listing actions related to this notice (see Preclusion and Expeditious Progress section for additional listing actions that were not related to this notice). On

December 17, 2007, we published a proposed rule to list 6 species of foreign Procellariids under the Act (72 FR 71298). We also published a final rule on January 16, 2008, to list 6 foreign bird species as endangered under the Act (73 FR 3146).

Findings on Resubmitted Petitions

This notice describes our resubmitted petition findings for 50 foreign species for which we had previously found proposed listing to be warranted but precluded. We have considered all of the new information that we have obtained since the previous findings, and we have updated the listing priority number (LPN) of each taxon for which proposed listing continues to be warranted but precluded, in accordance with our Listing Priority Guidance published September 21, 1983 (48 FR 43098). Such a priority ranking guidance system is required under section 4(h)(3) of the Act. Using this guidance, we assign each taxon an LPN of 1 to 12, whereby we first categorize based on the magnitude of the threat(s) (high versus moderate-to-low), then by the immediacy of the threat(s) (imminent versus nonimminent), and finally by taxonomic status; the lower the listing priority number, the higher the listing priority (i.e., a species with an LPN of 1 would have the highest listing priority).

As a result of our review of 50 foreign species, we find that warranted-but-precluded findings remain appropriate for 20 species. We emphasize that we are not proposing these species for listing by this notice, but we do anticipate developing and publishing proposed listing rules for these species in the future, with an objective of making expeditious progress in addressing all 20 of these foreign species within a reasonable timeframe.

Also as a result of this review, we find that proposing 30 taxa for listing under the Act is warranted. We will promptly publish proposals to list these 30 taxa, listed below in taxonomic order: Junín flightless grebe (*Podiceps taczanowskii*), greater adjutant stork (*Leptoptilos dubius*), Andean flamingo (*Phoenicoparrus andinus*), Brazilian merganser (*Mergus octosetaceus*), Caucau Guan (*Crax alberti*), blue-billed curassow (*Penelope perspicax*), Cantabrian capercaillie (*Tetrao urogallus cantabricus*), gorgeted wood-quail (*Odontophorus strophium*), Junín rail (*Laterallus tuerosi*), Jerdon's Courser (*Rhinoptilus bitorquatus*), slender billed curlew (*Numenius tenuirostris*), Marquesan imperial pigeon (*Ducula galeata*), salmon-crested cockatoo (*Cacatua moluccensis*), southeastern rufous-vented ground-cuckoo (*Neomorphus geoffroyi dulcis*), Margaretta's hermit (*Phaethornis malaris margarettae*), black-breasted puffleg (*Eriocnemis nigrivestis*), Chilean woodstar (*Eulidia yarrellii*), Esmeraldas woodstar (*Chaetocerus berlepschi*), royal cinclodes (*Cinclodes aricomae*), white-browed tit-spinetail (*Leptasthenura xenothorax*), black-hooded antwren (*Formicivora erythronotos*), fringe-backed fire-eye (*Pyriglena atra*), brown-banded antpitta (*Grallaria milleri*), Kaempfer's tody-tyrant (*Hemitriccus kaempferi*), ash-breasted tit-tyrant (*Anairetes alpinus*), Peruvian plantcutter (*Phytotoma raimondii*), St. Lucia forest thrush (*Cichlherminia herminieri sanctaeluciae*), Eiao Polynesian warbler (*Acrocephalus cafier aquilonis*), medium tree-finch (*Camarhynchus pauper*), and cherry-throated tanager (*Nemosia rourei*).

Our warranted finding is based on a species' LPN, as well as a recent court order. We have found all taxa with LPNs of 2 or 3, as reported in the 2007 Notice of Review (72 FR 20184), to be warranted for proposed listing under the Act, because these species face threats that are both imminent and high in magnitude. In addition to the LPN directing our findings, on January 23, 2008, the United States District Court ordered the Service to propose listing rules for five foreign bird species, actions which had been previously determined to be warranted but precluded: the Chilean woodstar (*Eulidia yarrellii*), Andean flamingo (*Phoenicoparrus andinus*), medium tree-finch (*Camarhynchus pauper*), black-breasted puffleg (*Eriocnemis nigrivestis*), and the St. Lucia forest thrush (*Cichlherminia herminieri sanctaeluciae*). Of these five species, only one, the medium tree-finch (*Camarhynchus pauper*), did not have an LPN number of 2 or 3. To comply with the court-order, however, we are declaring the medium tree-finch to be warranted for proposed listing at this time, in addition to the 29 species that were reported with LPNs of 2 or 3 in our 2007 Notice of Review, for which we have already begun to prepare proposed listing rules.

Based on our review of 50 species, we did not find any taxa to be no longer warranted for listing. Table 1 provides a summary of all updated determinations of the 50 taxa in our review. Any changes in LPN are explained in the species summaries in the text of this notice. Taxa in Table 1 of this notice are assigned to two status categories, noted in the ''Category'' column at the left side of the table. We identify the taxa for which we find that listing is warranted but precluded by a ''C'' in the category column, referring to these taxa as ''candidates'' under the Act. The other category is for those species for which we find that proposed listing is warranted, and we designate these taxa with a ''P,'' indicating that proposed rules to list these taxa under the Act will be published promptly. The column labeled ''Priority'' indicates the LPN for all taxa for which proposed listing is warranted but precluded. Following the scientific name of each taxon (third column) is the family designation (fourth column) and the common name, if one exists (fifth column). The sixth column provides the known historic range for the taxon. The avian species in Table 1 are listed taxonomically.

Findings on Species for Which Listing Is Warranted

Below are our 12-month resubmitted petition findings on the 30 taxa found by this notice to be warranted for proposed listing under the Act.

Birds

Junín Flightless Grebe (*Podiceps taczanowskii*)

The Junín flightless grebe is endemic to Lake Junon, a large lake that covers 35,385 acres (ac) (14,320 hectares (ha)) in the central Andes of Peru at 13,386 feet (ft) (4,080 meters (m)) above sea level (Fjeldså 1981; Fjeldså 2004; Fjeldså and Krabbe 1990; INRENA 1996). Historically, the species was likely distributed throughout the lake, but it is now absent from the northwest portion of the lake due to contamination from mining wastes (Fjeldså 1981).

The lake is bordered by extensive reed marshes and reaches a depth of 32.8 ft (10 m) at the center. The reed marshes are continuous in some areas of the lake shore, but they form a mosaic with stretches of open water in other areas. Considerable stretches of the lake are shallow, supporting dense growth of stonewort (*Chara* spp.) (del Hoyo et al. 1992). The Junín flightless grebe prefers open lake habitat and remains in the center of the lake when it is not breeding. During the breeding season, however, it nests in stands of tall *Scirpus californicus tatora* or bays and channels along the outer edge of the reed marshes surrounding the lake (O'Donnel and Fjedså 1997). The Junín flightless grebe feeds predominantly on fish (*Orestias spp.*), which constitute approximately 90 percent of its diet (del Hoyo et al. 1992).

The Junín flightless grebe has experienced dramatic population declines since the early 1960s when there were at least 1,000 individuals (F. Gill and R.W. Storer, as cited in Fjeldså 2004). Prior to the 1960s, the Junín flightless grebe had been described as ''extremely abundant on the lake'' (Morrison 1939). However, by 1979, the population was estimated to be 250 to 300 birds, indicating a rapid and extensive decline (Harris 1981, as cited in O'Donnell and Fjeldså 1997). From 1979 through 2004, population estimates fluctuated between 50 to 375 birds (J. Fjeldså 2005, as cited in Butchart et al. 2006; O'Donnel and Fjeldså 1997). In 2004, the population estimate was 100 to 300 birds (BirdLife International 2007); however, in dry years (*e.g.*, 1983–1987, 1991, 1994–1997), the population was reduced to 100 birds or fewer (Elton 2000; Fjeldså 2004). Short-term population increases ranging from 200 to 300 birds have occurred in years with high rainfall levels related to the El Niño Southern-Oscillation (ENSO) (1997–1998 and 2001–2002) (T. Valqui and PROFONANPE 2002, as cited in Fjeldså 2004). In 2007, the population once more declined due to a high-mortality weather event (Hirschfeld 2007).

The Junín flightless grebe is considered ''Critically Endangered'' by the IUCN (International Union for Conservation of Nature) Red List because of the species' rapid decline, highly restricted range, and increasing exposure to contaminants produced by the mining industry (Birdlife International 2006). Variations in lake water levels of up to 23 ft (7 m) at a time are linked to electrical power generation by a local hydroelectric power station. These water-level fluctuations have reduced prey populations, resulting in increased food competition with white-tufted grebes (*Rollandia rolland*). Frequent manipulation and drawdowns of the lake's water level also prevent foraging, nest building, and breeding in drought years (BirdLife International 2007). In addition, contamination from mining wastes (Fjeldså 1981; Martin and McNee 1999) has reduced the amount of available habitat in the northern section of the lake by diminishing or eliminating stands of submerged aquatic vegetation (Fjeldså 2004; ParksWatch 2006). Greater concentration of contaminants in the lake as a result of droughts (T. Valqui and J. Barrio in litt. 1992, as cited in Collar et al. 1992) has coincided with mortality of Junín flightless grebes (T. Valqui and J. Barrio in litt. 1992, as cited in Collar et al. 1992), and is believed either to have directly caused the mortalities or to have resulted in mortality of the grebes by reducing their prey (Fjeldså 2004). Threats to this species and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

Greater Adjutant Stork (*Leptoptilos dubius*)

The current range of the greater adjutant stork consists of two breeding populations, one in India and the other in Cambodia. Recent sighting records of this species from the neighboring countries of Nepal, Bangladesh, Vietnam, and Thailand are presumed to be wandering birds from one of the two populations in India or Cambodia (Birdlife International 2007).

The greater adjutant stork frequents marshes, lakes, paddy fields, and open forest, and may also be found in dry areas, such as grasslands and fields. In India, much of the native habitat has been lost. The greater adjutant stork often occurs close to urban areas, feeding in and around wetlands in the breeding season, and disperses to feed on carcasses and to scavenge at trash dumps, burial grounds, and slaughter houses at other times of the year. The natural diet of the greater adjutant stork consists primarily of fish, frogs, reptiles, small mammals and birds, crustaceans, and carrion (BirdLife International 2007; Singha and Rahman 2006).

This species breeds in colonies during the dry season (winter) in stands of tall trees near water sources. In India, the breeding sites are commonly associated with bamboo forests which provide protection from wind (Singha et al. 2002). The greater adjutant stork constructs platform nests made of sticks in the upper lateral limbs of large trees (Singha et al. 2002). In Cambodia, the greater adjutant stork breeds in freshwater flooded forest and disperses to seasonally inundated forest, tall wet grasslands, mangroves, and intertidal flats to forage. At the Kulen Promtep Wildlife Sanctuary, it is known to nest only in evergreen forests (Clements et al. 2007b). At two breeding sites near the city of Guwahati in the State of Assam, the most recent survey data show that the number of breeding birds has declined from 247 birds in 2005 to 118 birds in 2007 (Hindu 2007).

During the nineteenth century, there were vast colonies of millions of greater adjutant storks in Burma, and del Hoyo et al. (1992) noted that in Calcutta there was ''almost one [stork] on every roof.'' However, during the twentieth century the species experienced a rapid decline, and currently the population estimate is 800 to 1,000 birds in two very small and highly disjunct breeding populations (BirdLife International 2007). The greater adjutant stork is classified as ''Endangered'' by the IUCN Red List (BirdLife International 2007).

Identified risks to this species include habitat destruction, particularly lowland deforestation and the felling of nest trees (Hindu 2007; Singha et al. 2002; Singha et al. 2006; WCS 2007); habitat modification from flooding and hydrological changes brought about by Mekong River dam development (Clements et al. 2007b; WCS 2007); direct exploitation, such as hunting and egg collection from nesting colonies (Clements et al. 2007a); and drainage, agricultural conversion, pollution, and over-exploitation of wetlands (BirdLife International 2007; Clements et al. 2007; Singha et al. 2003). The Assam population is also negatively impacted by the loss of a readily available food source, due to the reduced number of open rubbish dumps for the disposal of carcasses and foodstuffs (BirdLife International 2007). Threats to this species and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Andean Flamingo (*Phoenicoparrus andinus*)

The Andean flamingo is the rarest of six flamingo species worldwide and one of three endemic to the high Andes of South America (Arengo in litt. 2007; Caziani et al. 2007; del Hoyo et al. 1992; Johnson et al. 1958; Johnson 1967; Line 2004). The Andean flamingo is found in lakes in the Andean altiplano (high plains) from southern Peru and southwestern Bolivia to northern Chile and northwest Argentina. A small section of the population winters in the lowlands of central Argentina, mainly at Mar Chiquita Lake (Blake 1977; Bucher 1992; Boyle et al. 2004; Caziani et al. 2006; Caziani et al. 2007; Fjeldså and Krabbe 1990; Hurlbert and Keith 1979; Kahl 1975). There have been several documented occurrences of Andean flamingos in Brazil, but it is unclear whether the species is accidental or a more frequent visitor (Bornschein and Reinert 1996; Sick 1993).

Andean flamingo habitat consists of plankton-rich, high-elevation, shallow lakes and salt flats (Fjeldså and Krabbe 1990). The range of the species becomes more restricted in the winter as low temperatures and aridity seasonally inhibit the suitability of some wetlands (Caziani et al. 2007; Mascitti and Bonaventura 2002). The Andean flamingo feeds in large flocks on diatoms of the genus *Surirella* from the benthic interface in water less than 3 ft (1 m) deep (Hurlbert and Chang 1983; Mascitti and Castañera 2006; Mascitti and Kravetz 2002).

Population assessments for this species vary greatly. In 1967, Charles Cordier estimated the number of Andean flamingos to be 250,000 to 300,000 birds (Johnson 1967). Kahl (1975) reviewed previous estimates and noted that Cordier's 1965 and 1968 population estimates varied by an order of magnitude (from 50,000 to 500,000) during that same time period. By 1986, R. Schlatter estimated the population to be fewer than 50,000 individuals, with a declining population trend (Johnson 2000). However, the accuracy of these early estimates has never been confirmed, making it difficult to establish trends.

Using a comprehensive sampling design and conducting simultaneous surveys at over 200 wetlands in Peru, Bolivia, Chile, and Argentina, Caziani et al. (2007) counted 33,918 Andean flamingos in January 1997; 27,913 in January 1998; 14,722 in June 1998; and 24,442 in July 2000. In the summer of 2005, Caziani et al. (2006) reported 31,617 Andean flamingos distributed throughout 25 wetlands, with 50 percent of the population located in five wetlands in Chile and Bolivia.

Long-lived species with slow rates of reproduction, such as the Andean flamingo, may appear to have robust populations, but can rapidly decline if reproduction does not keep pace with mortality. Andean flamingo recruitment was very low from the late 1980s to the mid-1990s, averaging only 800 chicks per year from 1988 through 1997. Recruitment appears to have improved in recent years, with a total of 13,201 Andean flamingo chicks hatched from 1997 through 2001 (Caziani et al. 2007), and an average of 3,000 chicks per year has fledged since 2000 (Amado et al. 2007 as cited in Arengo in litt. 2007). However, in some years breeding success is extremely limited; in 1997, only 200 chicks were observed to have hatched (Caziani et al. 2007). The reasons for such variation appear to be related to annual climatic conditions (Caziani et al. 2007). When climatic conditions are favorable, breeding takes place, whereas, when climatic conditions are unfavorable breeding is abandoned, very limited, or takes place at alternative breeding grounds, which tend to be less productive (Bucher et al. 2000).

The IUCN categorizes the Andean flamingo as ''Vulnerable'' because it has undergone a rapid population decline, it is exposed to ongoing exploitation and declines in habitat quality, and finally, although previous exploitation has decreased, the longevity and slow breeding of flamingos suggest that the legacy of past threats may persist through future generations (BirdLife International 2007).

Experts consider the greatest threats to the Andean flamingo to be habitat degradation caused by mining, agricultural, and residential/urban development, and tourism (Arengo in litt. 2007). Mining takes place in or near many of the wetlands occupied by the Andean flamingo, including successful breeding sites (Corporación Nacional Forestal 1996a; Soto 1996; Ugarte-Nunez and Mosaurieta-Echegaray 2000). Loss of habitat due to excavations in the lakebed and extraction of water are attributed to mining, which also causes extensive degradation of water quality. Chemical pollution produced by the mining and metallurgical industries and recent petroleum spills are also responsible for the degradation of water resources (OAS/UNEP and ALT 1999, as cited in Rocha 2002). Pollution from mining wastes has been reported as a risk factor to flamingos in Argentina (Laredo 1990 as cited in Administración de Parques Nacionales 1994), although it was not reported whether the risk was due to direct mortality of flamingos or due to a reduction in their food supply. In Chile, where Andean flamingo breeding colonies are concentrated and where mineral and hydrocarbon exploration and exploitation have increased in the last two decades, both the number of successful breeding colonies and the total production of chicks of Andean Flamingos have declined since the 1980s (Parada 1992, Rodríguez and Contreras 1998, as cited in Caziani et al. 2007).

Water consumption for agriculture and domestic use can cause serious declines in water levels at important breeding sites (Messerli et al. 1997), and increased tourism is likely to further stress already tenuous water budgets as hotels and restaurants are established (RIDES 2005). Other potential risks to the species include overutilization of individuals (Valqui et al. 2000) and eggs (Caziani et al. 2007) as a food resource and collection of feathers (Valqui et al. 2000). Threats to the Andean flamingo and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

Brazilian Merganser (*Mergus octosetaceus*)

The Brazilian merganser is a diving duck that occurred historically in riverine habitats throughout southern Brazil, northeastern Argentina, and eastern Paraguay (Hughes et al. 2006). The species is considered extinct in Mato Grosso do Sul, Rio de Janeiro, Sao Paolo, and Santa Catarina (BirdLife International 2007). There is only one recent record of the species from Misiones, Argentina (Benstead 1994; Hearn 1994, as cited in Collar et al. 1994), and it was last recorded in Paraguay in 1984 (BirdLife International 2007).

Currently the species is found in extremely low numbers at six highly disjunct localities, of which five are in southeastern Brazil, and one is in northeastern Argentina and, possibly, extreme eastern Paraguay (BirdLife International 2007; Hughes et al. 2006). The species inhabits shallow clear-water streams and rapid rivers, preferably surrounded by dense tropical forests, and it is believed to be a highly sedentary, monogamous species, presumably maintaining its territory all year (del Hoyo et al. 1992; Bruno et al. 2006; Ducks Unlimited 2007; Hughes et al. 2006). The Brazilian merganser is a good swimmer and diver, and feeds primarily on fish, and occasionally aquatic insects and snails (Collar et al. 1992).

Recent records from Brazil and a newly discovered northern range extension indicate that the status of this species is better than previously considered, as several highly disjunct populations were located in 2002 (BirdLife International 2007; Hughes et al. 2006). However, the IUCN categorizes the species as ''Critically Endangered'' (BirdLife International 2007). Additionally, the population is estimated at between 50 to 249 individuals, and the trend is decreasing (BirdLife International 2007).

Identified risks to the species include habitat loss and degradation, fragmentation, and hydrological changes with perturbation and pollution of rivers, which are predominately the result of deforestation, agriculture, and diamond mining in the Serra da Canastra area (Bianchi et al. 2005; Bartmann 1994 and 1996, as cited in BirdLife International 2007; Bruno et al. 2006; Collar et al. 1994; Ducks Unlimited 2007; Hughes et al. 2006; Lamas and Santos 2004). Each breeding pair of Brazilian mergansers requires relatively long segments of river—up to ca. 7.5 miles (mi) (12 kilometers (km))—and the species is sensitive to human disturbance, including activities associated with expanded human presence such as tourism and scientific research programs (Braz et al. 2003; Bruno et al. 2006). Dam construction has destroyed suitable habitat, especially in Brazil and Paraguay (BirdLife International 2007). The species is highly adapted to shallow, rapid-flowing riverine conditions and, therefore, cannot tolerate the lacustrine (i.e., lake-like) conditions of reservoirs that result from dam-building activities within their occupied range (Hughes et al. 2006).

The Brazilian merganser is legally protected in Brazil, and four of Brazil's protected areas represent the major sites where the species occurs (del Hoyo et al. 1992; Hughes et al. 2006). These sites are critical for protecting some of the key remaining subpopulations of the Brazilian merganser (del Hoyo et al. 1992; Braz et al. 2003; Bianchi et al. 2005; Bruno et al. 2006; BirdLife International 2007). The Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis (IBAMA) in Brazil has established eight committees to develop and monitor conservation strategies for the country's ''endangered'' species, including the Brazilian merganser (Marinia and Garcia 2004). These committees developed an Action Plan for Conservation of the Brazilian Merganser, which has recently been published by the government of Brazil (Hughes et al. 2006). Despite these protections, threats to the Brazilian merganser continue. Therefore, we find that proposing this species for listing under the Act is warranted.

Cauca Guan (*Penelope perspicax*)

The Cauca guan is a medium-sized cracid with a bright red dewlap. It is dull brownish-gray, with mainly chestnut rear parts. It has whitish-scaled feather edges from head to mantle and breast (BirdLife International 2008). The Cauca guan is endemic to the slopes of the west and central Andes (Risaralda, Quindio, Valle del Cauca, and Cauca) in Colombia (Collar et al. 1992). The historic range is estimated to have been approximately 9,614 mi² (24,900 km²) (Renjifo 2002). In the early part of the twentieth century, the Cauca guan inhabited the dry forests of the Cauca, Dagua, and Patía Valleys (Renjifo 2002). Today, most of the dry forests have been eliminated or highly fragmented, such that continuous forest exists only above 6,562 ft (2,000 m) (Renjifo 2002). At the beginning of the twentieth century through the 1950s, the species was considered common (Renjifo 2002; BirdLife International 2007). Between the 1970s and 1980s, there was extensive deforestation in the Cauca Valley, and the species went unobserved during this time, leading researchers to suspect that the Cauca guan was either extinct or on the verge of extinction (Brooks and Strahl 2000; del Hoyo et al. 1994; Hilty 1985; Hilty and Brown 1986). The species was rediscovered in 1987 (Renjifo 2002). In the late 1990s, Ucumarí Regional Park was considered the stronghold of the species (BirdLife International 2007). However, the species has not been observed again in that location since 1995 (Wege and Long 1995).

Cauca guan populations are characterized as small, containing only tens of individuals or, in rare instances, hundreds (Renjifo 2002). BirdLife International (2007) reported that the largest subpopulation contained an estimated 50 to 249 individuals; however, they did not specify to which population this refers, and these figures are not found in any other literature regarding population surveys of the Cauca guan. Kattan et al. (2006) conducted the only two population surveys in 2000 and 2001 (Muñoz et al. 2006). They estimated population densities at two locations—Otún-Quimbaya Flora and Fauna Sanctuary (Risaralda) and Reserva Forestal de Yotoco (Valle de Cauca)—to be between 144 and 264 individuals and 35 to 61 individuals, respectively (Kattan et al. 2006). Kattan et al. (2006) examined 10 additional localities, based on locality data reported by Renjifo (2002). Visual confirmations were made at only 2 of the 10 localities, and auditory confirmations were made at 5 of the 10 localities (Kattan et al. 2006). In 2006, Kattan (in litt., as cited in Muñoz et al. 2006) estimated the global population to be between 196 and 342 individuals. The IUCN categorizes the species as ''Endangered'' due to its small, contracted range, composed of widely fragmented patches of habitat (BirdLife International 2007) and considers the overall population to be in decline (BirdLife International 2007; Kattan 2004; Renjifo 2002). The Cauca guan is listed as ''Endangered'' under Colombian law, which prohibits commercial and sport hunting of the species (ECOLEX 2007). The level of enforcement is uncertain, however, despite this protection. Poaching continues to be a problem for the Cauca guan and may play a role in the possible local extirpation of the species from at least two protected areas (Collar et al. 1992; del Hoyo et al. 1994; Strahl et al. 1995).

Extensive habitat destruction and fragmentation since the 1950s have resulted in an estimated 95 percent range reduction of this species (Chapman 1917; Collar et al. 1992; Kattan et al. 2006; Renjifo 2002; Rios et al. 2006). As a result, although it prefers mature, tropical, humid forests, the Cauca guan exists primarily in fragmented and isolated secondary forest remnants, forest edges, and in plantations of the nonnative Chinese ash trees (Fraxinus chinensis) that are located within 0.62 mi (1 km) of primary forest (Renjifo 2002; Kattan et al. 2006; Rios et al. 2006). Its current range is estimated to be less than 290 mi² (750 km²), of which only 216 mi² (560 km²) is considered suitable habitat (BirdLife International 2007; Kattan et al. 2006; Rios et al. 2006). It is estimated that more than 30 percent of this loss of habitat has occurred within the species' last 3 generations (30 years) (Renjifo 2002), and recent studies indicate that the rate of habitat destruction is accelerating (Butler 2006; FAO 2003).

Cauca guans, the largest birds in their area of distribution, are considered among those species most rapidly depleted by hunting (Redford 1992; Renjifo 2002). It serves as a major source of subsistence protein for indigenous people (Brooks and Strahl 2000), although hunting by local residents is illegal (del Hoyo et al. 1994; Muñoz et al. 2006; Renjifo 2002; Rios et al. 2006). Threats to the Cauca guan and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Blue-Billed Curassow (*Crax alberti*)

The blue-billed curassow is a large, mainly black, terrestrial cracid. The species historically occurred in northern Colombia, from the base of the Sierra Nevada de Santa Marta, west to the Sinú valley, through the Río Magdalena (BirdLife International 2007; Cuervo and Salaman 1999; del Hoyo et al. 1994). The species' historic range encompassed an approximate area of 41,197 mi² (106,700 km²) (Cuervo 2002). There were no confirmed observations of blue-billed curassows between 1978 and 1997 (Brooks and Gonzalez-Garcia 2001), and surveys conducted in 1998 failed to locate any males (BirdLife International 2007), prompting researchers to believe the species to be extinct in the wild (del Hoyo et al. 1994). However, a series of observations reported in 1993 were later confirmed (Cuervo 2002).

The current range of the blue-billed curassow is estimated to be 807 mi² (2,090 km²) (BirdLife International 2007) of fragmented, disjunct, and isolated tropical, moist, and humid lowlands and premontane forested foothills in the Rio Magdalena and lower Cauca Valleys of the Sierra Nevada de Santa Marta Mountains, where it feeds on fruit, shoots, invertebrates, and possibly carrion. The species is more commonly found below 1,968 ft (600 m) (del Hoyo et al. 1994), but can be found at elevations up to 3,937 ft (1,200 m) (Collar et al. 1992; Cuervo and Salaman 1999; del Hoyo et al. 1994; Donegan and Huertas 2005; Salaman et al. 2001).

In 1993, sightings were reported in the northern Departments of Córdoba (at La Terretera, near Alto Sinú) and Bolívar (in the Serranía de San Jacinto) (Williams in litt., as cited in BirdLife International 2007). Additional observations were made in the northernmost Department of La Guajira in 2003 (in the Valle de San Salvador Valley) (Strewe and Navarro 2003). More recently, individuals have been observed in the tropical forests of the more central Departments of Antioquía, and Santander and Boyacá Departments, and in the southeastern Department of Cauca (BirdLife International 2007; Cuervo 2002; Donegan and Huertas 2005; Ochoa-Quintero et al. 2005; Urueña et al. 2006). Experts consider the most important refugia for this species to be: (1) Serranía de San Lucas (Antioquía); (2) Paramillo National Park (Antioquía and Córdoba Departments); (3) Bajo Cauca-Nechí Regional Reserve (Antioquía and Córdoba Departments); and (4) Serranía de las Quinchas Bird Reserve (Santander and Boyacá Departments) (BirdLife International 2007; Cuervo 2002).

The blue-billed curassow is categorized as ''Critically Endangered'' by the IUCN Red List (BirdLife International 2007) and is considered a ''Critically Endangered'' species under Colombian law, pursuant to paragraph 23 of Article 5 of the Law 99 of 1993, as outlined in Resolution No. 584 of 2002 (ECOLEX 2007b). The blue-billed curassow is identified as an immediate conservation priority by the Cracid Specialist Group (Brooks and Strahl 2000). There is little information on population numbers for the various reported localities. In 2003, the population at Serranía de las Quinchas (Boyacá Department) was estimated to be between 250 and 1,000 birds. The only other information on the subpopulation level is a report from Strewe and Navarro (2003), based on field studies conducted between 2000 and 2001, that hunting had nearly extirpated the blue-billed curassow from a site in San Salvador. In 1994, the IUCN estimated the blue-billed curassow population at between 1,000 and 2,499 individuals (BirdLife International 2007). In 2001, Brooks and Gonzalez-Garcia (2001) estimated the total population to be much less than 2,000 individuals. In 2002, it was estimated that the species had lost 88 percent of its habitat and half of its population within the species' previous 3 generations (30 years) (Cuervo 2002).

Rapid deforestation and habitat loss throughout the lowland forests across northern Colombia over the past 100 years has extirpated the blue-billed curassow from a large portion of its previous range and continues to impact remaining populations (Brooks and Gonzalez-Garcia 2001; Collar et al. 1992; Cuervo and Salaman 1999). Additionally, oil extraction, gold mining, government defoliation of illegal drug crops, and increased human encroachment put the blue-billed curassow at risk (BirdLife International 2007). Blue-billed curassows are hunted by indigenous people and local residents for sustenance, sport, trade, and entertainment (Brooks 2006; Brooks and Gonzalez-Garcia 2001; Brooks and Strahl 2000; Cuervo and Salaman 1999), involving the species at all life stages, with eggs and chicks collected in some areas for sale at local markets or for domestic use (Brooks 2006; Cuervo 2002). Threats to the blue-billed curassow and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Cantabrian Capercaillie (*Tetrao urogallus cantabricus*)

The Cantabrian capercaillie is a subspecies of the western capercaillie (*T. ugogallus*). Currently it is restricted to the Cantabrian Mountains in northwest Spain. This grouse's range is separated by the Pyrenees Mountains from its nearest neighboring capercaillie subspecies (*T. u. aquitanus*) by a distance of more than 186 mi (300 km) (Quevedo et al. 2006).

The Cantabrian capercaillie occurs in mature beech forests (*Fagus sylvatica*) and mixed beech and oak forests (*Quercus robur, Q. petraea,* and *Q. pyrenaica*) at elevations ranging from 2,625 to 5,900 ft (800 to 1,800 m). The Cantabrian capercaillie also inhabits other microhabitat types such as broom (*Genista spp.*), meadow, and heath (*Erica* spp.) selectively throughout the year (Quevedo et al. 2006). Bilberry (*Vaccinium myrtillus*) is an important component of its diet, and it also feeds on beech buds, catkins of birch (*Betrula alba*), and holly leaves (*Ilex aquifolium*) (Rodriguez and Obeso 2000, as cited in Pollo et al. 2005).

In 2004, at the species level, the western capercaillie (*Tetrao urogallus*) was assessed by the IUCN as a species of ''Least Concern'' (BirdLife International 2004a). However, the IUCN Species Survival Commission's Grouse Specialist Group has noted that the subspecies qualifies to be listed as ''Endangered'' according to the IUCN Red List criteria (Storch 2000). In the year 1998–1999, it was estimated there were 1,900 to 2,000 pairs and that the subspecies was in decline (BirdLife International 2004b). This subspecies is currently classified as ''Vulnerable'' in Spain, which affords it protection from hunting. Although hunting the capercaillie is prohibited in Spain, poaching still occurs. It is unknown what the incidence of poaching is or what impact it has on the subspecies (Storch 2000, 2007).

Habitat degradation, loss, and fragmentation influence the population dynamics of the Cantabrian capercaillie throughout its range (Storch 2000, 2007). This subspecies' historic range has declined by more than 50 percent (Quevedo et al. 2006). The current range is severely fragmented, with 22 percent in low forest habitat, and most of the remaining suitable habitat is in small patches of less than 25 ac (10 ha) (Garcia et al. 2005). Research conducted on other subspecies of capercaillie indicates that the size of forest patches is correlated to the number of males that gather in leks (courtship grounds) to display and that below a certain forest patch size, leks are abandoned (Quevedo et al. 2006).

Patches of good quality habitat are scarce and discontinuous, particularly in the central portions of the species' range (Quevedo et al. 2006), and leks in the smaller forest patches have been abandoned during the last few decades. The leks that remain are now located farther from forest edges than those that were occupied in the 1980s (Quevedo et al. 2006). Recent studies indicate that habitat fragmentation may have a greater effect on this subspecies than previously recognized (Quevedo et al 2005; Vandermeer and Carvajal 2001), and if further habitat fragmentation occurs, the Cantabrian capercaillie population could end up in a few isolated subpopulations too small to ensure the subspecies' long-term survival (Grimm and Storch 2000).

Forest silviculture practices affect both the quantity, as well as the quality, of suitable habitat for the Cantabrian capercaillie. Forest structure plays an important role in determining habitat suitability and occupancy for the subspecies. Quevedo et al. (2006) found that open forest structure with well-distributed bilberry shrubs, an important component of the species' diet (Rodriguez and Obeso 2000, as reported in Pollo et al. 2005), was the preferred habitat type of Cantabrian capercaillie.

Management of forest resources for timber production causes significant changes in forest structure, such as species composition, tree density and height, forest patch size, and understory vegetation (Pollo et al. 2005). Such silviculture practices continue to negatively affect the quality, quantity, and distribution of suitable habitat available for this subspecies, particularly by reducing the availability of bilberry food resources and potentially reducing the availability of suitably sized breeding grounds.

Recurring fires have also been implicated as a factor in the decline of the subspecies (Lloyd 2007). Threats to the Cantabrian capercaillie and its habitat are ongoing, and we find that proposing this subspecies for listing under the Act is warranted.

Gorgeted Wood-Quail (*Odontophorus strophium*)

The gorgeted wood-quail is endemic to the west slope of the East Andes, in the Magdalena Valley (Donegan and Huertas 2005). It is currently known only in the central Colombian Department of Santander, with less than 10 sightings (del Hoyo et al. 1994; Fjelds and Krabbe 1990; Hilty and Brown 1986).

The gorgeted wood-quail prefers montane temperate and humid subtropical forests dominated by roble (*Tabebuia rosea*), and secondary growth forests in proximity to mature forests (Sarria and Álvarez 2002), especially those dominated by oak (*Quercus humboldtii*). The species is most often found at elevations between 5,741 and 6,726 ft (1,750 and 2,050 m) (BirdLife International 2007; Donegan et al. 2003; Donegan and Huertas 2005; Sarria and Álvarez 2002; Turner 2006; Wege and Long 1995). The gorgeted wood-quail is primarily terrestrial (Fuller et al. 2000), living on the forest floor and feeding on fruit, seeds, and arthropods (Collar et al. 1992; del Hoyo et al. 1994; Fuller et al. 2000). It is probably dependent on primary-growth forest for at least part of its life cycle, although it has also been found in degraded habitats and secondary-growth forest (BirdLife International 2007).

The species is classified as ''Critically Endangered'' by the IUCN Red List due to its small and highly fragmented range, with recent population records from only two areas. Logging and hunting are believed to be causing some declines in range and population size (BirdLife International 2004). The population is estimated at between 250 and 999 individuals (BirdLife International 2007).

Since the seventeenth century, the west slope of the East Andes has been extensively logged and converted to agriculture (Stiles et al. 1999). Forest habitat loss below 8,200 ft (2,500 m) has been almost complete (Stattersfield et al. 1998), with habitat reduced in many areas to highly fragmented relict patches on steep slopes and along streams (Stiles et al. 1999). In the early part of the twentieth century, the gorgeted wood-quail was known only in the oak forests in the Department of Cundinamarca. However, extensive deforestation and habitat conversion for agricultural use nearly denuded all the oak forests in Cundinamarca below 8,202 ft (2,500 m) (BirdLife International 2007; Hilty and Brown 1986). Subsequent surveys have not located the species in this area since 1954 (Collar et al. 1992; Fuller et al. 2000; Sarria and Álvarez 2002), and researchers consider the gorgeted wood-quail to be locally extirpated from Cundinamarca (BirdLife International 2007; Fuller et al. 2000; Sarria and Álvarez 2002; Wege and Long 1995). The species has recently been confirmed to exist in three locations, and its current range is between 4 mi^2 (10 km^2) (Sarria and Álvarez 2002) and 10.42 mi^2 (27 km^2) (BirdLife International 2007). These localities are in two disjunct areas within the Department of Santander. Serranoa de los Yarguoes is in northern Santander and the other two localities are adjacent to each other in southern Santander (Donegan and Huertas 2005). The species has lost 92 percent of its former habitat (Sarria and Álvarez 2002), and habitat loss through logging and land conversion to agricultural purposes continues throughout its range (BirdLife International 2007; Collar et al. 1992; Collar et al. 1994; Donegan et al. 2003; Hilty and Brown 1986; Sarria and Álvarez 2002; Stattersfield et al. 1998). Threats to the gorgeted wood-quail and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

Junín Rail (*Laterallus tuerosi*)

The Junín rail is endemic to Lake Junín. The lake is large, covering 35,385 ac (14,320 ha) in the central Andes of Peru at 13,386 ft (4,080 m) above sea level (BirdLife International 2000; Fjeldså 1983). The Junín rail is known from only two sites on the southwest lakeshore, near Ondores and Pari, but it may occur in other portions of the 37,066 ac (15,000 ha) of marshlands surrounding Lake Junín (Fjeldså 1983).

The species' habitat preferences are not fully understood, but it is known to inhabit marshy vegetation located around the margins of Lake Junín. The Junín rail has been observed in the interior of large stands of *Juncus* spp. on the southeast shoreline of the lake and in mosaics of open marshes, in association with *Juncus* spp., mosses, and low herbs (Fjeldså 1983).

Rigorous population estimates for the Junín rail have not been made. In 1983, however, the species was believed to be common based on anecdotal reports of two local fishermen (Fjeldså 1983). Based on these accounts, BirdLife International (2000, 2007) estimated that the population might range between 1,000 and 2,500 individuals. BirdLife International, however, acknowledged that the data quality is poor and that the actual population size might be much smaller (BirdLife International 2000).

The Junín rail is categorized as ''Endangered'' by the IUCN because its range is limited to the shores of a single lake where habitat quality is declining, and the population is very small and believed to be declining (BirdLife International 2007). The Junín rail is considered an ''Endangered'' species by the Peruvian government under Supreme Decree No. 034–2004–AG, which prohibits hunting, taking, transport, or trade of this species, except as permitted by regulation.

One of the key factors contributing to the species' decline is adverse habitat modification. Dam operations cause seasonal lake-level fluctuations of up to 6 ft (2 m) (Martin and McNee 1999). Because few reed-beds are now permanently inundated, tall reeds (*Scirpus tatora*) have virtually disappeared from the lake's shoreline (O'Donnel and Fjeldså 1997). Long-term drawdowns of water levels lead to desiccation of the *Juncus* spp. marshes, and it has been suggested that the Junín rail may be particularly susceptible to such effects because they tend to occupy dry or shallow-water lakeshore sites (Eddleman et al. 1988).

Marsh desiccation also provides easy access to the shore for large livestock herds (primarily sheep, but also cattle, and to a lesser extent llamas and alpacas) to move into the wetlands surrounding the lake, resulting in overgrazing and soil compaction (INRENA 2000, as cited in ParksWatch 2006). Given the large number of livestock that are currently located around the lake (approximately 60,000 to 70,000), habitat destruction and trampling of nests and fledglings negatively impact this species (BirdLife International 2000; BirdLife International 2007; Collar et al. 1992).

Another threat to the Junín rail's habitat is the contamination of Lake Junín from mining wastes. There are a number of mining operations (lead, copper, and zinc) to the north of Lake Junín, and wastewater from these mines runs untreated into the lake via the Rio San Juan (Fjeldså 1981; Martin and McNee 1999). The Rio San Juan (the primary input of water into the Lake) exhibits elevated levels of several trace metals in comparison to local background values (Martin and McNee 1999). In addition, concentrations of fertilizer by-products such as ammonium and nitrate have been found to be elevated (Martin and McNee 1999), and agricultural insecticides, which wash into the lake from the surrounding fields and through drainage systems from villages around the lake, have been detected (ParksWatch 2006). The contaminant load increases substantially during the wet season when agricultural run-off is greater (Martin and McNee 1999).

Cattail (*Typha spp.*) harvesting and burning also destroy the Junín rail's habitat (ParksWatch 2006), resulting in long-term impacts to the species' habitat (Eddleman et al. 1988). Cattails are harvested for handicrafts and livestock forage and are periodically burned to encourage shoot renewal (ParksWatch 2006). Threats to the Junín rail and its habitat continue, and we find that proposing this species under the Act is warranted.

Jerdon's Courser (*Rhinoptilus bitorquatus*)

The Jerdon's courser is endemic to the Eastern Ghats of the states of Andhra Pradesh and extreme southern Madhya Pradesh in India. The species was thought to be extinct for approximately 86 years until 1986, when it was rediscovered in Lankamalai. It has since been located at six additional sites in the vicinity of the Velikonda and Palakonda hills, in the southern State of Andhra Pradesh (Birdlife International 2006). It prefers sparse, thorny areas dominated by *Acacia* spp., *Zizyphus* spp., and *Carissa* spp. (BirdLife International 2006). The Jerdon's courser may also inhabit scrub forest consisting of *Cassia* spp., *Hardwickia* spp., *Dalbergia* spp., *Butea* spp., and *Anogeissus* spp., interspersed with patches of bare ground, in gently undulating rocky foothills (BirdLife International 2006).

This species' population is estimated at 50 to 249 birds (Birdlife International 2006). Very few individuals have been recorded thus far, mainly due to the species' nocturnal and secretive habits (BirdLife International 2006). Negative impacts to the species include exploitation of the scrub-forest, livestock grazing, disturbance by humans and livestock (BirdLife International 2006), and construction of canals (Jegananthen et al. 2005). Jeganathan et al. (2004) found that Jerdon's courser occurrence is strongly correlated with the density of bushes and trees, which is, in turn, negatively affected by mismanaged livestock grazing, woodcutting, and land clearing for agricultural production. The State of Andhra Pradesh has experienced intensive agricultural growth in recent years (Senapathi et al. 2006). From 1991 through 2000, a net loss of 14.6 percent of scrub habitat in the Cuddapah District and parts of the Nellore District in Andhra Pradesh took place, while the amount of land occupied by agricultural fields more than doubled during the same time period (Senapathi et al. 2006). The main cause for the loss of scrub habitat was conversion to agriculture, while gains in scrub habitat came largely at the expense of native deciduous forest due to mechanical clearing and fire (Jeganathan et al. 2004b). Researchers believe that suitable habitat conditions for the Jerdon's courser could be created through the use of a combination of well-managed animal grazing and woodcutting to maintain optimal height, density, and species composition of shrubs for the species. However, over-utilization of scrub habitat could also result in local courser extirpations (Jeganathan et al. 2004a; Senapathi et al. 2006). If not well-managed, increased levels of woodcutting and livestock grazing, as well as mechanical clearing of scrub habitat to create pasture, orchards, and agricultural fields, are all land uses likely to create habitat that is low in quality, highly-fragmented, and unsuitable for use by the Jerdon's courser. From 1991 through 2000, the patch size of scrub habitat declined significantly (Senapathi et al. 2006). Continuing encroachment of human settlement into areas currently occupied by the courser is likely to result in increased livestock grazing pressure and additional land conversion for agricultural purposes.

The Jerdon's courser is categorized as ''Critically Endangered'' on the IUCN Red List because of its small, declining population and habitat that is being reduced by livestock overgrazing and disturbance (BirdLife International 2004). The species is also listed under Schedule I of the Indian Wildlife Protection Act of 1972. Hunting of Schedule I-listed species is strictly prohibited. The Indian Wildlife Protection Act provides for the designation and management of Sanctuaries and National Parks for the purposes of protecting, propagating, or developing wildlife or its environment. Two areas have been established to protect the habitat of the Jerdon's courser. Suitable habitat, however, outside of these Protected Areas continues to be lost through its conversion for development and agriculture. Threats to Jerdon's courser and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

Slender-Billed Curlew (*Numenius tenuirostris*)

The slender-billed curlew migrates along a west-southwest route from Siberia through central and eastern Europe (predominantly Russia, Kazakhstan, Ukraine, Bulgaria, Hungrary, Romania, and Yugoslavia) to southern Europe (Greece, Italy, and Turkey) and North Africa (Algeria, Morocco, and Tunisia). The species has only been confirmed breeding near Tara, Siberia, Russia, between 1909 and 1925, and the only known nests were found on the northern limit of the forest-steppe habitat (Birdlife International 2006). During seasonal migrations and the winter months, the slender-billed curlew utilizes a wide variety of habitats, including coastal marshes, steppe grassland, fish ponds, saltpans, brackish lagoons, tidal mudflats, semi-desert, brackish wetlands, and sandy farmlands in close proximity to lagoons (Hirschfeld 2007).

From the second half of the nineteenth century until 1920, the slender-billed curlew was considered an abundant bird (Chandrinos 2000). Flocks of more than 100 slender-billed curlews were recorded in Morocco as late as 1970. However, population declines have been observed since 1980 (BirdLife International 2006). BirdLife International (2008) reports that in 1994 the population estimate was 50–270 individuals, but the lack of recent confirmed sightings, despite extensive survey efforts, indicates that the population may now include less than 50 birds. Surveys were conducted between 1987 and 2000 in various sections of the species' historic range and covered hundreds of miles (and the corresponding number of kilometers) of habitat. Not a single slender-billed curlew, however, was located during these efforts (CMS 2004; Gretton et al. 2002).

The slender-billed curlew is classified as ''Critically Endangered'' by the IUCN, because the species has an extremely small population size, and the number of birds recorded annually continues to fall, likely representing a continuing population decline (BirdLife International 2004). The species is listed under Appendix I of CITES; commercial trade of this species is strictly prohibited (UNEP–WCMC 2008).

The slender-billed curlew is also listed under Appendices I and II of the Convention on Migratory Species (CMS) (BirdLife International 2004). In an effort to safeguard the slender-billed curlew, a Memorandum of

Understanding (MOU) was developed under CMS auspices and became effective on September 10, 1994. The MOU area covers 30 Range States in Southern and Eastern Europe, Northern Africa and the Middle East. As of December 31, 2000, the MOU had been signed by 18 Range States and three co-operating organizations. An International Action Plan for the Conservation of the slender-billed Curlew has been prepared by BirdLife International (Council of Europe, 1996), and approved by the European Commission and endorsed by the Fifth Meeting of the CMS. Conservation priorities include effective legal protection for the slender-billed curlew and its look-alikes, locating its breeding grounds as well as key wintering and passage sites, applying appropriate protection and management of its habitat, and increasing the awareness of politicians in the affected countries. The CMS website includes an update on the progress being made under the slender-billed curlew MOU. It states that conservation activities have already been undertaken or are underway in Albania, Bulgaria, Greece, Italy, Morocco, Russian Federation, Ukraine and Iran. However, no details of these activities are provided.

The slender-billed curlew is listed on Annex I of the European Union Wild Bird Directive (BirdLife International 2004), which provides a framework for the conservation and management of wild birds in Europe. Although this Directive sets objectives for activities intended to protect wild birds, the legal implementation and achievement of these objectives are at the discretion of each Member State (DEFRA 2008). This species is also listed on Appendix II of the Bern Convention (COE 1979), ''a binding international legal instrument in the field of nature conservation, which covers the whole of the natural heritage of the European continent and extends to some States of Africa'' (COE n.d.). This agreement, however, would not afford protections to the species' breeding habitats in the forest-steppe of Russia.

Historically, hunting levels have been high along the species' entire migratory flyway, especially Russia, and are believed to be the primary factor for the species' previous decline (BirdLife International 2006). Threats to the species on its current breeding grounds are largely unknown due to the lack of information on its nesting localities. However, modification of the forest-steppe habitat within the species' breeding range suggests that the species may be at risk due to loss of its breeding habitat. The forest-steppe has been partially cultivated, and much of the steppe has been developed for intensive agricultural purposes (Gretton 1996).

Progress is underway in some range nations to conserve habitat, prevent hunter misidentification of the species, and increase awareness about the species' precarious status; however, range nations have had differing levels of success in the implementation of needed protections. Threats to the slender-billed curlew and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Marquesan Imperial-Pigeon (*Ducula galeata*)

The Marquesan imperial-pigeon, a very large, broad-winged pigeon, is endemic to Nuku Hiva, the largest of the Marquesas Islands in French Polynesia (BirdLife International 2007). Nuku Hiva is a volcanic island 130 mi^2 (337 km^2) in area; most of the island was originally forested except for the drier north-western plain, where shrub savanna is now predominant. Following conservation recommendations, small numbers of Marquesan imperial-pigeons were translocated beginning in 2000, to the Vaiviki Valley of a second island, Ua Huka, which has been classified as a protected area since 1997. This island contains suitable habitat for this species and is free of mammalian predators (BirdLife International 2007; Blanvillian et al. 2007). The remaining Marquesan imperial-pigeon populations are small, with an estimated 80 to 150 birds on Nuku Hiva (Villard et al. 2003) and 32 birds on Ua Huka (Blanvillian et al. 2007).

The Marquesan imperial-pigeon prefers remote wooded valleys from 820 to 4,265 ft (250 to 1,300 m) in elevation in the west and north of Nuku Hiva. It also inhabits secondary forest and edge habitat near banana and orange plantations (BirdLife International 2007; Blanvillian and Thorsen 2003). The species appears to have strong site-fidelity for its feeding and night roosting sites (Villard et al. 2003).

The Marquesan imperial-pigeon has been categorized as ''Critically Endangered'' by the IUCN since 1994, because it has a very small population size with a decreasing trend and only inhabits one tiny island (aside from the population that is being established at Ua Huka through release efforts). The species appears to owe its survival to the existence of habitat in several areas which are difficult for hunters and introduced species to access (BirdLife International 2007).

The pigeon is protected under the French Environmental Code, which means that the destruction or poaching of eggs or nests or the mutilation, destruction, capture, poaching, intentional disturbance, taxidermy, transport, peddling, use, possession, offer for sale, or purchase of individuals is prohibited by law. Currently, there is no evidence that collection for trade of this species is occurring.

Loss of habitat is believed to have had a large impact on the reduced distribution of the Marquesan imperial-pigeon. Continued grazing by feral goats prevents regeneration of trees, furthering the impacts to previously modified habitat (Thorsen et al. 2002) The introduced black rat (*Rattus rattus*) contributes to habitat degradation on Nuku Hiva by consuming flowers and fruit, thereby inhibiting habitat regeneration (Powlesland et al. 1997).

Transmittal of diseases from domestic pigeons or poultry, or from other introduced avian species imported to Nuku Hiva, has been suggested as a potential risk to this species (Blanvillian et al. 2007). The introduced black rat, although not believed to be a significant predator on adult pigeons (Villard et al. 2003), preys on eggs and young pigeons, potentially putting the species at risk. Rats are also believed to compete for food resources that would otherwise be available to the pigeons (Powlesland et al. 1997). Feral cats have also been introduced on the islands and are suspected to be a predator of adult and juvenile pigeons when they are feeding on low shrubs such as guava (*Psidium guajava*) (Rare Bird Yearbook 2008; Thorsen et al. 2002).

Hunting is believed to be one of the primary contributors to this species' decline and to local extirpations on neighboring islands (Villard et al. 2003). Despite the ban on hunting in French Polynesia since 1967, and the fully protected status of the Marquesan imperial-pigeon species, illegal hunting of the species still occurs. There are no estimates of the current extent of illegal hunting; but long-lived species such as the Marquesan imperial-pigeon with low fecundity rates are generally more affected by the loss of breeding adults than species with shorter life-spans and higher fecundity rates (Clout et al. 1995). Threats to this species and its habitat are ongoing, and we find that proposing the Marquesan imperial-pigeon for listing under the Act is warranted.

Salmon-Crested Cockatoo (*Cacatua moluccensis*)

This cockatoo is endemic to the islands of Ambon, Haruku, Seram, and Saparua in South Maluku, Indonesia. It was formerly a common species of the lowlands within its range (del Hoyo et al. 1997). Although the species was regarded as locally common in 1970, the following decade saw a dramatic decline (Juniper and Parr 1998). Currently, the species is believed to survive in one area on Ambon; however, almost the entire population is restricted to Seram, where, during the 1990s, it suffered declines of 20 to 40 percent in one region. The species is still locally common in Manusela National Park and probably in east Seram. There are no recent records of the species on Haruku and Saparua (BirdLife International 2000).

The salmon-crested cockatoo is largely a resident in lowland rainforest below 3,280 ft (1,000 m) in elevation. The highest densities of cockatoos were encountered in unlogged forest below 590 ft (180 m), illustrating the importance of primary lowland forest (BirdLife International 2007). In a study of the density and distribution of the salmon-crested cockatoo, Kinnaird et al. (2003) confirmed that the highest densities of cockatoos occurred in primary forest sites with good forest structure and found that the lowest density was a logged site with low stature forest. Marsden (1998) found that density estimates of salmon-crested cockatoos in unlogged forest below 984 ft (300 m) were more than double those in logged forests. Habitat rich in strangler fig trees (*Ficus* spp.) and *Octomeles sumatranus*, the tree species the cockatoos prefer for nesting, was also likely to produce the highest densities of cockatoos (Kinnaird et al. 2003). The diet of salmon-crested cockatoos consists of seeds, nuts, young coconuts (*Cocos nucifera*) (the birds chew through the outer layers of green coconuts to get at the soft pulp), berries, and insects and their larvae (Forshaw 1989; Juniper and Parr 1998).

The species is listed as ''Vulnerable'' on the IUCN Red List because it has suffered a rapid population decline as a result of trapping for the pet bird trade and because of deforestation in its small range (BirdLife International 2004). Current populations are estimated at 62,400 individuals, with a decreasing population trend; the decline for the past 10 years or 3 generations is estimated at 30 to 49 percent (BirdLife International 2007b).

By the 1980s, salmon-crested cockatoo populations were declining rapidly due to uncontrolled trapping for the pet bird trade (BirdLife International 2007a). Concerns about unrestricted trade of parrots, including the salmon-crested cockatoo, led to a CITES Appendix-II listing of all Psittaciformes spp. in 1981 (CITES 2008). After the CITES listing, some 74,509 individual salmon-crested cockatoos were exported from Indonesia from 1981 to 1990 (BirdLife International 2000). The level of imports from Indonesia from 1983 to 1987, as reported to CITES, averaged 8,500 to 9,500 birds per year (CITES 1989b); trade reported in 1985 and 1987 exceeded the quota set by Indonesia by over 1,300 and 3,661 birds, respectively (CITES 1989a). In October 1989, the salmon-crested cockatoo was transferred to CITES Appendix I, which precludes commercial international trade. However, trappers reportedly remained active, and wild-caught birds were being openly sold in the domestic market (Metz and Nursahid 2004). Interviews in villages suggest that perhaps as many as 4,000 birds are still being captured each year (BirdLife International 2001).

Currently, logging impedes salmon-crested cockatoo conservation. Nearly 50 percent of Seram is held within logging concessions, with more than 75 percent held within lowland habitat, prime salmon-crested cockatoo habitat. Only 14 percent of the forests are in protected areas, and logging concessions overlap more than 30 percent of these protected areas, with conflicts over the boundaries of parks and logging concessions. Small-scale illegal logging also occurs within these protected areas. Unsustainable logging practices, which destroy the forest canopy, dramatically reduce habitat available for cockatoos, especially if large nest trees are harvested (Kinnaird et al. 2003).

In addition, the salmon-crested cockatoo's habitat is being degraded and threatened by agriculture, human settlement, and hydroelectric power projects (BirdLife International 2007a). The species has been considered a pest to coconut palms, and consequently has been persecuted, at least historically (BirdLife International 2000).

In 2000, a program was launched to promote ecotourism which was linked to a local project to raise awareness about the plight of the salmon-crested cockatoo. Current conservation measures suggest continuing and expanding the awareness program and using the salmon-crested cockatoo as the island's flagship species to reduce trapping pressure and encourage local support for the survival of the species (BirdLife International 2007a). At the present time, however, the threats to the salmon-crested cockatoo and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

Southeastern Rufous-Vented Ground Cuckoo (*Neomorphus geoffroyi dulcis*)

The southeastern rufous-vented ground-cuckoo is one of seven subspecies of the rufous-vented ground-cuckoo (*Neomorphus geoffroyi*). The species as a whole ranges from Nicaragua to central South America, occurring at several disjunct localities (del Hoyo et al. 1997; Howard and Moore 1980; Payne 2005; Sibley and Monroe 1990). There is currently little concern for the conservation status of the whole species, but the *N. g. dulcis* subspecies, the southeastern rufous-vented ground cuckoo, has experienced serious declines (BirdLife International 2007). Historically, the southeastern rufous-vented ground-cuckoo subspecies had a widespread distribution in southeastern Brazil from Espirito Santo to Rio de Janeiro (del Hoyo et al. 1997), where it has likely always been locally rare (IUCN 1981). This subspecies may now, however, be extinct throughout its entire range; the last confirmed sighting was in 1977 in the Sooretama Biological Reserve north of the Doce River in Esprito Santo (Payne 2005; Scott and Brooke 1985). A recent photographic record (ca. 2004) of a single bird indicates that the subspecies may still occur at Doce River State Park in Minas Gerais (Scoss et al. 2006), but there are no population figures beyond this information.

The southeastern rufous-vented ground cuckoo inhabits tropical lowland evergreen forests, where it feeds on large insects, scorpions, centipedes, spiders, small frogs, lizards, and occasionally seeds and fruit (del Hoyo et al. 1997). It is a solitary subspecies that is dependent upon large blocks of undisturbed tropical lowland forest within the Atlantic Forest biome (del Hoyo et al. 1997; IUCN 1981; Payne 2005; Sick 1993). These birds can run and can flutter to an elevated perch to lookout and to roost, but they are not capable of sustained flight (Payne 2005). Therefore, major rivers and other extensive areas of non-habitat are thought to impede their movements.

Since 1981, the southeastern rufous-vented ground-cuckoo, has been categorized as ''Endangered'' on the IUCN Red List (IUCN 1981). It is formally recognized as ''Endangered'' in Brazil, and is directly protected by legislation promulgated by the Brazilian government (ECOLEX 2007; IUCN 1981). These protections prohibit the following activities with regard to this species: export and international trade, collection, research, and captive propagation. They also provide measures which help to protect remaining suitable habitat, such as prohibition of exploitation of the remaining primary forests within the Atlantic forest biome and management of various practices in primary and secondary forests, such as logging, charcoal production, reforestation, recreation, and water resources (ECOLEX 2007). The existing regulatory mechanisms that apply to the southeastern rufous-vented ground-cuckoo would appear to be largely adequate if fully enforced; however, there is currently a lack of enforcement of them (BirdLife International 2003a; Conservation International 2007c; Costa 2007; Neotropical News 1997b; Peixoto and Silva 2007; Scott and Brooke 1985; The Nature Conservancy 2007; Venturini et al. 2005). As a result, significant threats to the subspecies' remaining habitats are ongoing.

Based on a number of recent estimates, 92 to 95 percent of the area historically covered by tropical forests within the Atlantic Forest biome has been converted or severely degraded as a result of various human activities (Höfling 2007; The Nature Conservancy 2007). In addition to the overall loss and degradation of native habitat within this biome, the remaining tracts of habitat are severely fragmented. Most of the tropical forest habitats believed to have been used historically by the southeastern rufous-vented ground-cuckoo have been converted or severely degraded by human activities (del Hoyo et al. 1997; IUCN 1981; Payne 2005; Scott and Brooke 1985; Sick 1993). Terrestrial insectivorous birds, such as the southeastern rufous-vented ground-cuckoo, are especially vulnerable to habitat modifications which increase the variability of insect food supplies (Goerck 1997), and the subspecies cannot occupy these extensively altered habitats. The subspecies is dependent upon large blocks of undisturbed forest habitat for its life-cycle requirements, and habitat destruction within the ground-cuckoo's range results in a patchy landscape, reducing the availability of the type of forest habitat necessary for the subspecies. Threats to the southeastern rufous-vented ground cuckoo and its habitat continue, and we find that proposing this subspecies for listing under the Act is warranted.

Margaretta's Hermit (*Phaethornis malaris margarettae*, previously known as *Phaethornis margarettae*)

Margaretta's hermit was first described as a new species in 1972 by A. Ruschi (Sibley and Monroe 1990). Current taxonomic studies place Margaretta's hermit as a subspecies of the great-billed hermit (*Phaethornis malaris*) (Sick 1993).

Margaretta's hermit is found in coastal east Brazil and inhabits the understory of inundated lowland forest, secondary growth, bamboo thickets, and shrubbery. This subspecies is currently limited to forest remnants; consequently, further habitat destruction could be detrimental to this subspecies (del Hoyo et al. 1999). The Margaretta's hermit is listed in Appendix II of CITES (CITES 2006).

The last confirmed occurrence of the Margaretta's hermit is from a relatively old (ca. 1978) sighting of the subspecies on a privately-owned remnant forest called Klabin Farm, which at the time was approximately 15.4 mi^2 (40 km^2) in Espiritu Santo, and the subspecies likely occurred at the Sooretama Biological Reserve in Espiritu Santo until around 1977 (IUCN 1981).

Most of the tropical forest habitats believed to have been used historically by the Margaretta's hermit have been converted or are severely degraded due to human activities related to land clearing and urban and agricultural development in coastal east Brazil, and the subspecies cannot occupy these extensively altered areas (del Hoyo et al. 1999; Höfling 2007; IUCN 1981; Sick 1993; The Nature Conservancy 2007). While the Margaretta's hermit is not strictly tied to primary forest habitats and can make use of secondary-growth forests, this does not lessen the risk to the subspecies from the effects of deforestation and habitat degradation. This is because Atlantic Forest birds that are tolerant of secondary-growth forests, yet that are also rare or have restricted ranges (i.e., less than 21,000 square km (8,100 square mi)), are threatened by these impacts equally as primary forest-obligate species (Harris and Pimm 2004). The last site known to be occupied by the Margaretta's hermit totaled only about 40 square km (15 square mi) (IUCN 1981). The susceptibility of rare, limited-range species that are tolerant of secondary-growth forests occurs for a variety of reasons. For example, many hummingbird species are susceptible to excessive sun and readily abandon their nests at altered forested sites with too much exposure (Sick 1993), as can occur with various human activities that result in partial clearing (*e.g.*, selective logging). In addition, management of plantations often involves intensive control of the site's understory cover (Rolim and Chiarello 2004; Saatchi et al. 2001). Even if the forest canopy structure remains largely intact, such management practices eventually result in loss of native understory plant species and severely alter understory structure and dynamics, which can be especially detrimental to pollinator species such as the Margaretta's hermit. Furthermore, even when forested lands are formally protected, the remaining fragments of habitat where the subspecies may still occur will likely continue to undergo degradation due to their altered dynamics and isolation (Tabanez and Viana 2000). Finally, secondary impacts that are associated with the above activities include severe fragmentation of the remaining tracts of forested habitat potentially used by the subspecies, and the potential introduction of disease vectors or exotic predators within the subspecies' historic range. As a result of the above influences, there is often a time lag between the initial conversion or degradation of suitable habitats and the extinction of endemic bird populations (Brooks et al. 1999a; Brooks et al. 1999b). Therefore, even without further habitat loss or degradation, the Margaretta's hermit remains at risk from past impacts to its suitable forested habitats.

Loss of this species' habitat is likely to continue due to the high pressure for coastal development. Threats to the Margaretta's hermit and its habitat are ongoing, and we find that proposing this subspecies for listing under the Act is warranted.

Black-Breasted Puffleg (*Eriocnemis nigrivestis*)

The black-breasted puffleg, endemic to Ecuador, is a member of the hummingbird family (Trochilidae). It is confined to the northern ridge crests of Volcán Pichincha near Quito, Ecuador (Fjeldså and Krabbe 1990; Ridgely and Greenfield 1986a; Ridgely and Greenfield 1986b). Volcán Pichincha reaches peaks at 15,699 ft (4,785 m) (Phillips 1998). The species has not been confirmed in the only other known sighting locality, the Volcán Atacazo, since 1902 (Collar et al. 1992; BirdLife International 2007).

This species prefers temperate elfin forests (comprised primarily of *Polyepsis* spp. trees) between 9,350 and 11,483 ft (2,850 and 3,500 m) (Fjeldså and Krabbe 1990; Ridgely and Greenfield 1986a; Ridgely and Greenfield 1986b). It is an altitudinal migrant, spending the breeding season (November to February) in the humid elfin forest and the rest of the year at lower elevations, as determined by flowering of certain plants (Bleiweiss and Olalla 1983; Collar et al. 1992; del Hoyo et al. 1999).

Habitat loss, specifically the felling of *Polylepis* spp. wood for conversion to charcoal, was the primary cause of historical black-breasted puffleg declines (Phillips 1998). Following more than 13 years without any observation of the species, the black-breasted puffleg was rediscovered on Volcán Pichincha in 1993 (Phillips 1998). The number of specimens in museum collections taken in the nineteenth century up until 1950 is over 100, suggesting the species was once more common (Collar et al. 1992).

The black-breasted puffleg is classified as ''Critically Endangered'' on the IUCN Red List because it has an extremely small range, and the population is restricted to one location (BirdLife International 2007). Its single population is estimated at 50 to 250 adult individuals, with a declining trend (BirdLife International 2007; del Hoyo et al. 1999). The population is believed to have declined by 50 to 79 percent in the past 10 years, or 3 generations, with more than 20 percent of this loss having occurred within the past 5 years. This rate of decline is predicted to continue (BirdLife International 2007). The species is also classified as ''Critically Endangered'' under Ecuadorian law (ECOLEX 2007).

Within the current range of the black-breasted puffleg (33 mi^2 (88 km^2)), approximately 93 percent of its habitat has been lost (BirdLife International 2007; Hirchfeld 2007). The ridge-crests within the range of the black-breasted puffleg are relatively level, and local settlers have cleared the majority of forested habitat within the species' range and converted it to potato cultivation and grazing (Bleiweiss and Olalla 1983; del Hoyo 1999). Some ridges are almost completely devoid of natural vegetation, and even if black-breasted pufflegs still occur in these areas, their numbers are most likely quite low (BirdLife International 2007).

In 2001, the area around the Volcáns Pichincha and Atacazo was established as the Yanacocha Reserve, and charcoal production within the reserve, which was considered the primary cause for the species' historical decline, was restricted (Bird Conservation 2005; Phillips 1998). The Yanacocha Reserve totals approximately 3,100 ac (1,250 ha) and contains approximately 2,372 ac (960 ha) of *Polylepis* forest (Hirchfeld 2007; World Land Trust 2007).

In 2001, the Ecuadorian government agreed to construct a pipeline to transport heavy oil from the Amazon basin to Esmaraldas on the Pacific Coast (Mindo Working Group 2001). The environmental impact study revealed that the proposed route went through black-breasted puffleg habitat (Mindo Working Group 2001). Satellite mapping showed that much of the area in puffleg habitat was already destroyed, with little remaining habitat above 9,186 ft (2,800 m). The black-breasted puffleg had previously been found at 10,171 ft (3,100 m) in an upper extension from the likely unsuitable forested zone lower down. The pipeline was proposed to pass through pasture slightly above this patch, risking further habitat destruction with the building of a road (Mindo Working Group 2001). The pipeline was recently constructed, transecting every major ecosystem on the Volcán Pichinche, including black-breasted puffleg habitat. The pipeline also deforested pristine habitat, making these areas more accessible and opening them up to further human infiltration (BirdLife International 2007). Threats to the black-breasted puffleg and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Chilean Woodstar (*Eulidia yarrellii*)

The Chilean woodstar is endemic to several river valleys from Tacna, Peru, to northern Antofagasta, Chile, close to the Pacific Coast. This area lies at the northern edge of the Atacama Desert, one of the driest places on Earth (Collar et al. 1992). Breeding populations are only known to occur in the Vitor and Azapa Valleys in extreme northern Chile (BirdLife International 2000; Estades et al. 2007). In the past, there were a few observations of the species in Tacna, Peru, close to the border of Chile, but the observations were infrequent, and there have been no confirmed observations in the last 2 decades (Collar et al. 1992; Fjeldså and Krabbe 1990).

The Chilean woodstar was described as a species of extremely limited range and very small total population size over 40 years ago (Johnson 1967). In September 2003, while using fixed-radius point counts to sample an area larger than the species' presumed range, Estades et al. (2007) found that the Chilean woodstar was restricted to the Azapa and Vitor Valleys of northern Chile, and that it was the rarest hummingbird in the Azapa Valley (Estades et al. 2007). Despite repeated searches, the species was not located in the Lluta Valley, where a breeding colony had been previously reported (Fjeldså and Krabbe 1990). The population was estimated to be about 1,539 individuals. In April 2004, the population was estimated at 758 individuals. The authors warned against interpreting their results as a population crash from 2003 to 2004, because the surveys in 2004 were conducted in April when food resources and woodstar populations are generally more widely dispersed than they are in September (Estades et al. 2007).

The Chilean woodstar inhabits riparian thickets, secondary growth, desert river valleys, arid scrub, agricultural lands, and gardens (Stattersfield et al. 1998). It relies on nectar-producing flowers for food, but also relies on insects for a source of protein (del Hoyo et al. 1999; Estades et al. 2007). The Chilean woodstar drinks nectar from the flowers of a variety of native and ornamental plants, as well as crops—including alfalfa, garlic, onion, and tomatoes (Estades et al. 2007).

The IUCN Red List categorizes the Chilean woodstar as ''Endangered'' because it inhabits a very small range, with all viable populations apparently confined to remnant patches in two desert river valleys. These valleys are heavily cultivated, and the extent, area, and quality of suitable habitat are likely declining (BirdLife International 2007). The Chilean woodstar is listed as an ''Endangered and Rare'' species in Chile and was also designated as a ''National Monument'' under Diario Oficial No. 38.501, which prohibits all hunting and capture of the species. These regulations do not, however, address the current and ongoing destruction and degradation of this species' habitat. The Chilean woodstar is listed in Appendix II of CITES (UNEP–WCMC 2008).

The historic range of the Chilean woodstar has been severely altered by extensive planting of olive and citrus groves in the valleys of northern Chile and southern Peru. The indigenous food plants of the species may have been seriously reduced when habitat for the species was converted to agriculture, but the woodstar apparently adapted to survive on introduced garden flowers (del Hoyo et al. 1999; Estades et al. 2007). However, loss of some native plant species may be a limiting factor for the survival of the species. Estades et al. (2007) reported that one of the reasons the Chilean woodstar disappeared from the Lluta Valley is likely due to the destruction of almost all of the chañares (*Geoffrea dicorticans*), which is considered one of the most important food resources for the species, but is unpopular with farmers who consider it undesirable and an attractant to mice. In addition, the use of insecticides to control the Mediterranean fruit fly (*Ceratitis capitata*) in the 1960s and early 1970s correlates with declines in Chilean woodstar abundance (Estades et al. 2007). The use of such pesticides has been reduced since the 1970s; however, Estades et al. (2007) reported that other insecticides that may harm the woodstar are still being used for some applications.

Chilean woodstars appear to rely primarily on introduced olive trees for nesting. Although olive trees are not exposed to as many pesticides as other fruit trees in the region, the use of high-pressure water spraying to control mold threatens nests, eggs, and chicks (Estades et al. 2007).

Future land-cover projections from the Millennium Ecosystem Assessment indicate that by 2050, 18 to 24 percent of the Chilean woodstar's range is likely to be unsuitable for the species (Jetz et al. 2007).

Estades et al. (2007) hypothesized that rapid population increases of the Peruvian sheartail hummingbird (*Thaumastura cora*), which shares the range of the Chilean woodstar, is a strong competitor for food or space (Estades et al. 2007). The sheartail is more aggressive than the Chilean woodstar; therefore, it is believed to displace the woodstar within its range. In Azapa, Peruvian sheartails occupy the lower parts of the valley where there is an ample supply of flowers in residential areas year-round. Chilean woodstars, on the other hand, are generally located in mid-valley agricultural areas, where there is a much higher risk of pesticide exposure. Threats to the Chilean woodstar and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

Esmeraldas Woodstar (*Chaetocercus berlepschi*, previously known as *Acestrura berlepschi*)

The Esmeraldas woodstar was first taxonomically described by Simon in 1889, who placed the species in the Trochilidae family, under the name *Chaetocercus berlepschi* (BirdLife International 2007). The species is also known by the synonym *Acestrura berlepschi*. CITES, BirdLife International (BirdLife International 2007), and the Integrated Taxonomic Information System (ITIS 2008) recognize the species as *Chaetocercus berlepschi*. We accept the species as *Chaetocercus berlepschi*, and change our reference to this species from our 2007 Notice of Review.

The Esmeraldas woodstar is restricted to a small area on the Pacific slope of the Andes of western Ecuador (Esmeraldas, Manabi, and Guayas), where only very rare and localized populations are found (BirdLife International 2007).

It ranges along the slopes of the coastal cordillera up to 1,640 ft (500 m) (del Hoyo et al. 1999; Ridgely and Greenfield 1986b; Williams and Tobias 1991). The current extent of the species' range is approximately 446 mi^2 (1,155 km^2) in 3 disjunct and isolated areas (BirdLife International 2007; Dodson and Gentry 1991).

The Esmeraldas woodstar generally prefers lowland, moist forest habitat (del Hoyo et al. 1999). It has also been recorded in the canopy of semi-humid secondary growth at 164 to 492 ft (50 to 150 m) in December through March, when it is believed to breed (Becker et al. 2000). The species has not been recorded in this habitat type at other times of year, and there is no evidence concerning its long-term ability to survive in this type of forest habitat (BirdLife International 2007).

The Esmeraldas woodstar is considered a rare, range-restricted species with highly localized populations in three general areas (BirdLife International 2007; del Hoyo et al. 1999). There have been no population surveys of this species. BirdLife International estimated that the population includes between 186 and 373 individuals, based on density estimates using similar species of hummingbirds (BirdLife International 2007).

This species is classified as ''Endangered'' by the IUCN Red List on the basis of occupying a small and severely fragmented range with ongoing and very rapid declines in range and, presumably, population (BirdLife International 2007). The species is listed in Appendix II of CITES (UNEP–WCMC 2008b). It is identified as an ''Endangered'' species under Ecuadorian law (ECOLEX 2007f). As such, hunting for sport or commercial purposes is prohibited (ECOLEX 2007g; ECOLEX 2007h). However, we do not consider hunting to be a risk to the Esmeraldas woodstar, so this law does not reduce any threats to the species.

The Esmeraldas woodstar inhabits one of the most threatened forest habitats within the Neotropics (del Hoyo et al. 1999). All forest types within the species' range have diminished rapidly due to logging and clearing for agriculture (Dodson and Gentry 1991). The woodstar inhabits a very small and severely fragmented range, which is decreasing rapidly in size. Ongoing declines in the bird's population are linked to persistent habitat destruction which destroys nesting, breeding, and feeding habitat (BirdLife International 2007). Persistent grazing by goats and cattle damages the understory and prevents regeneration of the forest that the woodstar utilizes (Dodson and Gentry 1991). Dodson and Gentry (1991) indicated that rapid habitat loss is continuing, at least in unprotected areas, and extant forests will soon be eliminated. In Manabi Province, the Esmeraldas woodstar may occur in Machalilla National Park (Collar et al. 1992), but it does not receive adequate protection because its habitat is threatened by illegal settlement, deforestation, livestock-grazing, and habitat clearance by people with land rights (BirdLife International 2007). Threats to the Esmeraldas woodstar and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Royal Cinclodes (*Cinclodes aricomae*)

The royal cinclodes occurs in the Andes of southeastern Peru (Cuzco, Apurimac, and Puno) and adjacent Bolivia (La Paz) (BirdLife International 2007). The species appears to be restricted to mature, humid *Polylepis* spp. woodlands that can sustain mossy ground-cover (Collar et al. 1992). Its diet consists primarily of invertebrates, small vertebrates (small frogs), and occasionally seeds (del Hoyo et al. 2003). It seeks food by probing through moss and debris on the forest floor (Collar et al. 1992; Fjeldså 2002b; del Hoyo et al. 2003), and likely requires territories as large as 5 to 7 ac (2 to 3 ha) due to its feeding strategy (Engblom et al. 2002).

The total royal cinclodes population was estimated to range between 100 and 150 individuals in 1990 (Fjeldså and Krabbe 1990). BirdLife International (2007) estimates the population size to be between 50 and 249 individuals. Detailed surveys of suitable habitat in Peru revealed only 189 individuals that were restricted to 1,554 ac (629 ha) (Chutas 2007). In Bolivia, the population is estimated at 30 individuals that are located on 1,236 ac (500 ha) of fragmented habitat (Purcell and Brelsford 2004). However, the royal cinclodes does not always respond to the tape-playback method that was used to census the population; therefore, the population estimate may not be indicative of the actual population size (Gomez in litt. 2007).

The IUCN Red List categorizes the royal cinclodes as ''Critically Endangered'' due to its extremely small population, which consists of tiny subpopulations that are severely fragmented and dependent upon a rapidly declining habitat (BirdLife International 2007). The royal cinclodes is completely dependent upon high-elevation humid *Polylepis* forests for its survival, and the ongoing loss of this habitat poses the greatest risk to this species. Based on comprehensive surveys and analyses of maps and satellite images, Fjeldså and Kessler

(1996, as cited in Fjeldså 2002a) estimated that *Polylepis* forests now cover less than 247,105 ac (100,000 ha) in Peru and 1,235,527 ac (500,000 ha) in Bolivia, and the majority of the forest is very dispersed with extensive bushy growth. Less than 1 percent of the *Polylepis* forest remains in the humid highlands, where *Polylepis* forests are able to grow tall and dense (Fjeldså 2002a). The royal cinclodes is particularly sensitive to reduced forest density, because decreased canopy cover permits desiccation of the mosses growing within humid *Polylepis* forests, which reduces foraging microhabitats for the species (Engblom et al. 2002).

Fire and livestock grazing are the important factors affecting the distribution of *Polylepis* forests. The vegetation is restricted to stream ravines, loose rocks, rock ledges, and sandy ridges—all places where fires cannot spread and livestock does not normally roam (Fjeldså 2002a; Fjeldså 2002b). Burning land between patches of *Polylepis* forests to stimulate the growth of grasses (chaqueo) for grazing prevents regeneration of native forests and is considered the key factor limiting the distribution of *Polylepis* forests (Fjeldså 2002b). Trampling and grazing by sheep and cattle further limit forest regeneration (Fjeldså 2002a) and can contribute to the degradation of remaining forest patches. Sheep and cattle have solid, sharp hooves that churn up the earth, damaging vegetation and triggering erosion (Purcell et al. 2004). The loss of nutrient-rich soils can also cause degradation and ultimate destruction of *Polylepis* forests (Fjeldså 2002b; Purcell et al. 2004).

As human populations increase in the high-Andes of Bolivia, many farmers burn patches of *Polylepis* forests to make agricultural fields for crops. The scarcity of arable land has even caused some farmers to burn *Polylepis* on steep hillsides that would not normally be considered suitable for cultivation (Hensen 2002). These farming practices continue to result in the rapid loss of *Polylepis* forests and amplified soil erosion. Firewood harvest is another significant threat to remaining patches of *Polylepis* forests. Road building and mining projects for the expanding human population around Bolivia's largest city, La Paz, have increased accessibility to remaining *Polylepis* forest fragments, further threatening the continued existence of the forests upon which the royal cinclodes depends (Purcell et al. 2004; Purcell and Brelsford 2004). Threats to the royal cinclodes and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

White-Browed Tit-Spinetail (*Leptasthenura xenothorax*)

The white-browed tit-spinetail is restricted to high-elevation—12,139 to 14,928 ft (3,700 to 4,550 m) above sea level—semi-humid *Polylepis* and *Polylepis-Gynoxys* woodlands (Collar et al. 1992). This species forages in pairs or small family groups, often in mixed species flocks, gleaning insects from bark crevices and moss and lichens on twigs, branches, and trunks (BirdLife International 2007; Engblom et al. 2002; Parker and O'Neill 1980).

Historically, the white-browed tit-spinetail may have occupied the once large and contiguous expanses of *Polylepis* forests of the high-Andes of Peru and Bolivia (Fjeldså 2002a), but it is now limited to remnant *Polylepis* forests in the Andes mountains of southeast Peru around Cuzco (Birdlife International 2007; Fjeldså and Krabbe 1990; InfoNatura 2007).

Fjeldså and Krabbe (1990) described the white-browed tit-spinetail as common in suitable habitat and numbering ''probably some hundreds,'' yet quite vulnerable to loss of its already restricted habitat. Other estimates of the species' total population size range from 250 to 1,000 (Fjeldså 2002b) to 500 to 1,500 (BirdLife International 2007; Engblom et al. 2002). Recently, only 305 individuals were reported, based on detailed surveys of suitable *Polylepis* forest habitat (Chutas 2007).

The IUCN categorizes the white-browed tit-spinetail as ''Endangered'' due to its very small and severely fragmented range and population, which continue to decline with habitat loss and lack of habitat regeneration (BirdLife International 2007). The white-browed tit-spinetail is listed as an ''Endangered'' species by the Peruvian government under Supreme Decree No. 034–2004–AG, which prohibits hunting, taking, transport, or trade of this species, except as permitted by regulation. However, the species' habitat is not protected by this law.

The principal factor affecting the distribution of *Polylepis* forests, the species' habitat, is the intensity of burning and grazing, which restricts vegetation growth to locations where fires cannot spread and cattle and sheep do not normally roam, such as ravines, boulders, rock ledges, and sandy ridges (Fjeldså 2002a and b). Many farmers, however, destroy *Polylepis* spp. by planting crops on steep hillsides unsuitable for cultivation (Hensen 2002). Harvesting of firewood from *Polylepis* forests is also a significant threat to the white-browed tit-spinetail's habitat (Aucca and Ramsay 2005; Engblom in litt. 2000). Trampling and grazing by sheep and cattle limit forest regeneration and can contribute to degradation of remaining forest patches (Fjeldså 2002a; Purcell et al. 2004). Remaining forest fragments are becoming more accessible to the expanding population around Bolivia's largest city through road building and mining projects, further threatening the survival of *Polylepis* forests upon which the white-browed tit-spinetail depends (Purcell et al. 2004).

Ongoing loss of the *Polylepis* habitat is considered the primary threat to this species' continued existence. Based on comprehensive surveys and analyses of maps and satellite images, Fjeldså and Kessler (1996, as cited in Fjeldså 2002a) estimated that *Polylepis* forests now cover less than 247,105 ac (100,000 ha) in Peru. In Bolivia, 1,235,527 ac (500,000 ha) of *Polylepis* forest remain, but most of it is very dispersed and bushy. However, less than 1 percent persists in the humid highland habitat for the white-browed tit-spinetail, where *Polylepis* forests can grow to be tall and dense (Fjeldså 2002a). According to Chutas (2007), the species is now confined to about 1,532 ac (620 ha) of habitat. From 1956 to 2005, the rate of forest patch habitat decline to the north of Cuzco, Peru, was only about 1 percent; however, the remaining habitat patches in this area are very small (mean patch size of 6.2 ac (2.5 ha)). During this same time-period, 10 percent of existing forest patches showed a decline in density, indicating that degradation might be a more serious threat than outright destruction in this area (Jameson and Ramsay 2007). Threats to the white-browed tit-spinetail and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Black-Hooded Antwren (*Formicivora erythronotos*, previously known as *Myrmotherula erythronotos*)

The black-hooded antwren inhabits early successional secondary growth habitats and the understory of remnant old-growth secondary forests in coastal southeastern Brazil (BirdLife International 2007; Harris and Pimm 2004). This antwren species was previously known only from 20 skins that were collected during the nineteenth century (E. Mendonça and L.P. Gonzaga in litt. 2000, as cited in BirdLife International 2007; Buzzetti 1998), and was believed to be extinct until it was rediscovered in 1987 (Harris and Pimm 2004). There have been recent reports that the species has been seen with increased frequency at a coastal reserve near Rio de Janeiro, the

Reserva Ecológica de Jacarepiá (Worldtwitch 2007).

The IUCN Red List classifies the species as ''Endangered,'' because it has a very small and highly fragmented range. The black-hooded antwren appears to be declining rapidly in response to continuing habitat loss. Currently, it is known to inhabit 7 sites, and the population is estimated at 1,000 to 2,499 birds with a decreasing population trend (BirdLife International 2007). The IUCN Red List notes, however, that data quality is poor for these estimates and that there is a serious need for new population demographic information on the species' current population size (BirdLife International 2007). This species is also formally recognized as ''Endangered'' under Brazilian law (Order No. 1.522) (ECOLEX 2007).

The black-hooded antwren resides in one of the most densely populated regions of Brazil, where deforestation has been occurring for more than 400 years (BirdLife International 2003). The species' habitat is currently threatened by ongoing urbanization, industrialization, and agricultural expansion. The antwren's habitat has been reduced to less than 10 percent of its original extent (Brown and Brown 1992, as cited in BirdLife International 2003; Höfling 2007; The Nature Conservancy 2007). Remaining tracts of suitable habitat near Rio de Janeiro and Sao Paulo are threatened by ongoing development of coastal areas, primarily for tourism enterprises (e.g., hotel complexes, beachside housing) and associated infrastructure, as well as widespread clearing for expansion of livestock pastures and plantations (Birdlife International 2007). Threats to the black-hooded antwen and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Fringe-Backed Fire-Eye (*Pyriglena atra*)

The fringe-backed fire-eye is known from the narrow coastal belt of Atlantic forest in the vicinity of Salvador, coastal Bahia (west of the town of Santo Amaro), forest patches along the Linha Verde highway, and north to southern Sergipe (in the vicinity of Crasto and Santa Luzia de Itanhia), Brazil (Pacheco and Whitney 1995, J. Minns in litt. 1998, B.M. Whitney in litt. 1999, and J. Mazar Barnett in litt. 2000; all as cited in BirdLife International 2007; Collar et al. 1992; del Hoyo et al. 2003). Recent fieldwork indicates that the species' distribution is not as disjunct as previously considered because it has been found in remnant forest and secondary-growth patches along the northern coast of Bahia at Conde and Jandaíra (Souza 2002, as cited in BirdLife International 2007). Although populations may have been vastly reduced over time, the species' preference for early successional secondary-growth habitat means its range is likely to have been underestimated (BirdLife International 2007). The fringe-backed fire-eye also favors the tangled, dense undergrowth of lowland forests as well as other semi-open habitats where horizontal perches are located close to the ground (BirdLife International 2007).

Currently, the population is estimated at 1,000 to 2,499 individuals (BirdLife International 2007), an increase from the population estimate in 2000, which indicated that between 250 and 999 individuals remained in the wild (BirdLife International 2000). The increase in the population estimate results from extension of the species' known range (del Hoyo et al. 2003), as well as indications that the distribution was not as disjunct as previously thought (Souza 2002, as cited in BirdLife International 2007). From 2000 to 2004, the fringe-backed fire-eye was categorized as ''Critically Endangered'' by the IUCN Red List, because of its extremely small range and declining habitat and because it was known from a few, highly-fragmented localities (IUCN 2002). While the fringe-backed fire-eye is now classified as ''Endangered'' by the IUCN Red List because the species' range is more extensive than previously known (BirdLife International 2007), it does still have a very small, fragmented range, within which the extent and quality of its habitat are continuing to decline and where it is only known from a few localities (BirdLife International 2007). The entire range of the fringe-backed fire-eye encompasses only about 1,924 mi^2 (4,990 km^2), with only 20 percent of this area considered occupied (BirdLife International 2007). Furthermore, the fringe-backed fire-eye has not been located at several sites from where it was previously known in Bahia (del Hoyo et al. 2003). The fringe-backed fire-eye is formally recognized as ''Endangered'' in Brazil and is directly protected by legislation (Collar et al. 1992; BirdLife International 2007; ECOLEX 2007), which prohibits or regulates international trade, hunting, collection, research, captive propagation, and general harm to the species. However, the greatest threat to the species continues to be habitat loss (BirdLife International 2007). Threats to the fringe-backed fire-eye and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Brown-Banded Antpitta (*Grallaria milleri*)

The brown-banded antpitta is endemic to the Volcan Ruíz-Tolima massif of the central Andes (Caldas, Risaralda, Quindío, and Tolima), Colombia (BirdLife International 2007). The species inhabits humid understory and forest floors of mid-montane and cloud forests between 5,905 and 8,530 ft (1,800 and 2,600 m) in areas with a high density of herbs and shrubs (del Hoyo et al. 2003; Kattan and Beltrán 1999). The species' current range is estimated to be 116 mi^2 (300 km^2) (BirdLife International 2007g). The species is known today in three areas in the upper Río Magdalena Valley: (1) The humid forests in the Central Andes of Colombia's Ucumarí Regional Park (Risaralda Department); the site is approximately 17 mi^2 (44 km^2) in the Otún River watershed (Kattan and Beltrán 1999); (2) the south-east slope of Volcán Tolima in the Río Toche Valley on private land (Tolima Department); this location is 0.02 mi^2 (0.05 km^2) in size at elevations ranging from 9,022 to 9,514 ft (2,750 to 2,900 m) (Beltrán and Kattan 2002); and (3) the Río Blanco river basin (Caldas Department); the site is a strip of land less than 124 linear mi (200 linear km) on the Central Cordilla, between 7,546 and 10,171 ft (2,300 and 3,100 m) in elevation (Kattan and Beltrán 2002).

Between the years 1911 and 1942, only 10 specimens were collected at elevations of 9,004 to 10,299 ft (2,745 to 3,140 m) in Caldas and Quindío (Kattan and Beltrán 1997). The species was not seen for more than 50 years, until it was rediscovered in May 1994, in Ucumarí Regional Park, Risaralda (Kattan and Beltrán 1997). Surveys conducted between 1994 and 1997 estimated that 106 individuals were present in a 0.24 mi^2 (0.63 km^2) area (Kattan and Beltrán 1997, 1999). Further observations of the species were made during 1998–2000 on the southeast slope of Volcán Tolima in the Río Toche Valley, where it is considered uncommon and local (López-Lanús et al. 2000, López-Lanús in litt. 2000, and P.G.W. Salaman in litt. 1999, 2000, as cited in BirdLife International 2007; Renjifo et al. 2002). A census of the population in the Río Blanco river basin was undertaken in June 2000. Researchers estimated the presence of at least 30 individuals, based on vocalizations they elicited in response to recordings of the species' alarm call (Beltrán and Kattan 2002).

The population of brown-banded antpitta is estimated by the IUCN to be between 250 and 999 birds (BirdLife International 2007). It is estimated that the species has lost up to 9 percent of its population in the last 10 years, or 3 generations, and that this rate of decline will continue over the next 10 years (BirdLife International 2007).

The IUCN has classified the brown-banded antpitta as ''Endangered'' since 1994, because it is known from very few locations, occupies a very small range, and habitat loss and degradation are continuing (BirdLife International 2007). It is identified as an ''Endangered'' species under Colombian law pursuant to paragraph 23 of Article 5 of the Law 99 of 1993 as outlined in Resolution No. 584 of 2002 (ECOLEX 2007).

Deforestation has greatly affected the current population size and distributional range of the brown-banded antpitta. Nearly all the other forested habitat below 10,827 ft (3,300 m) in the Central Andes, where the brown-banded antpitta occurred historically, has been deforested and cleared for agricultural land use (BirdLife International 2007). The remaining forests providing suitable habitat for the brown-banded antpitta have become fragmented and isolated and are either surrounded by, or being converted to, pasture and agricultural crops (e.g. , coffee plantations, potatoes, beans) (Beltrán and Kattan 2002; BirdLife International 2007; Collar et al. 1992; Kattan and Beltrán 1997; Kattan and Beltrán 2002). By 1998, approximately 85 percent of forested habitat at altitudes between 6,234 ft (1,900 m) and 10,499 ft (3,200 m), where the species is most likely to be found, had been converted to other land uses (BirdLife International 2007; Cuervo 2002; Stattersfield et al. 1998), and forest conversion has continued. Cuervo (2002) estimated that the available suitable habitat for this species totals no more than 310 mi^2 (500 km^2), although the species is estimated to only occupy an area 116 mi^2 (300 km^2) in size (BirdLife International 2007). Threats to the brown-banded antpitta and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

Kaempfer's Tody-Tyrant (*Hemitriccus kaempferi*, previously known as *Idioptilon kaempferi*)

The Kaempfer's tody-tyrant is very rare and has a very small, extremely fragmented range in Brazil which is estimated to be about 7.3 mi^2 (19 km^2) (BirdLife International 2007). The species is only known from three localities in Santa Catarina, Brazil (with recent records from just two): one record at Salto do Piraí near Villa Nova in 1929, one specimen that was collected at Brusque in 1950, and another in Reserva Particular do Patrimônio Natural de Volta Velha, near Itapoá in 1998 (Barnett et al. 2000; L.N. Naka in litt. 1999; as cited in BirdLife International 2007). It inhabits humid lowland Atlantic forest. At one of these localities, Salto do Piraí, the species has typically been found in habitats which include forest edge, well-shaded secondary growth, and sections of low, epiphyte-laden open woodland near watercourses (Barnett et al. 2000). It feeds predominantly in the midstory of medium-sized trees, and mated pairs appear to remain within small, well-defined areas (Barnett et al. 2000).

In 2004, the IUCN changed the Kaempfer's tody-tyrant's decade-long classification on the Red List from ''Endangered'' to ''Critically Endangered,'' because the species has an extremely small and fragmented range, with recent records from only two locations, and ongoing deforestation is occurring in the vicinity of these sites (Birdlife International 2007). The population estimate is 1,000 to 2,499 individuals and declining (BirdLife International 2007). The Atlantic forest has been extensively deforested, and the lowland forest continues to be cleared in the vicinity of the two remaining sites (BirdLife International 2007; Höfling 2007; The Nature Conservancy 2007). The Kaempfer's tody-tyrant is protected by Brazilian law. These protections prohibit the following activities with regard to this species: export and international trade, collection and research, captive propagation, and also provide measures which help to protect remaining suitable habitat, such as prohibition of exploitation of the remaining primary forests within the Atlantic forest biome and management of various practices in primary and secondary forests, such as logging, charcoal production, reforestation, recreation, and water resources (ECOLEX 2007). The species is restricted to one 15 km^2 (6 mi^2) protected area and in adjacent forest (Barnett et al. 2000; BirdLife International 2007). This habitat area is insufficient for the long-term survival of the Kaempfer's tody-tyrant, particularly since, for various reasons (e.g., lack of funding, personnel, or local management commitment), Brazil's current capacity to achieve its stated natural resource objectives in protected areas is limited (ADEJA 2007; Bruner et al. 2001; Costa 2007; IUCN 1999; Neotropical News 1996; Neotropical News 1999). Therefore, even with the expansion or further designation of protected areas, it is likely that not all of the identified resource concerns for the Kaempfer's tody-tyrant (e.g., residential and agricultural encroachment, resource extraction, unregulated tourism, grazing) would be sufficiently addressed at these sites.

Threats to the Kaempfer's tody-tyrant and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Ash-Breasted Tit-Tyrant (*Anairetes alpinus*)

The ash-breasted tit-tyrant is a small New World flycatcher (family *Tyrannidae*) (del Hoyo et al. 2004), confined to humid *Polylepis* forests in the Andes Mountains of Peru and Bolivia (BirdLife International 2007; Collar et al. 1992; Fjeldså and Krabbe 1990; InfoNatura 2007). *A. alpinus* consists of two subspecies, the nominate subspecies, *A. alpinus alpinus*, which occurs on the west Andean slope in northern Peru (Ancash, La Libertad), and *A. alpinus bolivianus*, which occurs in southeast Peru (Cuzco, Apurimac) and northwest Bolivia (La Paz) (BirdLife International 2007; del Hoyo et al. 2004).

Historically, the ash-breasted tit-tyrant may have been well-distributed in the previously large, contiguous expanses of *Polylepis* forest of the high-Andes of Peru and Bolivia (Fjeldså 2002a); however, it is now restricted to remnant patches of these forests in Peru (Cuzco, Apurimac, and Corredor Conchucos) and Bolivia (La Paz) (Birdlife International 2007; Collar et al. 1992; Fjeldså and Krabbe 1990; InfoNatura 2007).

The ash-breasted tit-tyrant is restricted to high-elevations—12,139 to 15,092 ft above sea level (3,700 to 4,600 m) (del Hoyo et al. 2004). Individuals forage alone, in pairs, groups of three, and occasionally in mixed-species flocks, making short trips to hover-glean or perch-glean near the tops and outer edges of *Polylepis* spp. shrubs and trees (del Hoyo et al. 2004; Engblom et al. 2002). We are unaware of any information that is available on the breeding behavior of the species. Juveniles have been observed in March and July around Cuzco, Peru (del Hoyo et al. 2004).

The ash-breasted tit-tyrant has been described as generally quite rare and local, with one to two pairs per occupied woodland (Fjeldså and Krabbe 1990). BirdLife International (2007) and Fjeldså (2002b) placed the population somewhere between 250 to 1,000 individuals. Gomez (2005, in litt. 2007) conducted intensive searches using song playback within 80 percent of the suitable habitat in Bolivia and found 180 individuals distributed within 14 forest patches. Chutas (2007) reported only 461 individuals, based on detailed surveys of suitable habitat, which contained the highest concentration of *Polylepis* forest in southeastern Peru.

The IUCN categorizes the ash-breasted tit-tyrant as ''Endangered'' because of its very small population, which is confined to a severely fragmented habitat undergoing a continuing decline in extent, area, and quality (BirdLife International 2007). The ash-breasted tit-tyrant is considered an ''Endangered'' species by the Peruvian government under Supreme Decree No. 034–2004–AG which prohibits hunting, taking, transport, or trade of this species, except as permitted by regulation. However, the species' habitat is not protected by this law. We are not aware of any regulations in Bolivia that are effective at protecting the habitat of the ash-breasted tit-tyrant.

The principal factor affecting the distribution of *Polylepis* forests, the species' habitat, is the intensity of burning and grazing, which restrict vegetation growth to locations where fires cannot spread, and cattle and sheep do not normally roam, such as ravines, boulders, rock ledges, and sandy ridges (Fjeldså 2002a and b). Many farmers, however, destroy *Polylepis* forests to plant crops, even on steep hillsides unsuitable for cultivation (Hensen 2002). Harvesting of firewood from *Polylepis* forests is also a significant threat to the ash-breasted tit-tyrant's habitat (Aucca and Ramsay 2005; Engblom in litt. 2000). Trampling and grazing by sheep and cattle limit forest regeneration and can contribute to degradation of remaining forest patches (Fjeldså 2002a). Remaining forest fragments are becoming more accessible to the expanding population around Bolivia's largest city through road building and mining projects, further threatening the survival of *Polylepis* forests upon which the ash-breasted tit-tyrant depends (Purcell et al. 2004; Purcell and Brelsford 2004).

The ash-breasted tit-tyrant is completely dependent upon high-elevation humid *Polylepis* forest for survival, and the ongoing loss of this habitat is believed to be the primary threat to this species. Less than 1 percent of this forest habitat remains in the humid highlands, where *Polylepis* forests can grow to be tall and dense (Fjeldså 2002a), providing habitat for the ash-breasted tit-tyrant. Only about 1,554 ac (629 ha) of habitat remain for the ash-breasted tit-tyrant in Cuzco and Apurimac, Peru (Chutas 2007), and 1,245 ac (504 ha) of *Polylepis* forest remains in La Paz, Bolivia (Purcell and Brelsford 2004). Habitat estimates for Corredor Conchucos (Peru), the area occupied by the northern ash-breasted tit-tyrant subspecies (*A. alpinus alpinus*), are not available, but Chutas (2007) reported only 30 individuals from this area. In Bolivia, approximately 507 ac (205 ha) of habitat have been destroyed by clear-cutting since the early 1990s; if the current rate of deforestation continues, projections indicate that all of the *Polylepis* forest in Bolivia will be destroyed within the next 3 decades (Purcell and Brelsford 2004). The rate of habitat decline is lower north of Cuzco, Peru (Cordillera de Vilcanota), with the loss of only 1 percent of forest patches from 1956 to 2005; however, the remaining habitat patches in this area were already quite small (mean patch size is 6.2 ac (2.5 ha)), and 10 percent of forest patches showed a decline in forest density over this time period, indicating that habitat degradation might be more problematic to the species than total destruction of forests in this area (Jameson and Ramsay 2007). Threats to the ash-breasted tit-tyrant and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Peruvian Plantcutter (*Phytotoma raimondii*)

The Peruvian plantcutter is endemic to the coastal desert of northwestern Peru, from sea level to 1,640 ft (500 m) (del Hoyo et al. 2004). The species is restricted to Peru's Talara region, which contains 60 to 80 percent of the population and highly fragmented forest patches around the Chiclayo area of Lambayeque (del Hoyo et al. 2004). BirdLife International (2007) estimates the total population to range between 500 and 1,000 individuals.

Peruvian plantcutters inhabit sparse desert scrub and coastal dunes scattered with large shrubs (del Hoyo et al. 2004). They also occupy riparian thickets and woodlands dominated by *Prosopis* spp. and *Acacia* spp. (del Hoyo et al. 2004). This species appears to prefer a high diversity of plant species, including specific shrubs and trees with low-hanging branches (Elton 2004; Williams 2005). Plantcutters are the only passerines with a predominantly leaf-eating diet (Bucher et al. 2003).

The Peruvian plantcutter is categorized as ''Endangered'' by the IUCN Red List due to ongoing habitat destruction and continuing degradation of its small and severely fragmented range (BirdLife International 2000; BirdLife International 2007). The Peruvian plantcutter is listed as ''Endangered'' by the Peruvian government under Supreme Decree No. 034–2004–AG which prohibits hunting, taking, transport, or trade of endangered species, except as permitted by regulation. However, the species' habitat is not protected by this law.

The major threat to the Peruvian plantcutter is believed to be loss of habitat due to agriculture, burning, grazing, timber cutting, and human use. Extirpation of the species from many sites occurred as conversion of heavily wooded coastal river valleys to irrigated agriculture took place (Lanyon 1975; Collar et al. 1992). Extensive stands of small- to medium-size trees, such as mesquite (*Prosopis spp.*), acacia (*Acacia* spp.), willow (*Salix* spp.), and *Capparis* spp., previously occupied the river valleys, but wooded areas are now confined to land where the lack of irrigation discourages cultivation (del Hoyo et al. 2004; Williams 2005). The remaining forest fragments are threatened by burning, grazing, timber cutting, firewood and charcoal production, and ongoing conversion for cultivation, primarily sugarcane. These factors are believed to have contributed to the destruction of previously occupied plantcutter habitat, which reduced or eliminated forage and nesting sites necessary for the species to thrive (BirdLife International 2000; del Hoyo et al. 2004).

Talara, owned by PetroPeru, the State-owned petroleum company, retains the largest contiguous area of intact habitat currently occupied by the Peruvian plantcutter. PetroPeru strictly bans trespassing; therefore, the population in this area has not been exposed to the same risk factors that it is subject to in the other forested areas. Estimates of the amount of habitat suitable for the plantcutter at Talara vary widely, from 123,553 ac (50,000 ha) (del Hoyo et al. 2004) to 4,942 ac (2,000 ha) (Williams 2005). Talara supports approximately 400 to 600 individuals or 60 to 80 percent of the global population of Peruvian plantcutters (del Hoyo et al. 2004; Williams 2005). Although PetroPeru historically held the land rights to the whole province of Talara, the land is now reverting to the Peruvian government, which is selling it to buyers who are likely to develop the beachfront property (Elton 2004). Attempts to create a protected reserve for the plantcutter on approximately 12,000 ac (4,860 ha) around Talara are reportedly not progressing as originally proposed (Elton 2004; Williams 2005). Future land-cover projections from the Millennium Ecosystem Assessment indicate that by 2050, 11 to 16 percent of the Peruvian plantcutter's range is likely to be unsuitable for the species (Jetz et al. 2007). Threats to the Peruvian plantcutter and its habitat continue, and we find that proposing this species for listing under the Act is warranted.

St. Lucia Forest Thrush (*Cichlhermina lherminieri sanctaeluciae*)

The St. Lucia forest thrush is endemic to the island of St. Lucia in the West Indies (Raffaele et al. 1998). This subspecies occupies mid- and high-altitude primary and secondary moist forest habitat in the coastal areas of the island. The St. Lucia forest thrush feeds on insects and berries that are found from ground level all the way up into the forest canopy (Raffaele 1998). The island of St. Lucia encompasses 151,905 ac (61,500 ha). Of this area, 31,048 ac (12,570 ha) are natural forest, 56 percent of which is located in Forest Reserves and the remaining 43 percent of forest is situated on private lands (Delegation of the European Commission 2004). Commercial harvest of timber is allowed on private land, but it is strictly prohibited within the Forest Reserves (Forestry Department Proceedings 2000).

Although the St. Lucia forest thrush's population was considered numerous in the late-1800s (Keith 1997), the subspecies' current population status is unknown. Recent sightings are rare, with only six confirmed sightings during the last few years (Dornelly 2007). The entire species of forest thrush (*Cichlhermina lherminieri*) is classified as ''Vulnerable'' by the IUCN Red List due to human-induced deforestation and introduced predators (IUCN 2006). The St. Lucia forest thrush is a fully protected species under St. Lucia's Wildlife Protection Act (WPA) of 1980 (Schedule 1), which has prohibited hunting of the subspecies since 1980. In addition, the WPA prohibits taking, damaging or destroying nests, eggs, or offspring of a fully protected species.

Identified risks to this species include habitat loss, competition with the bare-eyed robin (*Turdus nudigenis*), brood parasitism by the invasive shiny cowbird (*Molothrus bonarientsis*), hunting by humans for food, and predation by mongoose and other introduced predators (Raffaele et al. 1998). The demand for agricultural land on St. Lucia has resulted in deforestation; approximately 33.7 percent of the island is under agricultural production (GOSL 2000). Another contributing factor to habitat loss is soil erosion. Approximately 80 percent of the island is composed of steep terrain, and poor agricultural practices have resulted in excessive soil erosion and loss of soil productivity, two factors which contribute to destruction of forest habitat in some areas and degradation of forest habitat in other locations (Bond 1990). Traditionally, forest resources have been used for many household products in daily use on St. Lucia. Currently, heating and cooking in the homes of island residents utilize forest resources; charcoal and firewood use combined account for 83 percent of St. Lucia's fuel supply (Forestry Department Proceedings, 2000).

Tropical storms and hurricanes frequently occur in the Caribbean Sea, and can have severe, widespread impacts on the terrestrial ecosystems of small islands. High winds are a primary threat to forest habitats due to the damage caused to the trees. They are often blown over or sustain severe damage to trunks and limbs, which can result in critical habitat loss to the St. Lucia forest thrush. During the last three decades, there has been an increase in the number of hurricanes and severe tropical storms experienced by St Lucia. After hurricane Allen in 1980, at least 55 percent of all dominant tree species had broken branches and many trees lost large portions of their crowns (Whitman 1980, as reported in GOSL 1993). Threats to the St. Lucia forest thrush are ongoing, and we find that proposing this species for listing under the Act is warranted.

Eiao Polynesian Warbler (*Acrocephalus percernis aquilonis*, previously known as *Acrocephalus mendanae aquilonis* and *Acrocephalus caffer aquilonis*)

The reed warblers of Polynesia have been divided into two species, the Tahiti reed-warbler (*Acrocephalus caffer*) and the Marquesas-reed warbler (*Acrocephalus mendanae*) (Birdlife International 2007a and b). However, new genetic research using mitochondrial DNA markers to develop a phylogeny of the eastern Polynesian taxa of reed-warblers of the Marquesas Archipelago has led to further proposed taxonomic changes for the reed-warblers on these islands. This proposed change separates the reed-warblers on the four northernmost islands in the Marquesas Archipelago into a separate species (*Acrocephalus percernis*) from those on the southern islands (*Acrocephalus mendanae*). The proposed taxonomic change maintains the subspecies delineations between the islands; the reed-warblers on Eiao Island remain a subspecies, now renamed *Acrocephalus percernis aquilonis* (Cibois et al. 2007).

The Eiao Polynesian warbler is endemic to a single island (Eiao) in the Marquesas Archipelago of French Polynesia in the Pacific Ocean. The Marquesas Archipelago is one of the most remote island chains in the world, lying between 404 and 600 mi (650 and 965 km) south of the equator and approximately 994 mi (1,600 km) northeast of Tahiti. Eiao Island is one of the northernmost islands in the Archipelago, encompassing 17 mi^2 (43.8 km^2) in area, and ranging in altitude from sea level to 1,890 ft (576 m) (Wikipedia 2007). The Eiao Polynesian warbler's preferred habitat is dry forest (Raust 2007).

Population densities of the Eiao Polynesian warbler are thought to be high within remaining suitable habitat, based on a recent study which found individual singing birds approximately every 130 to 165 ft (40 to 50 m). Total numbers are estimated to be greater than 2,000 birds (Dr. P. Raust, pers. comm. to Amedee Brickey, USFWS 2007). This estimate is much higher than the 100 to 200 individuals estimated in 1987 by Thibault (as previously cited in USFWS 2007). It is not clear if the subspecies' population actually increased from 1987 to 2007, or if the different population estimates can be attributed to the use of different survey methodologies. We have no reliable information on the population trend of this subspecies. The Eiao Polynesian warbler is a protected subspecies in French Polynesia. The conservation status of this newly designated subspecies has not been categorized on the IUCN Red List.

Although currently uninhabited by humans, Eiao Island's natural vegetation has been heavily impacted by introduced domestic livestock (sheep and swine); part of the island has even been denuded of all vegetation. As a result, only 10 to 20 percent of the island contains the Eiao Polynesian warbler's preferred dry forest habitat (Raust 2007). Suitable subspecies' habitat is limited to steep slopes that are inaccessible to domestic livestock. While Eiao Island was declared a Nature Reserve by French Polynesia in 1992, we are not aware of any plans to protect the habitat of the Eiao Polynesian warbler.

Introduced mammals and birds have been implicated in loss of endemic birds in the Marquesas and may impact the Eiao Polynesian warbler. Two species of nonnative rats, the Polynesian rat (*Rattus excluans*) and the black rat, were introduced to Eiao Island during the late nineteenth century (Thibault and Myers 2000, as reported in Thibault et al. 2002) and are thought to have contributed to the decline of the Eiao Polynesian warbler. However, recent research indicates that reed-warblers in the Marquesas Archipelago nest sufficiently high in trees to avoid significant predation from rats (Thibault et al. 2002). The most destructive introduced avian predator in the Marquesas, the common myna (*Acridotheres tristis*), has not been found on Eiao Island. If the myna expands its range and colonizes Eiao Island, there is a chance it could impact the Eiao Polynesian warbler (Thibault et al. 2002).

Another potential risk to the Eiao Polynesian warbler is destruction of habitat by tsunamis and cyclones. French Polynesia, and in particular the Marquesas Archipelago, are frequently affected by tsunamis; the waves observed in the Marquesas are generally 2 to 10 times higher than waves recorded in Tahiti (Hebert et al. 2001). The Eiao Polynesian warbler is also exposed to high winds during tropical cyclones, which often displace individuals. Indirect effects occur during the aftermath of a storm when subspecies are impacted by the loss of food supplies, foraging substrates, and roost sites, increasing their vulnerability to predators and disease. Large-scale climate models predict increased intensity of tropical cyclones impacting island chains in the Pacific, including the Marquesas Archipelago (Meehl et al. 2007). Threats to this subspecies and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Medium Tree-Finch (*Camarhynchus pauper*)

The medium tree-finch is endemic to Floreana in the Galapagos Islands, Ecuador (BirdLife International 2007). Its habitat is montane evergreen and tropical deciduous forest (Stotz et al. 1996), primarily above 328 ft (100 m). Population numbers of this species are poorly known, with indirect estimations at 1,000 to 2,499 birds (BirdLife International 2007). However, Stotz et al. (1996) consider the relative abundance of the species to be ‘‘common.’’ Population trends are unknown.

This poorly known species is considered ‘‘Vulnerable’’ by the IUCN because it has a very small range and is restricted to a single island where introduced species are a potential threat (BirdLife International 2004) due to herbivore degradation and loss of habitat and possibly predator-caused mortality (BirdLife International 2007; Jackson 1985). In addition, agricultural activities (Cruz and Cruz 1996) and free-ranging domestic livestock continue to destroy and degrade the habitat of the medium tree-finch (BirdLife International 2007). The recent discovery of an introduced parasitic fly (*Philornis downsi*) on Floreana Island (Kleindorfer et al. MS, as cited in Grant et al. 2005) has raised concerns about the impact this parasite might be having on the medium tree-finch (Fessl et al. 2006). In an experimental study conducted on nearby Santa Cruz Island, Fessl et al. (2006) found that high mortality of nestlings was directly attributable to parasitism by *P. downsi*, as evidenced by a near threefold increase in fledgling success in a parasite-reduced group versus a parasite-infested control group. Further, because species with small broods have been found to suffer higher parasite loads and therefore higher nestling mortality (Fessl and Tebbich 2002), infestation of *P. downsi* on species with naturally low clutch sizes, such as the medium tree-finch, is of particular concern (Fessl et al. 2006).

In 1959, Ecuador designated 97 percent of the Galapagos land area as a National Park, leaving 3 percent of the remaining land area distributed between Santa Cruz, San Cristóbal, Isabela, and Floreana Islands. National Park protection, however, does not mean the area is to be maintained in a pristine condition. The park land area is divided into various zones signifying the level of human use (Parque Nacional Galapagos Ecuador n.d.). Although Floreana Island includes a large ‘‘conservation and restoration’’ zone, it does include a significant sized ‘‘farming’’ zone (Parque Nacional Galapagos Ecuador n.d.), where agricultural and grazing activities may continue to impact the habitat.

The Galapagos Islands were declared a World Heritage Site in 1979, as they were recognized to be ‘‘cultural and natural heritage of outstanding universal value.’’ The aim of establishment as a WHS is conservation of the site for future generations (UNESCO World Heritage Centre 2008). However, due to threats to this site posed by invasive species, increasing tourism, and immigration, in June, 2007, the World Heritage Committee placed the Galapagos on the ‘‘List of World Heritage in Danger,’’ with the intent of increasing support for their conservation (UNESCO World Heritage Centre News 2007). In March 2008, the UNESCO World Heritage Centre/United Nations Foundation project for invasive species management provided funding of 2.19 million U.S. dollars (USD) to the Ecuadorian National Environmental Fund's ‘‘Galapagos Invasive Species’’ account to support invasive species control and eradication on the islands. In addition, the Ecuador government previously had contributed 1 million USD to this fund (UNESCO World Heritage Centre News 2008), demonstrating the government of Ecuador's commitment to reducing the threat of invasive species to the islands. At the present time, however, threats to the medium tree-finch and its habitat caused by introduced species continue, and we find that proposing this species for listing under the Act is warranted.

Cherry-Throated Tanager (*Nemosia rourei*)

The cherry-throated tanager inhabits primary forest habitats in Espírito Santo and, possibly, Minas Gerais and Rio de Janeiro, Brazil (Bauer et al. 2000; BirdLife International 2007; Venturini et al. 2005). Because the cherry-throated tanager was only known from a single specimen collected in the 1800s and a reliable sighting of eight individuals from 1941, the species was presumed to be extinct (Collar et al. 1992; Ridgely and Tudor 1989; Scott and Brooke 1985). However, the species was rediscovered in 1998 (Bauer et al. 2000; Venturini et al. 2005). Since then, the cherry-throated tanager has been documented at three sites of remnant primary forest in south-central Espírito Santo (Bauer et al. 2000; Scott 1997; Venturini et al. 2005). Two of the currently occupied sites are in private ownership and the third, which is believed to be used only sporadically by the species, is within the Augusto Ruschi Biological Reserve (Venturini et al. 2005).

The cherry-throated tanager is endemic to the Atlantic Forest biome and inhabits the upper canopies of trees within humid, montane, primary forests (Bauer et al. 2000; BirdLife International 2007; Venturini et al. 2005). It is a primary forest-obligate species that typically forages for insects within the interior crowns of tall, epiphyte-laden trees and occasionally lower down—ca. 6.6 ft (2 m)—at the forest edge (Bauer et al. 2000; BirdLife International 2007; Venturini et al. 2005). Cherry-throated tanagers can be found in mixed-species flocks and appear to require relatively large territories—ca. 1.544 mi^2 (3.99 km^2) (Venturini et al. 2005). Within its current distribution, the species makes sporadic use of coffee (*Coffea* spp.), pine (*Pinus* spp.), and eucalyptus (*Eucalyptus* spp.) plantations, presumably as travel corridors between remaining patches of primary forest (Venturini et al. 2005). Little is known about the breeding behavior of the cherry-throated tanager (Venturini et al. 2002).

The IUCN categorizes the species as ‘‘Critically Endangered’’ because its extant population is estimated to be between 50 and 249 individuals. The population is extremely small and highly fragmented, and presumed to be declining (BirdLife International 2007). There is even speculation that the IUCN population estimate is too high, considering that the maximum number of individuals recorded in the only 2 confirmed populations is 19 (Venturini et al. 2005).

Based on a number of recent estimates, 92 to 95 percent of the area historically covered by tropical forests within the Atlantic Forest biome has been converted or severely degraded as a result of various human activities (The Nature Conservancy 2007; Höfling 2007). In addition to the overall loss and degradation of native habitat within this biome, the remaining tracts of habitat are severely fragmented. Most of the tropical forest habitats believed to have been used historically by the cherry-throated tanager have been converted or severely degraded by human activities (Bauer et al. 2000; BirdLife International 2007; Ridgely and Tudor 1989). Even when they are formally protected, the remaining fragments of primary forest habitat where the species may still occur will likely undergo further degradation due to their altered dynamics and isolation between forest fragments (Tabanez and Viana 2000).

The cherry-throated tanager is formally recognized as ''Endangered'' in Brazil and is directly protected by legislation promulgated by the Brazilian government (BirdLife International 2007; ECOLEX 2007). These protections prohibit the following activities with regard to this species: Export and international trade, collection and research, captive propagation, and also provide measures which help to protect remaining suitable habitat, such as prohibition of exploitation of the remaining primary forests within the Atlantic forest biome and management of various practices in primary and secondary forests, such as logging, charcoal production, reforestation, recreation, and water resources (ECOLEX 2007). The owners of Fazenda Pindobas IV and Caetes, two sighting areas, have cooperated in protecting cherry-throated tanager habitat in these areas, and efforts are underway to solidify protection of these privately owned areas (BirdLife International 2007; Venturini et al. 2005). Elsewhere, for various reasons (e.g., lack of funding, personnel, or local management commitment), Brazil's current capacity to achieve its stated natural resource objectives in protected areas is limited (ADEJA 2007; Bruner et al. 2001; Costa 2007; IUCN 1999; Neotropical News 1996; Neotropical News 1999). Therefore, even with the further designation of protected areas, it is likely that not all of the identified resource concerns for the cherry-throated tanager (e.g., residential and agricultural encroachment, resource extraction, unregulated tourism, grazing) would be sufficiently addressed.

Threats to the cherry-throated tanager and its habitat are ongoing, and we find that proposing this species for listing under the Act is warranted.

Findings on Species for Which Listing Is Warranted but Precluded

We have found that, for the 20 taxa discussed below, publication of proposed listing rules will continue to be precluded over the next year due to the need to complete pending, higher-priority listing actions. We will continue to monitor the status of these species as new information becomes available (see Monitoring, below). Our review of new information will determine if a change in status is warranted, including the need to emergency list any species or change the LPN of any of the species.

Birds

Southern Helmeted Curassow (*Pauxi unicornis*)

The southern helmeted curassow is known from central Bolivia and central and eastern Peru (Collar et al. 1992). In Bolivia, the subspecies (*P. unicornis unicornis*) is known from the adjacent Amboró and Carrasco National Parks (Herzog and Kessler 1998). The southern helmeted curassow is one of the least frequently encountered bird species in South America because of the inaccessibility of its preferred habitat and its apparent intolerance of human disturbance (Herzog and Kessler 1998). It has been reported from only two Peruvian and three Bolivian localities, which are fairly close together (Collar et al. 1992; Cox et al., as cited in Herzog and Kessler 1998). In Bolivia, it remained unknown to science until 1937 (Cordier 1971). In Amboró National Park, the curassows are sighted regularly on the upper Rio Saguayo (Wege and Long 1995). Field surveys on the Peru-Bolivia border, including one in 2004, have failed to locate any birds (BirdLife International 2007a; Herzog et al. 1999; Herzog and Kessler 1998; Mee et al. 2000), and limited local reports suggest that the bird is rare (Herzog et al. 1999; Herzog and Kessler 1998). In 2005, a team from Armonia Association (BirdLife in Bolivia) saw one and heard three southern helmeted curassows (*P. unicornis koepckeae*) in the Sira Mountains of central Peru—this is the first sighting of the distinctive endemic Peruvian race since 1969 (BirdLife International 2008).

The southern helmeted curassow inhabits dense, humid, lower montane forest and adjacent evergreen forest at 1,476 to 3,936 ft (450 to 1,200 m) (Cordier 1971; Herzog and Kessler 1998). This species prefers nuts of the almendrillo tree (*Byrsonima wadsworthii*) as its major source of food (Cordier 1971). It also consumes other nuts, seeds, fruit, soft plants, larvae, and insects (BirdLife International 2000).

The southern helmeted curassow was previously classified as ''Vulnerable'' on the IUCN Red List. After further assessment, it was uplisted in 2005 to ''Endangered'' because the species is estimated to be declining very rapidly due to uncontrolled hunting and habitat destruction. It has a small range and is known from few locations in a narrow elevational band, which continues to be subject to habitat loss (BirdLife International 2004). The population is estimated at 10,000 to 19,999 birds, with a future projected decline over the next 10 years or 3 generations of 50 to 79 percent (BirdLife International 2007b). Professional hunters have caused a decline in this species in Bolivia; the species is often hunted for meat and its casque, or horn (Collar et al. 1992), which the local people use to fashion cigarette-lighters (Cordier 1971). Other risks to the species include forest clearing for staple and export crops, road building, and rural development (Dinerstein et al. 1995, as cited in BirdLife International 2007a; Fjeldså in litt. 1999, as cited in BirdLife International 2007a; Herzog and Kessler 1998). In Peru, potential oil exploration threatens the species' habitat (MacLeod in litt. 2000, as cited in BirdLife International 2007a) and is opening the foothills to colonization and additional hunting (BirdLife International 2007a).

Large parts of the southern helmeted curassow's range are protected, at least on paper, by inclusion in the Amboro and Carrasco National Parks (300,000 ha (750,000 ac) and 616,413 ha (1,175,000 ac), respectively), which nominally protect the species from hunting and declining habitat resulting from development and road-building, although hunting of the curassow for meat is still reported throughout its range (BirdLife International 2000). The Association Armonia has being conducting field surveys to estimate the population and identify the most important sites for this species, and are evaluating human impact on the species' natural habitat (Llampa 2007). In addition, Armonia is carrying out an environmental awareness project to inform local people about this unique bird (BirdLife Intenational 2008) and training workshops with the park guards (Llampa 2007).

The southern helmeted curassow does not represent a monotypic genus. It faces threats that are moderate in magnitude as the population is fairly large; however, the population trend has been declining rapidly. The threats to the species are imminent and ongoing. Therefore, it receives a priority rank of 8.

Bogota Rail (*Rallus semiplumbeus*)

The Bogota rail is found in the East Andes of Colombia on the Ubaté-Bogotá Plateau in Cundinamarca and Boyacá. It occurs in the temperate zone, at 2,500–4,000 m (8,202–13,123 ft) (occasionally as low as 2,100 m (6,890 ft)) in savanna and páramo marshes (BirdLife International 2007). This rail frequents wetland habitats with vegetation-rich shallows that are surrounded by tall, dense reeds and bulrushes. It feeds along the water's edge, in flooded pasture land, and along small overgrown dikes and ponds (Varty et al. 1986; Fjeldså and Krabbe 1990 as cited in BirdLife International 2006). This species is omnivorous, consuming a diet that includes aquatic invertebrates, insect larvae, worms, molluscs, dead fish, frogs, tadpoles, and plant material (Varty et al. 1986; BirdLife International 2006).

The Bogota rail is listed as ''Endangered'' by IUCN, primarily because its range is very small and is contracting due to widespread habitat loss and degradation. Furthermore, available habitat has become widely fragmented (BirdLife International 2007). The Ubaté-Bogotá Plateau formerly held enormous marshes and swamps, but few lakes with suitable habitat now remain. All major savanna wetlands are seriously threatened, mainly by drainage, but also by agricultural encroachment, erosion, diking, eutrophication, insecticides, tourism and hunting activities, burning, trampling by cattle, harvesting of reeds, fluctuating water levels, and increased water demand (BirdLife International 2007). The current population is estimated to range between 1,000 and 2,499 individuals, and the trend is decreasing (BirdLife International 2007). Although the Bogota rail is declining, it is still uncommon to fairly common, with some notable populations, including nearly 400 birds at Laguna de Tota, some 50 territories at Laguna de la Herrera, approximately 110 birds at Parque La Florida, and other populations at La Conejera marsh and Laguna de Fuquene (BirdLife International 2007). Some of the birds occur in protected areas such as Chingaza National Park and Carpanta Biological Reserve. However, most savanna wetlands are virtually unprotected.

The Bogota rail does not represent a monotypic genus. Because there are still a number of substantial subpopulations and the species has been recorded at over 21 localities, we find it is subject to threats that are moderate in magnitude. We find that the threats are imminent due to the ongoing degradation of the species' wetland habitat. Therefore, it receives a priority rank of 8.

Takahe (*Porphyrio hochstetteri,* previously known as *P. mantelli*)

The Takahe, a flightless rail endemic to New Zealand, is the world's largest extant member of the rail family (del Hoyo et al. 1996). The species, *Porphyrio mantelli,* has been split into *P. mantelli* (extinct) and *P. hochstetteri* (extant) (Trewick 1996). BirdLife International (2000) incorrectly assigned the name *P. mantelli* to the extant form, while the name *P. hochstetteri* was incorrectly assigned to the extinct form. Fossils indicate that this bird was once widespread throughout the North and South Islands. The Takahe was thought to be extinct by the 1930s until its rediscovery in 1948 in the Murchison Mountains, Fjordland (South Island) (Bunin and Jamieson 1996; New Zealand Department of Conservation (NZDOC) 2008b). Soon after its rediscovery, a Takahe Special Area of 193 mi^2 (500 km^2) was set aside in Fiordland National Park for the conservation of Takahe (Crouchley 1994; NZDOC 2008c). Today, the species is present in the Murchison and Stuart Mountains and has been introduced to four island reserves (Kapiti, Mana, Tiritiri Mantangi, and Maud) (Collar et al. 1994). The population in the Murchison Mountains is important because it is the only mainland population that has the potential for sustaining a large, viable population (NZDOC 1997).

Originally, the species occurred throughout forest and grass ecosystems. Today, Takahe occupy alpine grasslands (BirdLife International 2007). They feed on tussock grasses during much of the year, with snow tussocks (*Chionochloa pallens, C. flavescens,* and *C. crassiuscula*) being their preferred food (Crouchley 1994). By June, the snow cover usually prevents feeding above tree line, and birds move into forested valleys in the winter and feed mainly on the rhizome of a fern (*Hypolepis millefolium*). Research by Mills et al. (1980) suggested that Takahe require the high carbohydrate concentrations in the rhizomes of the fern to meet the metabolic requirement of thermoregulation in the mid-winter, subfreezing temperatures. The island populations eat introduced grasses (BirdLife International 2007). Takahe form pair bonds that persist throughout life and generally occupy the same territory throughout life (Reid 1967). Their territories are large, and Takahe defend them aggressively against other Takahe, which means that they will not form dense colonies even in very good habitat. They are long-lived birds, probably between 14 and 20 years (Heather and Robertson 1997), which have a low reproductive rate, with clutches consisting of 1–3 eggs. Only a few pairs manage to consistently rear chicks each year. Although under normal conditions this is generally sufficient to maintain the population, populations recover slowly from catastrophic events (Crouchley 1994).

The Takahe is listed as ''Endangered'' on the IUCN Red List, because it has an extremely small population (BirdLife International 2006). When rediscovered in 1948, it was estimated that the population was about 260 pairs (del Hoyo 1996; Heather and Robertson 1997). By the 1970s, Takahe populations had declined dramatically and it appeared that the species was at risk of extinction. In 1981, the population reached a low at an estimated 120 birds. Since then, the population has fluctuated between 100 and 180 birds (Crouchley 1994). At first, translocated populations increased only slowly, probably due to young pair-bonds and the quality of the founding population (Bunin et al. 1997). In recent years, the total Takahe population has had significant growth; in 2004, there was a 13.6 percent increase in the number of adult birds, with the number of breeding pairs up 7.9 percent (BirdLife International 2005). As of August 2007, birds in the Takahe Special Area had increased to 168, and the current national population was 297. Island reserves appeared to be at carrying capacity (NZDOC 2007). Overall, population numbers are slowly increasing due to intensive management of the island reserve populations, but fluctuations in the remnant mainland population continue to occur (BirdLife International 2000).

The main cause of the species' historical decline was competition for tussock grasses by grazing red deer (*Cervus elaphus*), which were introduced after the 1940s (Mills and Mark 1977). The red deer overgrazed the Takahe's habitat, eliminating nutritious plants and preventing some grasses from seeding (del Hoyo et al. 1996). The NZDOC has controlled red deer through an intensive hunting program in the Murchison Mountains since the 1960s, and now the tussock grasses are close to their original condition (BirdLife International 2005).

Predation by introduced stoats (*Mustela erminea*) is believed to be a current risk to the species (Bunin and Jamieson 1995; Bunin and Jamieson 1996; Crouchley 1994). The NZDOC is running a trial stoat control program in a portion of the Takahe Special Area to measure the effect on Takahe survival and productivity. Initial assessment indicates a positive influence (NZDOC 2007). Other potential competitors or predators include the introduced brush-tailed possum (*Trichosurus vulpecula*) and the threatened weka (*Gallirallus australis*), a flightless woodhen endemic to New Zealand (BirdLife International 2007). In addition, severe weather is a natural limiting factor to this species (Bunin and Jamieson 1995). Weather patterns in the Murchison Mountains vary from year to year. High chick and adult mortality may occur during extraordinarily severe winters, and poor breeding may result from severe stormy weather during spring breeding season (Crouchley 1994). Research confirms that severity of winter conditions adversely affects survivorship of Takahe in the wild, particularly of young birds (Maxwell and Jamieson 1997).

Since 1983, the NZDOC has been involved in managing a captive-breeding and release program to boost Takahe recovery. Excess eggs from wild nests are managed to produce birds suitable for releasing back into the wild population in the Murchison Mountains. Some of these captive-reared birds have also been used to establish four predator-free offshore island reserves. Since 1984, these birds have increased the total population on islands to about 60 birds (NZDOC 2008a). Captive-breeding efforts have increased the rate of survival of chicks reaching 1 year of age from 50 to 90 percent (NZDOC 1997). However, Takahe that have been translocated to the islands have higher rates of egg infertility and low hatching success when they breed, contributing to the slow increase in the islands' populations. Researchers postulated that the difference in vegetation between the native mainland grassland tussocks and that found on the islands might be affecting reproductive success. After testing nutrients from all available food sources, they concluded that there was no effect, and advised that a supplementary feeding program for the birds was not necessary or recommended (Jamieson 2003). Further research on Takahe established on Tiritiri Matangi Island estimated that the island can support up to 8 breeding pairs, but suggested that the ability of the island to support Takahe is likely to decrease as the grass/shrub ecosystem reverts to forest. The researchers concluded that although the four island populations fulfilled their role as an insurance against extinction on the mainland at the time of the study, given impending habitat changes on the islands, it is unclear whether these island populations will continue to be viable in the future without an active management plan (Baber and Craig 2003a; Baber and Craig 2003b). Maxwell and Jamieson (1997) studied survival and recruitment of captive-reared and wild-reared Takahe on Fiordland. They concluded that captive rearing of Takahe for release into the wild increases recruitment of juveniles into the population.

There is growing evidence that inbreeding can negatively affect small, isolated populations. Jamieson et al. (2006) suggested that limiting the potential effects of inbreeding and loss of genetic variation should be integral to any management plan for a small, isolated, highly-inbred island species, such as the Takahe. Failure to address these concerns may result in reduced fitness potential and much higher susceptibility to biotic and abiotic disturbances in the short term and an inability to adapt to environmental change in the long term.

The Takahe does not represent a monotypic genus. The current wild population is small and the species' distribution is extremely limited. It faces threats that are moderate in magnitude because the NZDOC has taken measures to aid the recovery of the species. The NZDOC has implemented a successful deer control program and implemented a captive-breeding and release program to augment the mainland population and establish four offshore island reserves. Predation by introduced species and reduced survivorship resulting from severe winters, combined with the Takahe's small population size and naturally low reproductive rate are threats to this species that are imminent and ongoing. Therefore, this species is assigned a priority rank of 8.

Chatham Oystercatcher (*Haematopus chathamensis*)

The Chatham oystercatcher is endemic to the Chatham Island group (Marchant and Higgins 1993; Schmechel and Paterson 2005), which lies 534 mi (860 km) east of mainland New Zealand. The Chatham Island group comprises two large, inhabited islands (Chatham and Pitt) and numerous smaller islands. Two of the smaller islands (Rangatira (also referred to as South East) and Mangere) are nature reserves, which provide important habitat for the Chatham oystercatcher. The Chatham Island group has a biota (i.e., plants and animals in a particular area) quite different from the mainland. The remote marine setting, distinct climate, and physical makeup have led to a high degree of endemism (i.e., the occurrence of species in a limited area) (Aikman et al. 2001). The southern part of the oystercatcher's range is dominated by rocky habitats with extensive rocky platforms. The northern part of the range is a mix of sandy beach and rock platforms (Aikman et al. 2001).

Pairs of oystercatchers occupy their territory all year, while juveniles and subadults form small flocks or occur alone on a vacant section of the coast. The nest is a scrape usually on a sandy beach just above spring-tide level or among rocks above the shoreline. On offshore islands, nests are usually well away from the territories of brown skua (*Catharacta antarctica lonnbergi*) and are often under the cover of small bushes or rock overhangs (Heather and Robertson 1997).

This species is classified as ''Endangered'' on the IUCN Red List, because it has an extremely small population (BirdLife International 2006). It is listed as ''Critically Endangered'' by the NZDOC (2008a), making it a high priority for conservation management (NZDOC 2007). In the early 1970s the population was approximately 50 birds (del Hoyo 1996). In 1988, based on past productivity information, it was feared that the species was at risk of extinction within 50–70 years (Davis 1988, as cited in Schmechel and Paterson 2005). However, the population increased by 30 percent overall between 1987 and 1999, except trends varied in different areas—increasing (northern Chatham Island, eastern Pitt Island), stable (Mangere Island), or decreasing (south Chatham Island, Rangatira) (Moore et al. 2001). A survey during the summer of 1987–88 recorded 100 to 110 birds (Marchant and Higgins 1993). A census conducted in 1998 revealed 142 birds, with 34 to 41 breeding pairs (Schmechel and O'Connor 1999). A survey undertaken in the breeding season 1999–2000 counted 125 to 126 birds, with 50 pairs (at least 40 breeding pairs). By 2004, the oystercatcher population included 88 breeding pairs and 311 birds, more than double the number of birds counted in 1998, when the intensive management program began (NZDOC 2008c). Although the population has significantly increased over the last 20 years, the population on Rangatira, an island free of mammalian predators, has gradually declined since the 1970s. The reason for the decline is unknown (Schmechel and O'Connor 1999), but population sizes can fluctuate even on islands free from predators (BirdLife International 2006).

Predation, habitat modification, natural disasters, and disturbance are factors that negatively impact the Chatham oystercatcher population (NZDOC 2001). Domestic cats (*Felis domesticus*), weka (*Gallirallus australis*), possum (*Trichosurus vulpecuta*), hedgehog (*Erinaceus eropaeus*), pigs (*Sus domestica*), black-backed gulls (*Larus dominicanus*), and harriers (*Circus approximans*) are potential predators of the Chatham oystercatcher eggs and young chicks, with cats possibly also preying on adults. Of these potential predators, cats and weka have been recorded on film predating on the species (NZDOC 2001). Rangatira and Mangere Islands are free of mammalian predators. Habitat modification by coastal vegetation—marram (European beachgrass) (*Ammophila arenaria*)—appears to have adversely affected oystercatcher breeding in northern locations on Chatham Island. At sites where marram has become established, the beach profile becomes steeper and the dune face moves closer to the high-water mark. Since oystercatchers prefer to nest in more open areas, the occurrence of marram appears to have forced the oystercatchers to nest further down the beach, where the spring tides or storm surges are more likely to destroy nests. The vegetation also creates a relatively dense cover that can conceal predators. During nesting, Chatham oystercatchers are sensitive to disturbance by people, farm stock, and dogs. Also, vehicles run over nests, and domestic sheep and cattle, which regularly use the beaches in northern Chatham Island, trample nests (NZDOC 2001).

The birds of the Chatham Island group are protected due to human intervention and management. The NZDOC focused conservation efforts in the early 1990s on predator trapping and fencing to limit domestic stock access to nesting areas. Some nests were moved away from the high tide mark, and nest manipulation may have helped to increase hatching success (NZDOC 2008b). In 2001, the NZDOC published a Chatham Island oystercatcher recovery plan covering the period 2001 through 2011. Nest manipulation, fencing, signage, intensive predator control, and a research program aimed at assessing the effects of predators, flooding, and management on breeding success have been underway for several years (BirdLife International 2006).

The Chatham oystercatcher does not represent a monotypic genus. The current population has 311 individuals and the species only occurs on the small Chatham Island group. It faces threats that are moderate in magnitude because the NZDOC has taken measures to aid the recovery of the species. Threats are imminent and ongoing. Therefore, it receives a priority rank of 8.

Orange-Fronted Parakeet (*Cyanoramphus malherbi*)

The orange-fronted parakeet, also known as Malherbe's parakeet, was treated as an individual species until it was proposed to be a color morph of the yellow-crowned parakeet, *C. auriceps*, in 1974 (Holyoak 1974). Further taxonomic analysis suggested that it should once again be considered a distinct species (Kearvell et al. 2003; ITIS 2008).

At one time, the orange-fronted parakeet was scattered throughout most of New Zealand, although the two records from the North Island are thought dubious (Harrison 1970). This species has never been common (Mills and Williams 1979). During the nineteenth century, the species' distribution included South Island, Stewart Island, and a few other offshore islands of New Zealand (NZDOC 2008c). Currently, there are four known remaining populations, all located within an 18.6-mi (30-km) radius in beech (*Nothofagus* spp.) forests of upland valleys within Arthur's Pass National Park and Lake Sumner Forest Park in Canterbury, South Island (NZDOC 2008b) and two populations established on Chalky and Maud Islands (Elliott and Suggate 2007). This species inhabits southern beech forests, with a preference for locales bordering stands of mountain beech (*N. solandri*) (del Hoyo 1997; Snyder et al. 2000; Kearvell 2002). It is reliant on old mature beech trees with natural cavities or hollows for nesting. Breeding is linked with the irregular seed production by *Nothofagus*; in mast years with a high abundance of seeds, parakeet numbers can increase substantially. In addition to eating seeds, the orange-fronted parakeet feeds on fruits, leaves, flowers, buds, and invertebrates (BirdLife International 2000).

The orange-fronted parakeet has an extremely small population and limited range. The species is listed as ''Critically Endangered'' on the IUCN Red List, ''because it underwent a population crash following rat invasions in 1990–2000, and it now has a tiny, severely fragmented, and declining population'' (BirdLife International 2006). It is listed in Appendix II of CITES (CITES 2008). The NZDOC (2008c) considers the orange-fronted parakeet, or kekeriki, to be the rarest parakeet in New Zealand. Because it is classified as ''Nationally Critical'' with a high risk of extinction, the NZDOC has been working intensively with the species to ensure its survival. The population is estimated at 100 to 200 individuals in the wild and declining (NZDOC 2008c).

There are several reasons for the species' continuing decline; one of the most prominent risks to the species is believed to be predation by introduced species, such as stoats (*Mustela erminea*) and rats (*Rattus* spp.) (BirdLife International 2007a). Large numbers of stoats and rats in beech forests cause large losses of parakeets. Stoats and rats are excellent hunters on the ground and in trees. When they exploit parakeet nests and roosts in tree holes, they particularly impact females, chicks, and eggs (NZDOC 2008d). The NZDOC introduced ''Operation ARK,'' an initiative to respond to predator problems in beech forests to prevent species' extinctions, including orange-fronted parakeets. Predators are methodically controlled with traps, toxins in bait stations, bait bags, and aerial spraying, when necessary (NZDOC 2008e). Despite these controls, predation by introduced species is still a threat because they have not been eradicated from this species' range.

Habitat loss and degradation are also considered threats to the orange-fronted parakeet (BirdLife International 2007b). Large areas of native forest have been felled or burnt, decreasing the habitat available for parakeets (NZDOC 2008d). Silviculture of beech forests aims to harvest trees at an age when few will become mature enough to develop suitable cavities for orange-fronted parakeets (Kearvell 2002). The habitat is also degraded by brush-tailed possum (*Trichosurus vulpecula*), cattle, and deer browsing on plants and changing the forest structure (NZDOC 2008d). This is a problem for the orange-fronted parakeet which uses ground and low growing shrubs while feeding (Kearvell et al. 2002).

Snyder et al. (2000) reported that hybridization with yellow-crowned parakeets had been observed at Lake Sumner. Other risks include increased competition between the orange-fronted parakeet and the yellow-crowned parakeet in a habitat substantially modified by humans, competition with introduced finch species, and competition with introduced wasps (*Vespula vulgaris* and *V. germanica*) for invertebrates as a dietary source (Kearvell et al. 2002).

The NZDOC closely monitors all known populations of the orange-fronted parakeet. Nest searches are conducted, nest holes are inspected, and surveys are carried out in other areas to look for evidence of other populations. In fact, the surveys successfully located another orange-fronted parakeet population in May 2003 (NZDOC 2008e). A new population was established in 2006 on the predator-free Chalky Island. Eggs were removed from nests in the wild and foster parakeet parents incubated the eggs and cared for the hatchlings until they fledged and were transferred to the island. Monitoring later in the year (2006) indicated that the birds had successfully nested and reared chicks. Additional birds will be added to the Chalky Island population, in an effort to increase the genetic diversity of the population (NZDOC 2008e). A second self-sustaining population has been established on Maud Island (NZDOC 2008a).

The orange-fronted parakeet does not represent a monotypic genus. The current wild population ranges between 100 and 200 individuals, and the species' distribution is extremely limited. It faces threats that are moderate in magnitude because the NZDOC has taken important measures to aid in the recovery of the species. The NZDOC implemented a successful captive-breeding program for the orange-fronted parakeet. Using captive-bred birds from the program, NZDOC established two self-sustaining populations of the orange-fronted parakeet on predator-free islands. The NZDOC monitors wild nest sites and is constantly looking for new nests and new populations, as evidenced by the 2003 discovery of a new population. Finally, the NZDOC determined that the species' largest threat is predation and initiated a successful program to remove predators. The threats of competition for food and highly altered habitat are imminent and ongoing. Therefore, this species is assigned a priority rank of 8 (Note: the priority rank was mistakenly listed as 4 in the 2007 Notice of Review; a species that has imminent threats of moderate to low magnitude is assigned a priority ranking of 8, as per the Service's 1983 Listing Priority Guidance (48 FR 43098)).

Uvea Parakeet (*Eunymphicus uvaeensis*)

This species, previously known as *Eunymphicus cornutus*, is currently treated as two species, *E. cornutus* and *E. uvaeensis* (BirdLife International 2007a). The Uvea parakeet is found only on the small island of Uvea in the Loyalty Archipelago, New Caledonia (Territory of France); the island is only 42 mi² (110 km²) (Juniper and Parr 1998). The Uvea parakeet is found primarily in old-growth forests, notably, those dominated by *Agathis australis* pines (del Hoyo et al. 1997). Most birds occur in about 7.7 mi² (20 km²) of forest in the north, although some individuals are found in strips of forest on the northwest isthmus and in the southern part of the island, with a total area of potential habitat of approximately 25.5 mi² (66 km²) (BirdLife International 2007a; CITES 2000b). The Uvea parakeet feeds on the berries of vines and the flowers and seeds of native trees and shrubs (del Hoyo et al. 1997). It also feeds on crops in adjacent cultivated land, and the greatest number of birds occurs close to gardens with papayas, which they utilize as food (BirdLife International 2007a). The species nests in cavities of native trees, and has a clutch size of 2 to 3 eggs with some double clutches (Robinet and Salas 1999).

Early population estimates were alarmingly low—70 to 90 birds and declining (Hahn 1993). Surveys by Robinet et al. (1996) in 1993 yielded estimates of approximately 600 birds. In 1999, it was believed that 742 individuals lived in northern Uvea, with 82 birds living in the south (Primot 1999, as cited in BirdLife International 2007a).

The species is listed as ''Endangered'' in the IUCN Red List, because it occupies a very small, declining area of forest on one small island (BirdLife International 2004). The species was uplisted from Appendix II to Appendix I of CITES in July 2000, due to its small population size, restricted area of distribution, loss of suitable habitat, and unsustainable trade of the species (CITES 2000b).

Identified risks to the Uvea parakeet include habitat loss, capture of juveniles for the pet trade, and predation (BirdLife International 2007b). The forest habitat of the Uvea parakeet is threatened by clearance for agriculture and logging. In 30 years, approximately 30 to 50 percent of primary forest has been destroyed (Robinet et al. 1996). The island has a young and increasing human population of almost 4,000 inhabitants. The increase in population will most probably lead to more destruction of forest for housing, cultivated fields, and plantations, especially coconut palms, the island's main source of income (CITES 2000a). The species is also put at risk by the illegal pet trade, mainly for the domestic market (BirdLife International 2007a). Nesting holes are cut open to extract nestlings, rendering the holes unsuitable for future nesting. The increasing lack of nesting sites is believed to be a limiting factor for the species (BirdLife International 2007a). Also, Robinet et al. (1996) suggested that although the impact of capture of juveniles on the viability of populations is not obvious with long-lived species that are capable of re-nesting, such as the Uvea parakeet, the current capture of 30 to 50 young Uvea parakeets each year by humans for pets may be unsustainable. In a study of the reproductive biology of the Uvea parakeet, Robinet and Salas (1999) found that the main causes of chick death were starvation of the third chick during the first week, raptor (presumably the native brown goshawk (*Accipiter fasciatus*)) predation of fledglings, and human harvest for the pet trade.

Although the Uvea parakeet has a number of predators, the absence of the ship rat (*Rattus rattus*) and Norwegian rat (*R. norvegicus*) on Uvea is a major factor contributing to its survival. There is concern that these rats may be introduced in the future (CITES 2000b). Introductions of Uvea parakeets to the adjacent island of Lifou (to establish a second population) in 1925 and 1963 failed (BirdLife International 2007a), possibly due to the presence of ship rats and Norwegian rats (Robinet in litt. 1997, as cited in Snyder et al. 2000). Robinet et al. (1998) studied the impact of rats in Uvea and Lifou on the Uvea parakeet. They concluded that Lifou is not a suitable place for translocating Uvea parakeets unless active habitat management is carried out to protect it from ship rats. They also suggested that it would be valuable to apply low intensity rat control of the Pacific rat (R. exulans) in Uvea immediately before the parakeet breeding season.

A recovery plan for the Uvea parakeet was prepared for the period 1997–2002, which included strong local participation in population and habitat monitoring (Robinet in litt. 1997, as cited in Snyder et al. 2000). The species has recently increased in popularity and is celebrated as an island emblem (Robinet and Salas 1997; Primot in litt. 1999, as cited in BirdLife International 2007a). Conservation actions, including in-situ management (habitat protection and restoration), recovery efforts (providing nest boxes and food), and public education on the protection of the parakeet and its habitat, are underway (Robinet et al. 1996). Increased awareness of the plight of the species and improvements in law enforcement capability are helping to address illegal trade of the species. In 1998, a captive-breeding program was initiated to restock the southern portion of Uvea. Measures are now being taken to control predators and prevent further colonization by rats (BirdLife International 2007a). Current Uvea parakeet numbers are increasing, but any relaxation of conservation efforts or introduction of nonnative rats or other predators could lead to a rapid decline of the species (BirdLife International 2007a).

The Uvea parakeet does not represent a monotypic genus. It faces threats that are moderate in magnitude because important management efforts have been put in place to aid in the recovery of the species. However, all of these efforts must continue to function, because this species is an island endemic with restricted habitat in one location. Threats to the species are imminent because illegal trade still occurs and the removal of 30 to 50 percent of the old-growth forest, which the birds are dependent upon for nesting holes, negatively impacts the reproductive requirements of the species. We assign this species a priority rank of 8.

Blue-Throated Macaw (*Ara glaucogularis*)

The blue-throated macaw is endemic to forest islands in the seasonally flooded Beni Lowlands (Lanos de Mojos) of Central Bolivia (Jordan and Munn 1993; Yamashita and de Barros 1997). It inhabits a mosaic of seasonally inundated savanna, palm groves, forest islands, and humid lowlands. This species is found in areas where palm-fruit food is available, especially *Attalea phalerata* (Jordan and Munn 1993; Yamashita and de Barros 1997). It inhabits elevations between 656 and 984 ft (200 and 300 m) (BirdLife International 2008c; Brace et al. 1995; Yamashita and de Barros 1997). These macaws are not found to congregate in large flocks; but are seen most commonly traveling in pairs, and on rare occasions may be found in small flocks (Collar et al. 1992). The blue-throated macaw nests between November and March in large tree cavities where one to two young are raised (BirdLife International 2000).

The taxonomic status of this species was long disputed, primarily because the species was unknown in the wild to biologists until 1992. Previously it was considered an aberrant form of the blue-and-yellow macaw (*A. ararauna*), but the two species are now known to occur sympatrically without interbreeding (del Hoyo et al. 1997). BirdLife International (2008c) estimated there are between 50 and 249 mature individuals in the wild, and the population has some fragmentation and is decreasing.

This species was historically at risk from trapping for the national and international cage-bird trade, and some illegal trade may still be occurring. Between the early 1980s and early 1990s, approximately 400 to 1,200 birds were exported from Bolivia, and many are now in captivity in the European Union and in North America (World Parrot Trust 2003). In 1984, Bolivia outlawed the export of live parrots (Brace et al. 1995). However, in 1993 (Jordan and Munn 1993) it was reported that an Argentinian bird dealer was offering illegal Bolivian dealers a high price for blue-throated macaws. Armonia Association (BirdLife in Bolivia) monitored the wild birds that passed through a pet market in Santa Cruz from August 2004 to July 2005. Although nearly 7,300 parrots were recorded in trade, the blue-throated macaw was absent in the market during the monitoring period, which may point to the effectiveness of the ongoing conservation programs in Bolivia (BirdLife International 2007). There are a number of blue-throated macaws in captivity, with over 1,000 registered in the North American studbook. Because these birds are not too difficult to breed, the supply of captive-bred birds has increased (Waugh 2007), helping to alleviate pressure on illegal collecting of wild birds, but not completely eliminating illegal collection.

The blue-throated macaw is also at risk from habitat loss and possible competition from other birds, such as other macaws, toucans, and large woodpeckers (BirdLife International 2008b; World Parrot Trust 2008). All known sites of the blue-throated macaw are on private cattle ranches, where local ranchers typically burn the pasture annually (del Hoyo 1997). This results in almost no recruitment of palm trees, which are central to the ecological needs of the blue-throated macaw (Yamashita and de Barros (1977)). In addition, in Beni many palms are cut down by the local people for firewood (Brace et al. 1995). Thus, although the palm groves are more than 500 years old, Yamashita and de Barros (1977) concluded that the palm population structure suggests long-term decline.

This species is categorized as ''Critically Endangered'' on the IUCN Red List, ''because its population is extremely small and each isolated subpopulation is probably tiny and declining as a result of illegal trade'' (BirdLife International 2004). It is listed in Appendix I of CITES (CITES 2006) and is legally protected in Bolivia (Juniper and Parr 1998). The Eco Bolivia Foundation patrols existing macaw habitat by foot and motorbike, and the Armonia Association is searching the Beni lowlands for more populations (Snyder et al. 2000). Additionally, the Armonia Association is building an awareness campaign aimed at the cattlemen's association to ensure that the protection and conservation of these birds is at a local level (e.g., protection of macaws from trappers and the sensible management of key habitats, such as palm groves and forest islands, on their property) (BirdLife International 2008a; Llampa 2007; Snyder et al. 2000).

The blue-throated macaw does not represent a monotypic genus. It faces threats that are moderate in magnitude because wild birds are no longer taken for the legal wild-bird trade as a result of the species' CITES listing, and it is also legally protected in Bolivia. Wildlife managers in Bolivia are actively protecting the species and searching for additional populations. Threats to the species are imminent and ongoing because hunters still trap the birds for the illegal bird trade and annual burning on private ranches continues. Therefore, we assigned this species a priority rank of 8.

Helmeted Woodpecker (*Dryocopus galeatus*)

The helmeted woodpecker is endemic to the southern Atlantic forest region of southeastern Brazil, eastern Paraguay, and northeastern Argentina (BirdLife International 2007). It is found in tall lowland and montane primary forest, in forest that has been selectively logged, and generally near large tracts of intact forest (BirdLife International 2007). This woodpecker feeds on beetle larvae which live beneath tree bark. The species forages primarily in the middle canopy of the forest interior (del Hoyo et al. 2002).

Recent field work on the helmeted woodpecker revealed that the species is less rare than once thought (BirdLife International 2007). It is listed as ''Vulnerable'' by the IUCN (BirdLife International 2007). The current population is estimated at between 10,000 and 19,999 individuals and decreasing (BirdLife International 2000). This estimate has a wide range, because the species is almost certainly underreported due to the difficulty of locating birds except when vocalizing, and since they are silent for much of the year. Numerous sightings since the mid-1980s include a pair in the Brazilian State of Santa Catarina in 1998, where the species had not been seen since

1946 (del Hoyo et al. 2002). Research is needed to clarify the species' current distribution and status (del Hoyo et al. 2002).

The greatest threat to the species is widespread deforestation, and the species is not common at any known site (BirdLife International 2007; Cockle 2008). In the Atlantic forest, more than 90% of the forest has been replaced by crops and pastures, and nearly all remaining forest has been subject to selective logging of large trees, with potentially severe consequences for cavity nesting birds such as woodpeckers; selectively logged forest contains significantly fewer nesting cavities than primary forest (Cockle 2008).

The helmeted woodpecker is protected by Brazilian law and populations occur in numerous protected areas throughout its range (BirdLife International 2007). These protections prohibit the following activities with regard to this species: export and international trade, collection and research, captive propagation, and also provide measures which help to protect remaining suitable habitat, such as prohibition of exploitation of the remaining primary forests within the Atlantic forest biome and management of various practices in primary and secondary forests, such as logging, charcoal production, reforestation, recreation, and water resources (ECOLEX 2007). However, for various reasons (*e.g.*, lack of funding, personnel, or local management commitment), Brazil's current capacity to achieve its stated natural resource objectives in protected areas is limited (ADEJA 2007; Bruner et al. 2001; Costa 2007; IUCN 1999; Neotropical News 1996; Neotropical News 1999). Therefore, it is likely that not all of the habitat protections for the helmeted woodpecker would be sufficiently addressed at these sites. The helmeted woodpecker does not represent a monotypic genus. The magnitude of threat to the species is moderate because the population is much larger than previously thought; however, the threat is imminent because the forest habitat, in particular, the availability of nesting cavities upon which the species depends, is being reduced by human activities. It therefore, receives a priority rank of 8.

Okinawa Woodpecker (*Dendrocopos noguchii*, previously known as *Sapheopipo noguchii*)

The Okinawa woodpecker lives in the northern hills of Okinawa Island, Japan. Okinawa is the largest island of the Ryukyus Islands, a small island chain located between Japan and Taiwan (Brazil, 1991; Stattersfield et al. 1998; Winkler et al. 2005). This species is confined to Kunigami-gun, or Yambaru, with its main breeding areas located along the mountain ridges between Mt. Nishime-take and Mt. Iyu-take, although it also nests in well-forested coastal areas (Research Center, Wild Bird Society of Japan 1993, as cited in BirdLife International 2001). It prefers undisturbed, mature, subtropical evergreen broadleaf forests, with tall trees greater than 7.9 in (20 cm) in diameter (del Hoyo 2002; Short 1982). Trees of this size are generally more than 30 years old and are confined to hilltops (Brazil 1991). Places with conifers appear to be avoided (Short 1973; Winkler et al. 1995). The Okinawa woodpecker has been sighted just south of Tanodake in an area of entirely secondary forest that was too young for nest building, but Brazil (1991) thought this may have involved birds displaced by the clearing of mature forests. The Okinawa woodpecker feeds on large arthropods, notably beetle larvae, spiders, moths, and centipedes, fruit, berries, seeds, acorns, and other nuts (del Hoyo 2002; Short 1982; Winkler et al. 2005). They forage in old-growth forests with large, often moribund trees, accumulated fallen trees, rotting stumps, debris, and undergrowth (Brazil 1991; Short 1973). This woodpecker nests in holes excavated in large old trees, often a hollow in *Castanopsis cuspidata* trees (del Hoyo 2002; Short 1982).

Until recently the Okinawa woodpecker was considered to belong to the monotypic genus *Sapheopipo*. This view was based on similarities in color patterns, external morphology, and foraging behavior. Winkler et al. (2005) analyzed partial nucleotide sequences of mitochondrial genes and concluded that this woodpecker belongs in the genus *Dendrocopos*. Given the other species in this genus, the Okinawa woodpecker is no longer considered to belong to a monotypic genus.

The Okinawa woodpecker is considered one of the world's rarest extant woodpecker species (Winkler et al. 2005). The elimination of forests by logging and the cutting and gathering of wood for firewood are the main causes of its small and lessening numbers (Short 1982), but the greatest danger to this woodpecker is the fragmentation of its population into scattered tiny colonies and isolated pairs (Short 1973). The species is categorized on the IUCN Red List as ''Critically Endangered,'' because it is comprised of a single diminutive, declining population, which is put at risk by the continued loss of old-growth and mature forest to logging, dam construction, agricultural clearing, and golf course construction. Its limited range and tiny population make it vulnerable to extinction from disease and natural disasters such as typhoons (BirdLife International 2004). During the 1930s, the Okinawa woodpecker was considered nearly extinct. By the early 1990s, the breeding population was estimated to be about 75 birds (BirdLife International 2008a). The current population estimate ranges between 146 and 584 individuals, with a projected future 10-year decline of 30 to 49 percent (BirdLife International 2008b). The species is legally protected in Japan and occurs in small protected areas on Mt. Ibu and Mt. Nishime (BirdLife International 2008a). The Yambaru, a forest area in the Okinawa Prefecture, was designated as a national park in 1996, and conservation organizations have purchased sites where the woodpecker occurs to establish private wildlife preserves (del Hoyo et al. 2002).

The Okinawa woodpecker faces threats that are moderate in magnitude because the species is legally protected in Japan and its range occurs in several protected areas. However, the threats to the species are imminent because the old-growth habitat, upon which the species is dependent, continues to be removed, and preferable habitat continues to be altered for agriculture and golf courses. It therefore receives a priority rank of 8 (Note: The priority number was changed from 7 to 8 because of the recent research showing that the Okinawa woodpecker belongs to a different genus and is no longer considered a monotypic species).

Yellow-Browed Toucanet (*Aulacorhynchus huallagae*)

The yellow-browed toucanet is known from only two localities in north-central Peru—La Libertad, where it is uncommon, and Rio Abiseo National Park, San Martin, where it is very rare (BirdLife International 2008; del Hoyo et al. 2002; Wege and Long 1995). Its estimated range is only 174 mi^2 (450 km^2) (BirdLife International 2008). There have been recent reports of the species from Leymebambe (T. Mark in litt. 2003, as cited in BirdLife International 2008). It inhabits a narrow altitudinal range between 6,970 and 8,232 ft (2,125 and 2,510 m), preferring the canopy of humid, ephiphyte-laden montane cloud forests, particularly areas that support *Clusia* trees (del Hoyo et al. 2002; Fjeldså and Krabbe 1990; Schulenberg and Parker 1997). This narrow distributional band may be related to the occurrence of the larger grey-breasted mountain toucan (*Andigena hypoglauca*) above 7,544 ft (2,300 m) and to the occurrence of the emerald toucanet (*Aulacorhynchus prasinus*) below 6,888 ft (2,100 m) (Schulenberg and Parker 1997). The species' restricted range remains unexplained, and recent information indicates that both of the suggested competitors have wider altitudinal ranges which completely encompass the range of the yellow-browed toucanet (Clements and Shany 2001, as cited in BirdLife International 2008; Collar et al. 1992; del Hoyo et al. 2002; J. Hornbuckle in litt. 1999, as cited in BirdLife International 2008). The yellow-browed toucanet does not appear to occupy all potentially suitable forest available within its range (Schulenberg and Parker 1997). Although it occurs within the large Rio Abiseo National Park, the population in the reserve is thought to be small (BirdLife International 2004; del Hoyo 2002).

Deforestation has been widespread in this region, but has largely occurred below the toucanet's altitudinal range (BirdLife International 2008; Barnes et al. 1995). However, coca growers have taken over forests within its altitudinal range, probably resulting in some reductions in the species' range and population (BirdLife International 2004; Plenge in litt. 1993, as cited in BirdLife International 2008). Nevertheless, much forest remains within the range of the yellow-browed toucanet, and most of the area is only lightly settled by humans; the limited range of this species is not well explained relative to the threats reported (BirdLife International 2008; Schulenberg and Parker 1997).

It is listed as ''Endangered'' on the IUCN Red List, because of its very small range and extant population records from only two locations (BirdLife International 2004). The current population size is unknown, but the population trend is believed to be decreasing (BirdLife International 2008).

The yellow-browed toucanet does not represent a monotypic genus. The magnitude of threat to the species is moderate, since habitat loss is largely recorded outside its range, and non-imminent due to the uncertainty of ongoing habitat loss from cocoa growers. Therefore, it receives a priority rank of 11.

Brasilia Tapaculo (*Scytalopus novacapitalis*)

The Brasilia tapaculo is found in swampy gallery forest, disturbed areas of thick streamside vegetation, and dense secondary growth of the bracken fern *Pteridium aquilinum*, from Goiás, the Federal District, and Minas Gerais, Brazil (Negret and Cavalcanti 1985, as cited in Collar et al. 1992; Collar et al. 1992; BirdLife International 2007). The Brasilia Tapaculo will occasionally colonize disturbed areas near streams (BirdLife International 2003). This species has only been recorded locally within Formas in Goiás, around Brasília. Particular sites where the species has been located, at low densities, include Serra Negra (on the upper Dourados River) and the headwaters of the São Francisco, both in Minas Gerais; and Serra do Cipó and Caraça in the hills and tablelands of central Brazil (BirdLife International 2003).

Although the species was once considered rare (Sick and Texeira 1979, as cited in Collar et al. 1992), it is now found in reasonable numbers in certain areas of Brasilia (D. M. Teixeira, in litt. 1987, as cited in Collar et al. 1992). The population is estimated at more than 10,000 birds, with a decreasing population trend (BirdLife International 2007). The IUCN categorizes *Scytalopus novacapitalis* as ''Near Threatened'' (BirdLife International 2007). The species occupies a very limited range and is presumably losing habitat around Brasilia. However, its distribution now appears larger than initially believed, and the swampy gallery forests where it is found are not conducive for forest clearing, leaving the species' habitat less vulnerable to this threat than previously thought. However, dam building for irrigation on rivers which normally flood gallery forests is an emerging threat (Antas 2007; D. M. Teixeira in litt. 1987, as cited in Collar et al. 1992). The majority of locations of this species lie within established reserves, and both fire risk and drainage impacts are reduced in these areas (Antas 2007). The Brasilia tapaculo is currently protected by Brazilian law (Bernardes et al. 1990, as cited in Collar et al. 1992), and it is found in six protected areas (Machado et al. 1998, Wege and Long 1995; as cited in BirdLife International 2007). Annual burning of adjacent grasslands limits the extent and availability of suitable habitat, as does wetland drainage and the sequestration of water for irrigation (Machado et al. 1998, as cited in BirdLife International 2007).

The Brasilia tapaculo does not represent a monotypic genus. The magnitude of threat to the species is moderate because the population is much larger than previously believed and preferred habitat is swampy and difficult to clear. Threats are imminent, however, because habitat is being drained or dammed for agricultural irrigation, and grassland burning limits the extent of suitable habitat. Therefore, it receives a priority rank of 8.

Codfish Island Fernbird (*Bowdleria punctata wilsoni*)

The Codfish Island fernbird is found only on Codfish Island—a Nature Reserve of 3,448 ac (1,396 ha)—located 1.8 mi (3 km) off the northwest coast of Stewart Island, New Zealand (IUCN 1979; McClelland 2007). There are five subspecies of fernbirds, each restricted to a single island and its outlying islands. The North and South Islands' subspecies are widespread and locally common. The Stewart Island and Snares' subspecies are moderately abundant (Heather and Robertson, 1997). In 1966, the status of the Codfish Island subspecies was considered relatively safe (Blackburn 1967), but estimates dating from 1975 indicated a gradually declining population numbering approximately 100 individuals (Bell 1975, as cited in IUCN 1979). McClelland (2007) wrote that in the past the subspecies was restricted to low shrubland on the top of Codfish Island with a few individuals around the coastal shrubland; the birds are thought to have been eliminated from forest habitat by the Polynesian rat (*Rattus exulans*) (McClelland 2007). The IUCN (1979) concluded that the subspecies' absence from areas of Codfish Island that it had formerly occupied in the mid-1970s evidenced a decline.

Fernbirds are sedentary, and their flight is weak. They are secretive and reluctant to leave cover. They feed in low vegetation or on the ground, eating mainly caterpillars, spiders, grubs, beetles, flies, and moths (Heather and Robertson, 1997).

Codfish Island's native vegetation has been modified by the introduced herbivore, the Australian brush-tailed possum (*Trichosurus vulpecula*). Fernbird populations have also been reduced due to predation by weka (*Gallirallus australis scotti*) and Polynesian rats (Merton 1974, pers. comm., as cited in IUCN 1979). Several conservation measures have been undertaken by the New Zealand DOC. The weka and possum were eradicated from Codfish Island in 1984 and 1987, respectively (McClelland 2007). The Polynesian rat was eradicated in 1997 (Conservation News 2002; McClelland 2007). The Codfish Island fernbird population is rebounding strongly with the removal of invasive predator species. The fernbird invaded the forest habitat, which greatly expanded the species' available habitat. Although there is no accurate estimate on the current size of the population (estimates are based on incidental encounter rates in the various habitat types on the island), the current population is believed to be several hundred. Thus, McClelland (2007) concluded that it is likely that the population has peaked and is now stable.

To safeguard the Codfish Island fernbird, the NZDOC established a second population on Putauhinu Island—a small (356-ac (144-ha)), privately owned island located approximately 25 mi (40 km) south of Codfish Island. The Putauhinu population established rapidly, and McClelland (2007) reported it is believed to be stable. While there are no accurate data on the population size or trends, the population is estimated to be 200 to 300 birds spread over the island (McClelland 2007).

The Codfish Island fernbird is a subspecies that is now facing threats that are low to moderate in magnitude because the removal of invasive predator species and the establishment of a second population have allowed for a strong rebound in the subspecies' population. Threats are non-imminent because conservation measures have eradicated nonnative predatory species from Codfish Island. However, even though efforts to remove nonnative predators have been successful, there is a continued risk that predators will be re-introduced to the island by boats transporting conservation and research staff to the islands. Given continued low numbers, with two populations in the low hundreds, we find that introduced predators remain a threat to this subspecies, though non-imminent.

The subspecies, therefore, receives a priority rank of 12 (Note: the priority rank was mistakenly listed as 9 in the 2007 Notice of Review; a subspecies that has non-imminent threats of moderate to low magnitude is assigned a priority ranking of 12, as per the Service's 1983 Listing Priority Guidance (48 FR 43098)).

Ghizo White-Eye (*Zosterops luteirostris*)

The Ghizo white-eye is endemic to Ghizo, a very densely populated island in the Solomon Islands in the South Pacific (BirdLife International 2007a). Birds are locally common in the remaining tall or old-growth forest, which is very fragmented and comprises less than 0.39 mi^2 (1 km^2). It is less common in scrub close to large trees and in plantations (Buckingham et al. 1995 and Gibbs 1996, as cited in BirdLife International 2007a), and it is not known whether these two habitats can support sustainable breeding populations (Buckingham et al. 1995, as cited in BirdLife International 2007a). The IUCN Red List classifies this species as ''Endangered,'' because of its very small population that is considered to be declining due to habitat loss. It further notes that the species would be classified as ''Critically Endangered'' if the species' range was judged to be severely fragmented (BirdLife International 2007c). The population estimate for this species is 250 to 999 birds. While there are no data on population trends, the species is suspected to be declining due to habitat degradation (BirdLife International 2007b). The very tall old-growth forest on Ghizo is still under some threat from clearance for local use as timber, firewood, and gardens, and the areas of other secondary growth, which are suboptimal habitats for this species, are under considerable threat from clearance for agricultural land (BirdLife International 2007a).

The Ghizo white-eye does not represent a monotypic genus. It faces threats that are moderate in magnitude because forest clearing, while a concern, does not appear to be proceeding at a pace to rapidly denude the habitat. Threats are imminent because the old-growth forest which the species is dependent upon is still being cleared for local use, and secondary growth is being converted for agricultural purposes. Therefore, we assign the species a priority rank of 8.

Black-Backed Tanager (*Tangara peruviana*)

The black-backed tanager is endemic to the coastal Atlantic forest region of southeastern Brazil, with records from Rio de Janeiro, Sao Paolo, Parana, Santa Catarina, Rio Grande do Sul, and Espirito Santo (Argel-de-Oliveira in litt. 2000, as cited in BirdLife International 2006). It is largely restricted to coastal sand-plain forest and littoral scrub, or restinga, and has also been located in secondary forests (BirdLife International 2007). The black-backed tanager is generally not considered rare within suitable habitat (BirdLife International 2007). It has a complex distribution with periodic local fluctuations in numbers owing to seasonal movements, at least in Rio de Janeiro and Sao Paolo (BirdLife International 2007). Clarification of the species' seasonal movements will provide an improved understanding of the species' population status and distribution (BirdLife International 2007). Population estimates range from 2,500 to 10,000 individuals (BirdLife International 2007), and it is considered ''Vulnerable'' by the IUCN (BirdLife International 2007). The species is negatively impacted by the rapid and widespread loss of habitat for beachfront development and occasionally appears in the illegal cage-bird trade (BirdLife International 2006).

The black-backed tanager does not represent a monotypic genus. The threat to the species is low to moderate in magnitude due to the species' fairly large population size and range. The threat is, however, imminent because the species is put at risk by ongoing rapid and widespread loss of habitat due to beachfront development. Therefore, we give this species a priority rank of 8 (Note: the priority rank was mistakenly listed as 9 in the 2007 Notice of Review; a species that has imminent threats of moderate to low magnitude is assigned a priority ranking of 8, as per the Service's 1983 Listing Priority Guidance (48 FR 43098)).

Lord Howe Pied Currawong (*Strepera graculina crissalis*)

The Lord Howe pied currawong is a separate subspecies from the five Australian mainland pied currawongs. It is endemic to the Lord Howe Island, New South Wales, Australia. The highly mobile birds can be found anywhere on the 7.7-mi^2 (20-km^2) island (Hutton 1991), as well as on offshore islands such as the Admiralty group (Garnett and Crowley 2000). The Lord Howe pied currawong breeds in rainforests and palm forests, particularly along streams. Their territories include sections of streams or gullies that are lined by tall timber (Garnett and Crowley 2000). The highest densities of nests are located on the slopes of Mt. Gower and in the Erskine Valley, with smaller numbers on the lower land to the north (Knight 1987, as cited in Garnett and Crowley 2000). The nest is placed high in a tree and is made of a cup of sticks lined with grass and palm thatch (Department of Environment & Climate Change (DECC) 2005). Most of the island is still forested, and the removal of introduced feral animals has resulted in the recovery of the forest understory (World Wildlife Fund (WWF) 2001).

The Lord Howe pied currawong is omnivorous and eats a wide variety of food, including native fruits and seeds (Hutton 1991), and is the only remaining native island vertebrate predator (DECC 2005). It has been recorded taking seabird chicks, poultry, and chicks of the Lord Howe woodhen (*Tricholimnas sylvestris*) and white tern (*Gygis alba*). Currawongs also feed on dead rats and have been observed to catch live rats and eat them (Hutton 1991). A Department of Environmental Conservation (DEC) scientist observed that food brought to nestlings was, in decreasing order, invertebrates, fruits, reptiles, and nestlings of other bird species (Lord Howe Island Board (LHIB) 2006).

The Lord Howe pied currawong is listed as ''Vulnerable'' under the New South Wales Threatened Species Conservation Act of 1995, because it has a limited range, only occurring on Lord Howe Island (DECC 2004). It also is listed as ''Vulnerable'' under the Commonwealth Environment Protection and Biodiversity Conservation Act of 1999. These laws provide a legislative framework to protect and encourage the recovery of vulnerable species (DEC 2006a). The Lord Howe Island Act of 1953, as amended, established the Lord Howe Island Board (LHIB); made provisions for the LHIB to care for, control, and manage the island; and established 75 percent of the land area as a Permanent Park Preserve (DEC 2006a). In 1982, the island was inscribed on the World Heritage List for its outstanding natural universal values (Department of the Environment and Water Resources 2007).

In the Action Plan for Australian Birds 2000 (Garnett and Crowley 2000), the population was estimated at approximately 80 mature individuals. In 2006, initial results from a color band survey suggested that the population size was about 180 to 200 individual birds (LHIB 2006). Complete results reported by the Foundation for National Parks & Wildlife (2007) estimated the breeding population to be 80 to 100 pairs, with a nesting territory in the tall forest areas of about 12 ac (5 ha) per pair. The population size is limited by the amount of available habitat and the lack of food during the winter (Foundation for National Parks & Wildlife 2007).

The Lord Howe Island draft Biodiversity Management Plan, which was out for comment in 2006, will become the formal National and NSW Recovery Plan (Plan) for threatened species and communities of the Lord Howe Island Group (DEC 2006a). The main current threat identified for the Lord Howe Island currawong is habitat clearing and modification (DEC 2006b). Lord Howe Island is unique among inhabited Pacific Islands in that less than 10 percent of the island has been cleared (WWF 2001) and less than 24 percent has been disturbed (DEC 2006a). Although large-scale clearing of native vegetation no longer occurs on Lord Howe Island, the impact of vegetation clearing on a small scale needs to be assessed (DEC 2006a). A lesser current risk to the species, but one which may account for its historical decline and continued low numbers, is human interactions (Garnett and Crowley 2000). Prior to the 1970s, locals would shoot currawongs due to the bird's habit of preying on nestling birds (Hutton 1991), and the currawongs remain unpopular with some residents. It is unknown what effect this localized killing has on the overall population size and distribution of this species (Garnett and Crowley 2000). Also, currawongs often prey on ship (black) rats and consequently may suffer mortality from non-target poisoning during rat-baiting programs (DEC 2006b). Close monitoring of the population is needed because this small, endemic population is susceptible to the introduction of avian disease or of new predators (Garnett and Crowley 2000). There is a long history of introduction of nonnative fauna (e.g., 18 introduced land birds, and 3 mammals now resident), and the introduction to Lord Howe Island of new exotic fauna and flora (including disease), by air or ship, is considered a major ongoing threat to endemic species, including the Lord Howe pied currawong (DEC 2006a).

The Lord Howe pied currawong is a subspecies facing threats that are low in magnitude and non-imminent. Therefore, it receives a priority rank of 12.

Invertebrates

Harris' Mimic Swallowtail (*Eurytides* (syn. *Mimoides) lysithous harrisianus*)

Harris' mimic swallowtail is a subspecies endemic to Brazil (Collins and Morris 1985). Although the *species'* range includes Paraguay, the *subspecies* has not been confirmed there (Collins and Morris 1985; Finnish University and Research Network (Funet) 2004). Occupying the lowland swamps and sandy flats above the tidal margins of the coastal Atlantic Forest, the subspecies prefers alternating patches of strong sun and deep shade (Brown 1996; Collins and Morris 1985). This subspecies is polyphagous, meaning that its larvae feed on more than one plant species (Kotiaho et al. 2005). Information on preferred hostplants and adult nectar-sources was published in the 12-month finding (69 FR 70580). This subspecies mimics at least three *Parides* species, including the fluminense swallowtail; details on mimicry were provided in the 12-month finding (69 FR 70580) and in the 2007 Notice of Review (72 FR 20184). Researchers believe that this mimicry system may cause problems in distinguishing this subspecies from the species that it mimics (Brown, in litt. 2004; Monteiro et al. 2004).

Harris' mimic swallowtail was previously known in Espirito Santo and Rio de Janeiro (Collins and Morris 1985; New and Collins 1991). However, there are no recent confirmations in Espirito Santo. In Rio de Janeiro, Harris' mimic swallowtail has recently been confirmed in three localities. Two colonies are located on the east coast of Rio de Janeiro, at Barra de São João and Macaé, and the other in Poço das Antas Biological Reserve, further inland. The Barra de São João colony is the best-studied colony. Since 1984, it has maintained a stable size, varying between 50 to 250 individuals (Brown 1996; K. Brown, Jr., in litt. 2004; Collins and Morris 1985), and was reported to be viable, vigorous, and stable in 2004 (K. Brown, Jr., in litt. 2004). There are no estimates of the size of the colony in Poço das Antas Biological Reserve, where it had not been seen for 30 years prior to its rediscovery there in 1997 (K. Brown, Jr., in litt. 2004). Population estimates are lacking for the colony at Macaé, where the subspecies was netted in Jurubatiba National Park in the year 2000, after having not been seen in the area for 16 years (Monteiro et al. 2004). The Brazilian Institute of the Environment and Natural Resources (Instituto Brasileiro do a Meio Ambiente de do Recursos Naturais Renováveis; IBAMA) considers this subspecies to be critically imperiled (MMA 2003; Portaria No. 1,522 1989) and ''strictly protected,'' such that collection and trade of the subspecies are prohibited (Brown 1996). Harris' mimic swallowtail was categorized on the IUCN Red List as ''Endangered'' in the 1988, 1990, and 1994 IUCN Red Lists (IUCN 1996). However, it has not been re-evaluated using the 1997 IUCN Red List criteria, nor has it been incorporated into the 2007 IUCN Red List database (IUCN 2007).

Habitat destruction is the main threat to this subspecies (Brown 1996; Collins and Morris 1985), especially urbanization in Barra de São João, industrialization in Macaé (Jurubatiba National Park), and previous fires in the Poço das Antas Biological Reserve. As described in detail for the fluminense swallowtail (below), Atlantic forest habitat has been reduced to 5 to 10 percent of its original cover. More than 70 percent of the Brazilian population lives in the Atlantic forest, and coastal development is ongoing throughout the Atlantic forest region (Butler 2007; Conservation International 2007; Critical Ecosystem Partnership Fund (CEPF) 2007a; Höfling 2007; Hughes et al. 2006; The Nature Conservancy 2007;

Peixoto and Silva 2007; Pivello 2007; World Food Prize 2007; WWF 2007).

Both Barra de São João and the Poço das Antas Biological Reserve, two of the known Harris' mimic swallowtail localities, lie within the São João River Basin. The current conditions at Barra de São João appear to be suitable for long-term survival of this subspecies. The Barra de São João River Basin encompasses a 535,240-ac (216,605-ha) area, 372,286 ac (150,700 ha) of which is managed as protected areas. The preferred landscape of open and shady areas (Brown 1996; Collins and Morris 1985) continues to be present in the region, with approximately 541 forest patches averaging 314 ac (127 ha) in size, covering nearly 68,873 ha (170,188 ac), and a minimum distance between forest patches of 0.17 mi (276 m) (Teixeira 2007). In studies between 1984 and 1991, Brown (1996) determined that Harris' mimic swallowtails in Barra de São João flew a maximum distance of 0.62 mi (1000 m); it follows that the average flying distance would be less than this figure. Thus, the average (0.17 mi (276 m)) distance between forest patches in the Barra de São João River Basin is clearly within the flying distance of this subspecies. The colony at Barra de São João has maintained a stable population size for 20 years, indicating that the conditions available there remain suitable.

Harris' mimic swallowtail ranges within two protected areas: Poço das Antas Biological Reserve and Jurubatiba National Park. These protected areas are described in detail for the fluminense swallowtail. In summary, the Poço das Antas Biological Reserve (Reserve) was established to protect the golden lion tamarin (*Leontopithecus rosalia*) (Decree No. 73,791 1974), but the Harris' mimic swallowtail, which occupies the same range, may benefit indirectly by efforts to conserve golden lion tamarin habitat (De Roy 2002; Teixeira 2007; WWF 2003). Habitat destruction caused by fires in Poço das Antas Biological Reserve appears to have abated, and the revised management plan indicates that the Reserve will be used for research and conservation, with limited public access (CEPF 2007a; IBAMA 2005). The Jurubatiba National Park (Park) is located in a region that is undergoing continuing development pressures from urbanization and industrialization (Brown 1996; CEPF 2007b; IFC 2002; Khalip 2007; Otero and Brown 1984; Savarese 2008), and there is no management plan in place for the Park (CEPF 2007b). However, as discussed for the fluminense swallowtail, the Park is considered to be in a very good state of conservation (Rocha et al. 2007).

Harris' mimic swallowtail does not represent a monotypic genus, but it is a subspecies. Based on the above information, we have determined that habitat destruction is a threat to the subspecies. The magnitude of the threat is low because suitable habitat continues to exist for this polyphagous subspecies; the best-studied colony has maintained a stable and viable size for nearly 2 decades; an additional locality has been confirmed; the subspecies is strictly protected by Brazilian law; and two colonies are located within protected areas. While the protected areas in which this subspecies is found continue to be threatened with potential habitat destruction from urbanization and industrialization, the threat of habitat destruction is non-imminent because such destruction within those protected areas is not ongoing at this time. Therefore, the subspecies is designated a priority rank of 12.

Jamaican Kite Swallowtail (*Eurytides marcellinus*)

The Jamaican kite swallowtail is endemic to Jamaica, preferring wooded, undisturbed habitat containing the West Indian lancewood (*Oxandra lanceolata*), the only known larval hostplant for this monophagous species (Bailey 1994; Collins and Morris 1985), meaning that its larvae feed only on a single plant species (Kotiaho et al. 2005). Adult plant preferences have not been reported. Since the 1990s, adult Jamaican kite swallowtails have been observed in the Parishes of St. Thomas and St. Andrew in the east; westward in St. Ann, Trelawny, and St. Elizabeth; and in the extreme western coast Parish of Westmoreland (Bailey 1994; Harris 2002; Möhn 2002; Smith et al. 1994; WRC 2001). The species was most recently sighted in mid-2007 in the Blue and John Crow Mountains National Park (see description below), where 4 individuals were observed (Jamaica Conservation and Development Trust (JCDT) and Green Jamaica 2007a). There is only one known breeding site in the eastern coast town of Rozelle (St. Thomas Parish) (Bailey 1994; Collins and Morris 1985; Garraway et al. 1993; Smith et al. 1994). Rozelle may also be referred to in the literature as Roselle (e.g., Anderson et al. 2007). According to Dr. Robert Robbins (in litt. 2004), it is possible that other breeding sites exist given the widely dispersed nature of the larval food plant. The Jamaican kite swallowtail maintains a low population level and occasionally becomes locally abundant in Rozelle during the breeding season in early summer and occasionally again in early fall (Bailey 1994; Brown and Heineman 1972; Collins and Morris 1985; Garraway et al. 1993; Smith et al. 1994). It experiences episodic population explosions, as described in the 12-month finding (69 FR 70580) and in the 2007 Notice of Review (72 FR 20184). The species is protected under Jamaica's Wildlife Protection Act of 1998 and is included in Jamaica's National Strategy and Action Plan on Biological Diversity, which has established specific goals and priorities for the conservation of Jamaica's biological resources (Schedules of The Wildlife Protection Act 1998). Beginning in 1985, the Jamaican kite swallowtail was categorized on the IUCN Red List as ''Vulnerable;'' it has not been re-evaluated using the 1997 criteria (Gimenez Dixon 1996).

Habitat modification is the primary threat to the Jamaican kite swallowtail. Monophagous butterflies tend to be more threatened than polyphagous species, in part due to their specific habitat requirements (Kotiaho et al. 2005). West Indian lancewood, the Jamaican kite's only known larval food plant, has been cleared for cultivation and felled for the commercial timber industry (Collins and Morris 1985; Windsor Plywood 2004). Although West Indian lancewood remains widely dispersed throughout the island (R. Robbins, in litt. 2004), the harvest and clearing of West Indian lancewood habitat reduces the availability of the plant (Bailey 1994; Collins and Morris 1985).

In Rozelle, the only known breeding site for this species (Bailey 1994; Collins and Morris 1985; Garraway et al. 1993; Smith et al. 1994), there has been extensive habitat modification for agricultural and industrial purposes, such as mining (Gimenez Dixon 1996; WWF 2001). The effect of historical habitat modification negatively impacts the swallowtail today, because the Jamaican kite does not thrive in disturbed habitats (Collins and Morris 1985). Rozelle is also subject to naturally occurring, high impact stochastic events, such as regularly-occurring hurricanes, as elaborated in the 2007 Notice of Review (72 FR 20184). According to the Economic Commission for Latin America and the Caribbean (ECLAC), United Nations Development Programme (UNDP), and Planning Institute of Jamaica (PIOJ) (2004), hurricane-related weather damage in the last 2 decades along the coastal zone of Rozelle has been more intense than in previous decades, resulting in the erosion and virtual disappearance of this once-extensive recreational beach. In 1988, it was estimated that Hurricane Gilbert caused a 75 percent reduction of Rozelle Beach due to erosion (UNEP-CEP 1989). Most recently, Hurricane Ivan, a Category 5 hurricane that hit the island in 2004, caused severe local damage to Rozelle Beach, including erosion of the cliff face and shoreline (ECLAC et al. 2004). Thus, while we do not consider stochastic events to be a primary threat factor for this species, the damage caused by hurricanes that have been increasing in severity and frequency within the past two decades is an unpredictable contributor to habitat loss.

Habitat destruction occurs in western Parishes, where adult Jamaican kite swallowtails have been observed. Cockpit Country, encompassing 30,000 ha (74,131 ac) of rugged forest-karst (a specialized limestone habitat) terrain, spans four western Parishes, including Trelawny and St. Elizabeth, where adult Jamaican kite swallowtails have been observed (Gordon and Cambell 2006). Although eighty-one percent of Cockpit Country remains forested (Tole 2006), fragmentation is occurring as a result of human-induced activities. Current threats to Cockpit Country include bauxite mining, unregulated plant collecting, extensive logging, conversion of forest to agriculture, illegal drug cultivation, and expansion of human settlements. These activities contribute to degradation of the hydrology system from in-filling, siltation, accumulation of solid waste, and invasion by nonnative, invasive species (Cockpit Country Stakeholders Group and JEAN (Gordon and Cambell 2006; Jamaica Environmental Advocacy Network 2007; Tole 2006). In 2003, the Jamaican National Environment and Planning Agency identified Rozelle and Cockpit Country (which spans at least four western Parishes, including Trelawny and St. Elizabeth, where adult Jamaican kites have been observed) as priority locations to receive protected area status within the next 5 to 7 years (NEPA 2003). The status of this proposal is not included in the 2007 Environmental Action Plan Status Report (NEPA 2007).

Currently, the Blue and John Crow Mountains National Park is the only protected area in which adult Jamaican kite swallowtails have been observed, including the most recent observation in mid-2007 (Bailey 1994; JCDT 2006; JCDT and Green Jamaica 2007a). Located on the inland portions of St. Thomas and St. Andrew and the southeast portion of St. Mary Parishes, the Park was created in 1993, encompassing 122,367 ac (49,520 ha) of mountainous, forested terrain that ranges in elevation from 492 to 7,402 ft (150 m to 2,256 m). The Park is considered one of the best-managed protected areas in Jamaica (JCDT 2006). Since 2006, regular patrols by Park Rangers have averaged 11 per month, resulting in interdiction of illegal activities including hunting, logging, and dumping (JCDT and Green Jamaica 2007b). Moreover, since December 2006, the Park has instituted ''Kite butterfly patrols'' to locate the Jamaican kite swallowtail, which resulted in the most recent observation of 4 individuals in mid-2007 (JCDT and Green Jamaica 2007a). However, deforestation is currently a threat to the species' habitat in the Blue Mountains (Tole 2006), and enforcement within the Park is hampered by lack of vehicles, limited computer access, and a lack of clearly defined Park boundaries.

The Jamaican kite swallowtail has been collected for commercial trade (Collins and Morris 1985; Melisch 2000; Schütz 2000) and has been protected under the Jamaican Wildlife Protection Act since 1998. This Act carries a maximum penalty of 1,439 USD (100,000 Jamaican dollars (J$)) or 12 months imprisonment and appears to be effectively protecting this species from illegal trade (NEPA 2005). This species is not listed under CITES, nor is it listed on the European Commission's Annex B (Eur-Lex 2008), both of which regulate international trade in animals and plants of conservation concern. However, we are not aware of any recent seizures or smuggling of this species into or out of the United States (Office of Law Enforcement, U.S. Fish and Wildlife Service, Arlington, Virginia, in litt. 2008) and we are unaware of any ongoing trade in this species. Therefore, we believe that overutilization is not currently a contributory risk factor to the Jamaican kite swallowtail.

The Jamaican kite swallowtail does not represent a monotypic genus. Habitat modification is the primary threat to this species and we have determined that overutilization is not currently a contributory risk factor. The current threat from habitat modification includes: (1) Historical habitat modification at the species' only known breeding site, which has lasting impacts on this species given that the species does not thrive in disturbed habitats; (2) ongoing habitat alteration throughout its adult range (including the felling of this species' larval plant food); and (3) the potential for stochastic events, such as hurricanes, to contribute to habitat loss. However, this threat is moderate in magnitude because Jamaica has taken regulatory steps to preserve the species and its habitat, and adults are being regularly observed within at least one protected area, indicating that the species continues to be viable. The threat from habitat modification is imminent because habitat destruction is ongoing. Therefore, it receives a priority rank of 8.

Fluminense Swallowtail (*Parides ascanius*)

The fluminense swallowtail is endemic to Brazil's ''restinga'' habitat within the Atlantic Forest region (Thomas 2003). Restingas form on sandy, acidic, and nutrient-poor soils in the tropical and subtropical moist broadleaf forests of coastal Brazil. Restinga habitat, also referred to as ''fluminense vegetation,'' is characterized by medium-sized trees and shrubs that are adapted to coastal conditions (Kelecom 2002). The species is monophagous (Otero and Brown 1984), meaning that its larvae feed only on a single plant species (Kotiaho et al. 2005); information on larval hostplant preferences is provided in the 2007 Notice of Review (72 FR 20184).

The species was historically reported in Rio de Janeiro, Espirito Santo, and Sao Paulo. However, there are no recent confirmations in Espirito Santo or Sao Paulo. In Rio de Janeiro, the species is reported in five localities, including: Barra de São João and Macaé (in the Restinga de Jurubatiba National Park), along the coast; and, Poço das Antas Biological Reserve, further inland (Keith S. Brown, Jr., Livre-Docent, Universidade Estadual de Campinas, Brazil, in litt. 2004; Soler 2005). Uehara-Prado and Fonseca (2007) recently reported a verified occurrence within Área de Tombamento do Mangue do rio Paraíba do Sul. Fluminense swallowtail has also been reported in Parque Natural Municipal do Bosque da Barra (Instituto Iguacu 2008).

The fluminense swallowtail is sparsely distributed throughout its range, reflecting the patchy distribution of its preferred habitat (Otero and Brown 1984; Tyler et al. 1994; Uehara-Prado and Fonseca 2007). However, the species can be seasonally common, with sightings of up to 50 individuals in one morning in the Barra de São João location. The population estimate in Barra de São João ranges from 20 to 100 individuals (Otero and Brown 1984). The colony within Poço das Antas Biological Reserve (Reserve) was rediscovered in 1997, after a nearly 30-year absence from this locality (K. Brown, Jr., in litt. 2004). Researchers noted only that ''large numbers'' of swallowtails were observed (K. Brown, Jr., in litt. 2004; Dr. Robert Robbins, Research Entomologist, National Museum of Natural History, Department of Entomology, Smithsonian Institution, Washington, D.C., in litt. 2004). There are no population estimates for the other colonies. However, individuals from the viable population in Barra de São João migrate widely in some years, which is likely to enhance inter-population gene flow among existing colonies (K. Brown, Jr., in litt. 2004).

Brazil considers the fluminense swallowtail to be ''Imperiled'' (MMA 2003; Portaria No. 1,522 1989). According to the 2007 IUCN Red List (Gimenez Dixon 1996), the fluminense swallowtail has been categorized as ''Vulnerable'' since 1983, based on its small distribution and a decline in the number of populations caused by habitat fragmentation and loss. However, this species has not been re-evaluated using the 1997 IUCN Red List categorization criteria.

Habitat destruction has been the main threat to this species (Brown 1996; Collins and Morris 1985; Gimenez Dixon 1996). Monophagous butterflies tend to be more threatened than polyphagous species (Kotiaho et al. 2005), and the restinga habitat preferred by fluminense swallowtails is a highly specialized environment that is restricted in distribution (K. Brown, Jr., in litt. 2004; Otero and Brown 1986; Uehara-Prado and Fonseca). Moreover, fluminense swallowtails require large areas to maintain viable populations (K. Brown, Jr., in litt. 2004; Otero and Brown 1986; Uehara-Prado and Fonseca). The Atlantic Forest habitat, which once covered 540,543 mi^2(1.4 million km^2), has been reduced to 5 to 10 percent of its original cover and harbors more than 70 percent of the Brazilian population (Butler 2007; Conservation International 2007; Critical Ecosystem Partnership Fund (CEPF) 2007a; Hfling 2007; The Nature Conservancy 2007; World Wildlife Fund (WWF) 2007). The restinga habitat upon which this species depends, has been reduced by 6.56 mi^2 (17 km^{2}2) each year between 1984 and 2001, equivalent to a loss of 40 percent of restinga vegetation over the 17-year period (Temer 2006). The major ongoing human activities that have resulted in habitat loss, degradation, and fragmentation include conversion for agriculture, plantations, livestock pastures, human settlements, hydropower reservoirs, commercial logging, subsistence activities, and coastal development (Butler 2007; Hughes et al. 2006; Pivello 2007; The Nature Conservancy 2007; Peixoto and Silva 2007; World Food Prize 2007; WWF 2007).

Uehara-Prado and Fonseca (2007) estimated that Rio de Janeiro contains 4,140,127 ac (1,675,457 ha) of suitable habitat (Uehara-Prado and Fonseca 2007). While the presence of suitable habitat should not be used to infer the presence of a species, this research should facilitate more focused efforts to identify and confirm additional localities and conservation status of the fluminense swallowtail (Uehara-Prado and Fonseca 2007). Analyzing the correlation between the distribution of fluminense swallowtail and the existing protected areas within Rio de Janeiro, Uehara-Prado and Fonseca (2007) found that only two known occurrences of the fluminense swallowtail correlated with protected areas, including the Poço das Antas Biological Reserve. The researchers concluded that the existing protected area system may be inadequate for the conservation of this species.

The Poço das Antas Biological Reserve and the Jurubatiba National Park are the only two protected areas considered large enough to support viable populations of the fluminense swallowtail (K. Brown, Jr., in litt. 2004; Otero and Brown 1984; R. Robbins, in litt. 2004). The Poço das Antas Biological Reserve (Reserve), established in 1974, encompasses 13,096 ac (5,300 ha) of inland Atlantic Forest habitat (CEPF 2007a; Decree No. 73,791 1974). According to the 2005 revised management plan (IBAMA 2005), the Reserve is used solely for protection, research, and environmental education. Public access is restricted, and there is an emphasis on habitat conservation, including protection of the R̃o São João. This river runs through the Reserve and is integral to creating the restinga conditions preferred by the fluminense swallowtail. The Reserve was plagued by fires in the late 1980s through the early 2000s, but there have been no recent reports of fires. Between 2001 and 2006, there was an increase in the number of private protected areas near or adjacent to the Poço das Antas Biological Reserve and Barra de São João (Critical Ecosystem Partnership Fund (CEPF) 2007a). Corridors are being created between existing protected areas and 13 privately protected forests, by planting and restoring habitat previously cleared for agriculture or by fires (De Roy 2002).

The Jurubatiba National Park (14,860 ha; 36,720 mi), located in Macaé and established in 1998 (Decree of April 29 1998), is one of the largest contiguous restingas (specialized sandy, coastal habitats) under protection in Brazil (CEPF 2007b; Rocha et al. 2007). The Macaé River Basin forms the outer edge of the Jurubatiba National Park (Park) (International Finance Corporation (IFC) 2002) and creates the restinga habitat preferred by the fluminense swallowtail (Brown 1996; Otero and Brown 1984). Rocha et al. (2007) described the habitat as being in a very good state of conservation, but lacking a formal management plan (Rocha et al. 2007). Threats to the Macaé region include industrialization for oil reserve and power development (IFC 2002) and intense population pressures (including migration and infrastructural development) (Brown 1996; CEPF 2007b; IFC 2002; Khalip 2007; Otero and Brown 1984; Savarese 2008).

Commercial exploitation has been identified as a potential threat to the fluminense swallowtail (Collins and Morris 1985; Melisch 2000; Schütz 2000). The species is easy to capture, and species with restricted distributions or localized populations, such as the fluminense swallowtail, tend to be more vulnerable to over-collection than those with a wider distribution (K. Brown, Jr., in litt. 2004; R. Robbins, in litt. 2004). This species has not been formally considered for listing in the Appendices of CITES (*http://www.cites.org*). However, the European Commission listed fluminense swallowtail on Annex B of Regulation 338/97 in 1997. (Dr. Ute Grimm, German Scientific Authority to CITES (Fauna), Bonn, Germany, in litt. 2008), and the species continues to be listed on this Annex (Eur-Lex 2008). This listing requires that imports from a non-European Union country be accompanied by a permit that is only issued if the Scientific Authority has made a positive non-detriment finding, a determination that trade in the species will not be detrimental to the survival of the species in the wild (U. Grimm, in litt. 2008). There has been no legal trade in this species into the European Union since its listing on Annex B (U. Grimm, in litt. 2008), and we are not aware of any recent reports of seizures or smuggling in this species into or out of the United States (Office of Law Enforcement, U.S. Fish and Wildlife Service, Arlington, Virginia, in litt. 2008). The fluminense remains strictly protected from commerce in Brazil (K. Brown, Jr., in litt. 2004). For the reasons outlined above, we believe that overutilization is not currently a contributory threat factor for the fluminense swallowtail.

Parasitism could be a factor threatening the fluminense swallowtail. Recently, Tavares et al. (2006) discovered four species of parasitic chalcid wasps (*Brachymeria* and *Conura* species; Hymenoptera family) associated with fluminense swallowtails. Parasitoids are species whose immature stages develop on or within an insect host of another species, ultimately killing the host (Weeden et al. 1976).

This is the first report of parasitoid association with fluminense swallowtails (Tavares et al. 2006). To date, there is no information as to the extent and effect that these parasites are having on the fluminense swallowtail.

Although Harris' mimic swallowtail and the fluminense swallowtail face similar threats, there are several dissimilarities that influence the magnitude of these threats. Fluminense swallowtails are monophagous (Otero and Brown 1984), meaning that its larvae feed only on a single plant species (Kotiaho et al. 2005). In contrast, Harris' mimic swallowtail is polyphagous (Brown 1996; Collins and Morse 1985), such that its larvae feed on more than one species of plant (Kotiaho et al. 2005). In addition, although their ranges overlap, Harris' mimic swallowtails tolerate a wider range of habitat than the highly specialized restinga habitat preferred by fluminense swallowtail. Also unlike the Harris' mimic swallowtail, fluminense swallowtails require a large area to maintain a viable population (K. Brown, Jr., in litt. 2004; Monteiro et al. 2004).

The fluminense swallowtail does not represent a monotypic genus. The species is currently at risk from habitat destruction and potentially from parasitism; however, we have determined that overutilization is not currently a contributory threat factor for the fluminense swallowtail. The current threat of habitat destruction is of high magnitude because the species: (1) Occupies highly specialized habitat; (2) requires large areas to maintain a viable colony; and (3) is only found within two protected areas considered to be large enough to support viable colonies. However additional populations have been reported, increasing previously known population numbers and distribution. The threat of habitat destruction is non-imminent because most habitat modification is the result of historical destruction that has resulted in fragmentation of the current landscape; however, the potential for continued habitat modification exists, and we will continue to monitor the situation. On the basis of this information, the fluminense swallowtail receives a priority rank of 5.

Hahnel's Amazonian Swallowtail (*Parides hahneli*)

Hahnel's Amazonian swallowtail is endemic to Brazil, found only on ancient sandy beaches, where the habitat is overgrown with dense scrub vegetation (Collins and Morris 1985; New and Collins 1991; Tyler et al. 1994). The species is likely to be monophagous; information on larval and adult hostplant preferences was provided in the 12-month finding (69 FR 70580) and in the 2007 Notice of Review (72 FR 20184).

Hahnel's Amazonian swallowtail is known in three localities along the tributaries of the middle and lower Amazon River basin in the states of Amazonas and Pará (Brown 1996; Collins and Morris 1985; New and Collins 1991; Tyler et al. 1994). Two of these colonies were rediscovered in the 1970s (Brown 1996; Collins and Morris 1985). The species is highly localized, reflecting the localized distribution of its highly specialized preferred habitat (K. Brown, Jr., in litt. 2004). We are unaware of any population estimates for this species, other than the fact that ''the area of its range is very lightly populated'' (K. Brown, Jr., in litt. 2004). This species is not nationally protected (MMA 2003; Portaria No. 1,522 1989), although Pará has included this species as ''Endangered'' on its newly created list of threatened species (Decreto No. 802 2008; Resolução 054 2007; Secco and Santos 2008). This listing requires the Pará government to monitor, protect, conserve, and restore the species and its habitat within the state, which will add to our understanding of the species' ecology (Resolução 054 2007). This species continues to be listed as ''Data Deficient'' by the IUCN Red List (Gimenez Dixon 1996).

Habitat alteration (e.g., for dam construction and waterway crop transport) and destruction (e.g., clearing for agriculture and cattle grazing) are ongoing in the states of Pará and Amazonas, where this species is found (Fearnside 2006; Hurwitz 2007). Because of this species' dependence on highly localized and extremely limited habitat, habitat alteration could be deleterious to the species (New and Collins 1991; Wells et al. 1983). However, because this species' ecological requirements continue to be poorly understood, we are unable to determine whether this species is currently being threatened by habitat alteration.

Hahnel's Amazonian swallowtail is collected for commercial trade (Collins and Morris 1985; Melisch 2000; Schütz 2000), as described in the 2007 Notice of Review (72 FR 20184). In the United States, there continues to be limited trade in the species over the internet, although it is unclear whether the specimens were recently collected. It is not illegal to trade this species in the United States, but possession of wildlife must be declared upon crossing U.S. borders. We are not aware of any recent seizures or smuggling of this species into or out of the United States (Office of Law Enforcement, U.S. Fish and Wildlife Service, Arlington, Virginia, in litt. 2008). This species has not been formally considered for listing in the Appendices of CITES (www.cites.org), but has been listed on Annex B of the European Union's (EU) Regulation 338/97 since 1997 (Eur-Lex 2008); Annex B listings are described under the fluminense swallowtail, above. According to Dr. Ute Grimm (German Scientific Authority to CITES (Fauna), Bonn, Germany, in litt. 2008), there has been no legal trade in this species in the EU since its listing. However, a French importer of exotic specimens is selling Amazonian swallowtail on the internet; multiple specimens of males, females and pairs are available for 18 Euros (28 USD); 20 Euros (32 USD); and 35 Euros (55 USD), respectively. This species is not nationally protected in Brazil (MMA 2003; Portaria No. 1,522 1989). Although the state of Pará recently prohibited capture of this species for purposes other than research (Decreto No. 802 2008), insufficient time has elapsed to determine how effectively this will prevent any wild collection of the species. There have been no recent discoveries of additional populations of Hahnel's Amazonian swallowtail (K.S. Brown, Jr., in litt. 2004) and, of the three known localities, two populations are in the State of Amazonas (Brown 1996; Collins and Morris 1985). Thus, of the populations, two-thirds are not protected from collection. According to experts, species with restricted distributions or localized populations, such as the Hahnel's Amazonian swallowtail, are more vulnerable to over-collection than those with a wider distribution (K. Brown, Jr., in litt. 2004; R. Robbins, in litt. 2004). Therefore, we believe that overutilization for commercial purposes, combined with insufficient regulatory mechanisms, constitute a threat to the Hahnel's Amazonian swallowtail.

Competition has been identified as a potential threat to this species. Researchers have posited that the Hahnel's Amazonian swallowtail might suffer from host-plant competition with any of three other butterfly species that occupy a similar range (Brown 1996; Collins and Morris 1985; Wells 1983) (See 2007 Notice of Review (72 FR 20184)). Therefore, competition may be a contributory threat factor for the Hahnel's Amazonian swallowtail.

Hahnel's Amazonian swallowtail does not represent a monotypic genus. The main threat to this species is overcollection combined with inadequate regulatory mechanisms to mitigate this threat. Habitat destruction and host-plant competition may be contributory threats. We are currently aware of only a small amount of trade in this species, so we rank the threat of overutilization as low to moderate and non-imminent. Thus, this species receives a priority rank of 11.

Kaiser-I-Hind Swallowtail (*Teinopalpus imperialis*)

The Kaiser-I-Hind swallowtail is native to the Himalayan regions of Bhutan, China, India, Laos, Myanmar, Nepal, Thailand, and Vietnam (Baral et al. 2005; Food and Agriculture Organization (FAO) 2001; FRAP 1999; Igarashi 2001; Masui and Uehara 2000; Osada et al. 1999; Shrestha 1997; TRAFFIC 2007; Tordoff et al. 1999; Trai and Richardson 1999). This species prefers undisturbed (primary), heterogeneous broad-leaved evergreen forests or montane deciduous forests, and flies at altitudes of 4,921 to 10,000 ft (1,500 to 3,050 m) (Collins and Morris 1985; Igarashi 2001; Tordoff et al. 1999). Information on this polyphagous species' biology and food plant preferences is provided in the 2007 Notice of Review (72 FR 20184). It should be noted that Collins and Morris (1985) reported that the adult Kaiser-I-Hind swallowtails do not feed. This is a correction to the 2007 Notice of Review (72 FR 20184), which stated that the adult food plant preferences were unknown. Since 1996, the Kaiser-I-Hind swallowtail has been categorized on the IUCN Red List as a species of ''Least Concern''; it has not been re-evaluated using the 1997 criteria (Gimenez Dixon 1996). The species is considered ''Rare'' by Collins and Morris (1985). Despite its widespread distribution, local populations are not abundant (Collins and Morris 1985). The known localities and conservation status of the species within each range country follows:

Bhutan: The species was reported to be extant in Bhutan (Gimenez Dixon 1996; FRAP 1999), although details on localities or status information were not provided.

China: The species has been reported in Fuji, Guangxi, Hubei, Jiangsu, Sichuan, and Yunnan Provinces (Collins and Morris 1985; Gimenez Dixon 1996; Igarashi and Fukuda 2000; Sung and Yan 2005; United Nations Environment Programme-World Conservation Monitoring Center (UNEP–WCMC) 1999). The species is classified by the 2005 China Species Red List as ''Vulnerable'' (China Red List 2006).

India: Assam, Manipur, Meghalaya, Sikkim, and West Bengal (Bahuguna 1998; Collins and Morris 1985; Gimenez Dixon 1996; Ministry of Environment and Forests 2005). There is no recent status information on this species (N. Chaturvedi, Curator, Bombay Natural History Society, Mumbai, India, in litt. 2007).

Laos: The species has been reported (Osada et al. 1999), but no further information is available (Southiphong Vonxaiya, CITES Coordinator, Vientiane, Lao, in litt. 2007).

Myanmar: The species has been reported in Shan, Kayah (Karen) and Thaninanthayi (Tenasserim) states (Collins and Morris 1985; Gimenez Dixon 1996). There is no status information.

Nepal: The species has been reported in Nepal (Collins and Morris 1985; Gimenez Dixon 1996), in the Central Administrative Region at two localities: Phulchoki Mountain Forest (Baral et al. 2005; Collins and Morris 1985) and Shivapuri National Park (Nepali Times 2002; Shrestha 1997). There is no status information.

Thailand: The species has been reported in the northern province of Chang Mai (Pornpitagpan 1999). The Scientific Authority of Thailand recently confirmed that the species has limited distribution in the high mountains (>1,500 m (4,921 ft)) of northern Thailand and is found within three national parks. However, no biological or status information was available (S. Choldumrongkul, Forest Entomology and Microbiology Group, Department of National Parks, Bangkok, Thailand, in litt. 2007).

Vietnam: The species has been confirmed in three Nature Reserves (Tordoff et al. 1999; Trai and Richardson 1999), and the species is listed as ''Vulnerable'' in the 2007 Vietnam Red Data Book, due to declining population sizes and area of occupancy (Dr. Le Xuan Canh, Director of the Institute of Ecology and Biological Resources, CITES Scientific Authority, Hanoi, Vietnam, in litt. 2007).

Habitat destruction is the greatest threat to this species, which prefers undisturbed high altitude habitat (Collins and Morris 1985; Igarashi 2001; Tordoff et al. 1999). In China and India, the Kaiser-I-Hind swallowtail populations are at risk from habitat modification and destruction due to commercial and illegal logging (Yen and Yang 2001; Maheshwari 2003). In Nepal, the species is at risk from habitat disturbance and destruction resulting from mining, fuel wood collection, agriculture, and grazing animals (Baral et al. 2005; Collins and Morris 1985; Shrestha 1997). Nepal's Forest Ministry considered habitat destruction to be a critical threat to all biodiversity, including the Kaiser-I-Hind swallowtail, in the development of their biodiversity strategy (HMGN 2002). Habitat degradation and loss caused by deforestation and land conversion for agricultural purposes is a primary threat to the species in Thailand (Hongthong 1998; FAO 2001). The species is afforded some protection from habitat destruction in Vietnam, where it has been confirmed in three Nature Reserves that have low levels of disturbance (Tordoff et al. 1999; Trai and Richardson 1999).

The Kaiser-I-Hind swallowtail is highly valued and has been collected for commercial trade, despite range country regulations prohibiting or restricting such activities (Collins and Morris 1985; Schutz 2000). In China, where the species is protected by the Animals and Plants (Protection of Endangered Species) Ordinance (1989), which restricts import, export and possession of the species, species purportedly derived from Sichuan were being advertised for sale on the internet for 60 USD. In India, the Kaiser-I-Hind swallowtail is listed on Schedule II of the Indian Wildlife Protection Act of 1972, which prohibits hunting without a license (Collins and Morris 1985; Indian Wildlife Protection Act 2006). However, between 1990 and 1997, illegally collected specimens were selling for 500 Rupees (12 USD) per female and 30 Rupees (0.73 USD) per male (Bahuguna 1998). In Nepal, the Kaiser-I-Hind swallowtail is protected by the National Parks and Wildlife Conservation Act of 1973 (His Majesty's Government of Nepal (HMGN) 2002). However, the Nepal Forestry Ministry determined in 2002 that the high commercial value of its ''Endangered'' species on the local and international market may result in local extinctions of species such as the Kaiser-I-Hind (HMGN 2002). In Thailand, the Kaiser-I-Hind swallowtail and 13 other invertebrates are listed under Thailand's Wildlife Reservation and Protection Act (WARPA) of 1992 (B.E. 2535 1992), which makes it illegal to collect wildlife (whether alive or dead) or to have the species in one's possession (S. Choldumrongkul, in litt. 2007; FAO 2001; Hongthong 1998; Pornpitagpan 1999). In addition to prohibiting possession, WARPA prohibits hunting, breeding, and trading; import and export are only allowed for conservation purposes (Jeerawat Jaisielthum, CITES Management Authority, Bangkok, Thailand, in litt. 2007). According to the Thai Scientific Authority, there are no captive breeding programs for this species; however, the species is offered for sale by the Lepidoptera Breeders Association (2008), being marketed as derived from a captive breeding program in Thailand. In Vietnam, Kaiser-I-Hind swallowtails are reported to be among the most valuable of all butterflies (World Bank 2005). The species was recently listed on Schedule IIB of Decree No. 32 (2006) on ''Management of endangered, precious and rare forest plants and animals.'' A Schedule IIB-listing restricts the exploitation or commercial use of species with small populations or considered by the country to be in danger of extinction (L.X. Canh, in litt. 2007). In a recent survey conducted by TRAFFIC Southeast Asia (2007), of 2000 residents in Hanoi, Vietnam, the Kaiser-I-Hind swallowtail was among 37 Schedule IIB-species that were actively being collected, and the majority of the survey respondents were unaware of legislation prohibiting collection of Schedule IIB-species. Thus, overutilization for illegal domestic and possibly international trade via the internet is a threat to this species, and within-country protections are inadequate to protect the species from illegal collection throughout its range.

The Kaiser-I-Hind swallowtail has been listed in CITES Appendix II since 1987 (UNEP–WCMC 2008a). Between 1991 and 2005, 160 Kaiser-I-Hind swallowtail specimens were traded internationally under CITES permits (UNEP WCMC 2006). The most recent CITES trade data are available for the year 2006. The only recorded international trade in this year was one shipment of two specimens, imported as personal effects into the United States from Vietnam (UNEP WCMC 2008b). Reports that the Kaiser-I-Hind swallowtail is being captive-bred in Taiwan (Yen and Yang 2001) remain unconfirmed. Since 1993, there have been no reported seizures or smuggling of this species into or out of the United States (Office of Law Enforcement, U.S. Fish and Wildlife Service, Arlington, Virginia, in litt. 2008). Therefore, on the basis of global trade data, we do not consider legal international trade to be a contributory threat factor to this species.

The Kaiser-I-Hind swallowtail does not represent a monotypic genus. The current threats of habitat destruction and collection are moderate to low in magnitude due to the species' wide distribution, but imminent due to ongoing habitat destruction, high market value for specimens, and inadequate domestic protections for the species or its habitat. Therefore, it receives a priority rank of 8.

Preclusion and Expeditious Progress

Below we describe the actions that continue to preclude the immediate proposal of listing rules for the 20 species described above. In addition, we summarize the expeditious progress we are making, as required by section 4(b)(3)(B)(iii)(II) of the Act, to add qualified species to the lists of endangered or threatened species and to remove from these lists species for which protections of the Act are no longer necessary.

Section 4(b) of the Act states that the Service may make warranted-but-precluded findings only if it can demonstrate that (1) An immediate proposed rule is precluded by other pending proposals and that (2) expeditious progress is being made on other listing actions. Preclusion is a function of the listing priority of a species in relation to the resources that are available and competing demands for those resources. Thus, in any given fiscal year (FY), multiple factors dictate whether it will be possible to undertake work on a proposed listing regulation or whether promulgation of such a proposal is warranted but precluded by higher priority listing actions.

The listing of foreign species under the Act is carried out by a different Service program than the domestic Endangered Species Program. The Division of Scientific Authority (DSA), within the Service's International Affairs program, is solely responsible for the development of all listing proposals for foreign species and promulgation of final rules, whether internally driven or as the result of a petition.

In the upcoming year, publication of proposed rules for the 20 species described above is precluded by the need to complete pending listing actions as described below. Of the actions listed below, preparation of a final listing rule for the six species of Procellariids is DSA's highest priority.

DSA will be working on a final listing determination for six species of foreign Procellariids that we proposed for listing on December 17, 2007 (72 FR 71298). Reaching a final decision on this proposed rule is consistent with the statutory deadlines under sections 4(b)(5) and 4(b)(6) of the Act and takes precedence over proposed listings that are warranted but precluded by higher priorities.

On January 23, 2008, the United States District Court ordered the Service to propose listing rules for five foreign bird species, actions which we previously considered to be warranted but precluded. These species are: the Chilean woodstar (*Eulidia yarrellii*), Andean flamingo (*Phoenicoparrus andinus*), medium tree-finch (*Camarhynchus pauper*), black-breasted puffleg (*Eriocnemis nigrivestis*), and the St. Lucia forest thrush (*Cichlherminia herminieri sanctaeluciae*). We, therefore, have a court-ordered responsibility to publish proposed listing rules for these five species by December 31, 2008.

The government of Mexico, through the National Commission for the Understanding and Use of Biodiversity (CONABIO), has petitioned us to delist the Morelet's crocodile (*Crocodylus moreletii*), a species that is under its jurisdiction and is listed under the Act. The petition was received by the Service on May 26, 2005. A 90-day finding was published on June 28, 2006 (71 FR 36743), indicating that the petitioned action may be warranted. The status review is currently in progress, and we must complete work on the 12-month finding on this petition, consistent with our responsibilities under section 4(b)(3) of the Act.

The government of Argentina has petitioned us to reclassify the broad-snouted caiman (*Caiman latirostris*) in Argentina from endangered to threatened under the Act. The petition was dated November 5, 2007. A 90-day finding was published on June 16, 2008 (73 FR 33968), indicating that the petitioned action may be warranted. The status review is currently in progress, and we must complete work on the 12-month finding on this petition, consistent with our responsibilities under section 4(b)(3) of the Act.

We are also in the process of making a final determination on whether to delist the Mexican bobcat (*Lynx rufus escuinapae*). The United States, with support from Mexico and other countries, proposed to transfer the Mexican bobcat from CITES Appendix I to Appendix II, based on the Mexican bobcat's widespread and stable status in Mexico and the questionable taxonomy of the subspecies. The U.S. proposal was accepted and the change went into effect on November 6, 1992. On July 8, 1996, we received a petition from the National Trappers Association, Inc. to delist the Mexican bobcat. Our 12-month finding and proposed rule were published on May 19, 2005 (70 FR 28895). Under section 4(b)(6) of the Act, we have a statutory responsibility to make a final determination.

We are also making a final determination on whether to delist the scarlet-chested parakeet (*Neophema splendida*) and the turquoise parakeet (*Neophema pulchella*). On September 22, 2000, we announced a review of all endangered and threatened foreign species in the Order Psittaciformes as part of a 5-year review under section 4(c)(2) of the Act (65 FR 57363). One commenter suggested we consider these two species for delisting. The individual provided substantial scientific information, including information and correspondence with the government of Australia (the range country of these species) regarding the status of both species. Under section 4(b)(6) of the Act, we have a statutory responsibility to complete this rulemaking process.

On January 4, 2005, we received a petition from 14 county officials representing 13 western States to list the Northern snakehead fish (*Channa argus*) as threatened or endangered under the Act, and further, to designate the Chesapeake Bay region as critical habitat. On March 5, 2005, we received a petition from a private individual to delist the tiger (*Panthera tigris*). On December 3, 2007, we received a petition from Canada's wood bison recovery team to reclassify the wood bison (*Bison bison athabascae*) under the Act. On January 31, 2008, we received a petition from the Environmental Law Clinic at the University of Denver on behalf of Friends of Animals to list 14 species of foreign parrots as endangered or threatened under the Act. The petitioned species include: Blue-throated macaw (*Ara glaucogularis*), blue-headed macaw (*Propyrrhura couloni*), crimson shining parrot (*Prosopeia splendens*), great green macaw (*Ara ambiguous*), grey-cheecked parakeet (*Brotogeris pyrrhoptera*), hyacinth macaw (*Anodorhynchus hyacinthinus*), military macaw (*Ara militaris*), Philippine cockatoo (*Cacatua haematuropygia*), red-crowned parrot (*Amazona viridigenalis*), scarlet macaw (*Ara macao*), thick-billed parrot (*Rhynchopsitta pachyrhyncha*), white cockatoo (*Cacatua alba*), yellow-billed parrot (*Amazona collaria*), and yellow-crested cockatoo (*Cacatua sulphurea*). We have a statutory responsibility under section 4(b)(3) of the Act to process these petitions.

At the current time, we are also preparing proposed listing rules for 25 additional species, petitioned actions that have been determined to be warranted in this Notice of Review. Finally, during the upcoming year, we will be preparing the 2009 Notice of Review, which will set priorities for the next set of listing actions. Using our best efforts to meet our statutory responsibilities under the Act is a high priority.

Despite the priorities which preclude publishing proposed listing rules, we are making expeditious progress in adding to and removing species from the Federal lists of threatened and endangered species. Our expeditious progress since publication of the 2007 Notice of Review, April 23, 2007, to the current date includes preparing and publishing the following: (1) Final rule listing the 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*) as endangered under the Act, published January 16, 2008 (73 FR 3146); (2) Proposed rule to list the Chatham petrel (*Pterodroma axillaris*), Fiji petrel (*Pterodroma macgillivrayi*), and the magenta petrel (*Pterodroma magentae*) as endangered, and the Cook's petrel (*Pterodroma cookii*), Galapagos petrel (*Pterodroma phaeopygia*), and the Heinroth's shearwater (*Puffinus heinrothi*) as threatened under the Act, published December 17, 2007 (72 FR 71298); (3) Notice of 90-day petition finding and initiation of status review of the broad-snouted caiman to determine if reclassification of the population in Argentina, as petitioned, is warranted under the Act, published June 16, 2008 (73 FR 33968); and (4) Notice of 90-day finding on a petition submitted by the Center for Biological Diversity (CBD) to list 12 species of penguin as threatened or endangered under the Act, published July 11, 2007 (72 FR 37695). The 12 penguin species in the CBD petition include: Emperor penguin (*Aptenodytes forsteri*), southern rockhopper penguin (*Eudyptes chrysocome*), northern rockhopper penguin (*Eudyptes moseleyi*), fiordland crested penguin (*Eudyptes pachyrhynchus*), snares crested penguin (*Eudyptes robustus*), erect-crested penguin (*Eudyptes sclateri*), macaroni penguin (*Eudyptes chrysolophus*), royal penguin (*Eudyptes schlegeli*), white-flippered penguin (*Eudyptula albosignata*), yellow-eyed penguin (*Megadyptes antipodes*), African penguin (*Spheniscus demersus*), and Humboldt penguin (*Spheniscus humboldti*). In our 90-day finding on this petition, we found that listing 10 of the 12 penguin species may be warranted, and we initiated a status review of these 10 species. We found that the petition did not provide substantial scientific or commercial information indicating that listing of either the snares crested penguin or royal penguin may be warranted. The 12-month petition finding addressing the other 10 species listed above is pending Departmental review.

Our expeditious progress also includes work on pending listing actions described above in our ''precluded finding,'' but for which decisions had not been completed at the time of this publication, including: (1) Final listing determination for six species of foreign Procellariids; (2) proposed listing rules for five foreign bird species that were court-ordered for publication; (3) proposed listing rules for 25 additional foreign bird species that were the subjects of listing petitions determined to be warranted in this Notice of Review; (4) 90-day finding on a petition to list the Northern snakehead fish as threatened or endangered under the Act; and (5) 90-day finding on a petition to list 14 species of foreign parrots as endangered or threatened under the Act.

We have endeavored to make our listing actions as efficient and timely as possible, given the requirements of the relevant law and regulations and the constraints relating to workload and personnel. We are continually considering ways to streamline processes or achieve economies of scale, such as by batching related actions together. Despite higher listing priorities that preclude us from issuing listing proposals for the 20 species mentioned in this Notice of Review, the actions described above collectively constitute expeditious progress.

Monitoring

Section 4(b)(3)(C)(iii) of the Act requires us to ''implement a system to monitor effectively the status of all species'' for which we have made a warranted-but-precluded 12-month finding, and to ''make prompt use of the [emergency listing] authority [under section 4(b)(7)] to prevent a significant risk to the well being of any such species.'' For foreign species, the Service's ability to gather information to monitor species is limited. The Service welcomes all information relevant to the status of these species, because we have no ability to gather data in foreign countries directly and cannot compel another country to provide information. Thus, this ANOR plays a critical role in our monitoring efforts for foreign species. With each ANOR, we request information on the status of the species included in the notice. Information and comments on the annual findings can be submitted at any time. We review all new information received through this process as well as any other new information we obtain using a variety of methods. We collect information directly from range countries by correspondence, from the peer-reviewed scientific literature, unpublished literature, scientific meeting proceedings, and CITES documents (including species proposals and reports from scientific committees). We also obtain information through the permit application processes under CITES, the Act, and the Wild Bird Conservation Act. We also consult with staff members of the Service's Division of International Conservation and the IUCN species specialist groups, and we attend scientific meetings to obtain current status information for relevant species. As previously stated, if we identify any species for which emergency listing is appropriate, we will make prompt use of the emergency listing authority under section 4(b)(7) of the Act.

Request for Information

We request the submission of any further information on the species in this notice as soon as possible, or whenever it becomes available. We especially seek information: (1) Indicating that we should remove a taxon from warranted status; (2) documenting threats to any of the included taxa; (3) describing the immediacy or magnitude of threats facing these taxa; (4) pointing out taxonomic or nomenclatural changes for any of the taxa; (5) suggesting appropriate common names; or (6) noting any mistakes, such as errors in the indicated historic ranges.

References Cited

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

Authors

This Notice of Review was authored by the staff of the Division of Scientific Authority, U.S. Fish and Wildlife Service (see **ADDRESSES** section).

Authority

This Notice of Review is published under the authority of the Endangered Species Act (16 U.S.C. 1531 *et seq.*).

TABLE 1.—CANDIDATE REVIEW

[C = listing warranted by precluded; P = to be proposed to be listed]

Status		Scientific name	Family	Common name	Historic range
Birds Category	Priority				
P	N/A	*Podiceps taczanowskii*	Podicipedidae	Junin flightless grebe	Peru.
P	N/A	*Leptoptilos dubius*	Ciconiidae	greater adjutant stork	South Asia.
P	N/A	*Phoenicopterus andinus*	Phoenicopteridae	Andean flamingo	Argentina, Bolivia, Chile, Peru.
P	N/A	*Mergus octosetaceus*	Anatidae	Brazilian merganser	Brazil.
P	N/A	*Penelope perspicax*	Craciidae	Cauca guan	Colombia.
C	8	*Pauxi unicornis*	Craciidae	southern helmeted curassow.	Bolivia, Peru.
P	N/A	*Crax alberti*	Craciidae	blue-billed curassow	Colombia.
P	N/A	*Tetrao urogallus cantabricus*	Tetraonidae	Cantabrian capercaillie	Spain.
P	N/A	*Odontophorus strophium*	Odontophoridae	gorgeted wood-quail	Colombia.
P	N/A	*Laterallus tuerosi*	Rallidae	Junin rail	Peru.
C	8	*Rallus semiplumbeus*	Rallidae	Bogota rail	Colombia.
C	8	*Porphyrio hochstetteri*	Rallidae	Takahe	New Zealand.
C	8	*Haematopus chathamensis*	Haematopodidae	Chatham oystercatcher	Chatham Islands, New Zealand.
P	N/A	*Rhinoptilus bitorquatus*	Glareolidae	Jerdon's courser	India.
P	N/A	*Numenius tenuirostris*	Scolopacidae	slender-billed curlew	Africa, Algeria, Bulgaria, southern Europe, Greece, Hungary, Italy, Kazakhstan, Morocco, Romania, Russia, Tunisia, Turkey, Ukraine, and Yugoslavia.
P	N/A	*Ducula galeata*	Columbidae	Marquesan imperial-pigeon.	Marquesas Islands, French Polynesia.
P	N/A	*Cacatua moluccensis*	Cacatuidae	salmon-crested cockatoo	South Moluccas, Indonesia.
C	8	*Cyanoramphus malherbi*	Psittacidae	orange-fronted parakeet	New Zealand.
C	8	*Eunymphicus uvaeensis*	Psittacidae	Uvea parakeet	Uvea, New Caledonia.
C	8	*Ara glaucogularis*	Psittacidae	blue-throated macaw	Bolivia.
P	N/A	*Neomorphus geoffroyi dulcis*	Cuculidae	southeastern rufous-vented ground cuckoo.	Brazil.
P	N/A	*Phaethornis malaris margarettae.*	Trochilidae	Margaretta's hermit	Brazil.
P	N/A	*Eriocnemis nigrivestis*	Trochilidae	black-breasted puffleg	Ecuador.
P	N/A	*Eulidia yarrellii*	Trochilidae	Chilean woodstar	Chile, Peru.
P	N/A	*Acestrura berlepschi*	Trochilidae	Esmeraldas woodstar	Equador.
C	8	*Dryocopus galeatus*	Picidae	helmeted woodpecker	Argentina, Brazil, Paraguay.
C	8	*Dendrocopus noguchii*	Picidae	Okinawa woodpecker	Okinawa Island, Japan.
C	11	*Aulacorhynchus huallagae*	Ramphastidae	yellow-browed toucanet	Peru.
P	N/A	*Cinclodes aricomae*	Furnariidae	royal cinclodes	Bolivia, Peru.
P	N/A	*Leptasthenura xenothorax*	Furnariidae	white-browed tit-spinetail	Peru.
P	N/A	*Formicivora erythronotos*	Thamnophilidae	black-hooded antwren	Brazil.
P	N/A	*Pyriglena atra*	Thamnophilidae	fringe-backed fire-eye	Brazil.
P	N/A	*Grallaria milleri*	Formicariidae	brown-banded antpitta	Colombia.

TABLE 1.—CANDIDATE REVIEW—Continued

[C = listing warranted by precluded; P = to be proposed to be listed]

Status		Scientific name	Family	Common name	Historic range
Birds Category	Priority				
C	8	*Scytalopus novacapitalis*	Conopophagidae	Brasilia tapaculo	Brazil.
P	N/A	*Hemitriccus kaempferi*	Tyrannidae	Kaempfer's tody-tyrant	Brazil.
P	N/A	*Anairetes alpinus*	Tyrannidae	ash-breasted tit-tyrant	Bolivia, Peru.
P	N/A	*Phytotoma raimondii*	Phytotomidae	Peruvian plantcutter	Peru.
P	N/A	*Cichlherminia iherminieri sanctaeluciae*.	Turdidae	St. Lucia forest thrush	St. Lucia Island, West Indies.
P	N/A	*Acrocephalus caffer aquilonis*	Sylviidae	Eiao Polynesian warbler	Marquesas Islands, French Polynesia.
C	12	*Bowdleria punctata wilsoni*	Sylviidae	Codfish Island fernbird	Codfish Island, New Zealand.
C	8	*Zosterops luteirostris*	Zosteropidae	Ghizo white-eye	Solomon Islands.
P	N/A	*Camarhynchus pauper*	Thraupidae	medium tree-finch	Floreana Island, Galapagos Islands, Ecuador.
P	N/A	*Nemosia rourei*	Thraupidae	cherry-throated tanager	Brazil.
C	8	*Tangara peruviana*	Thraupidae	black-backed tanager	Brazil.
C	12	*Strepera graculina crissalis*	Cracticidae	Lord Howe pied currawong.	Lord Howe Islands, New South Wales.

Status		Scientific name	Synonyms	Common name	Historic range
Invertebrates Category	Priority				
C	12	*Eurytides lysithous harrisianus*.	*Graphium lysithous harrisianus; Mimoides lysithous harrisianus*.	Harris' mimic swallowtail	Brazil, Paraguay.
C	8	*Eurytides marcellinus*	*Graphium marcellinus; Neographium marcellinus; Protographium marcellinus* (*nom. inv.*); *Protesilaus marcellinus*.	Jamaican kite swallowtail	Jamaica.
C	5	*Parides ascanius*	n/a	Fluminense swallowtail	Brazil.
C	11	*Parides hahneli*	n/a	Hahnel's Amazonian swallowtail.	Brazil.
C	8	*Teinopalpus imperialis*	n/a	Kaiser-I-Hind swallowtail	Bhutan, China, India, Laos, Myanmar, Nepal, Thailand, Vietnam.

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