

DEPARTMENT OF THE INTERIOR

Fish and Wildlife Service

50 CFR Part 17

[Docket No. FWS-R9-IA-2009-0059]
[96100-1671-0000-B6]

[RIN 1018-AV77]

Endangered and Threatened Wildlife and Plants; Listing Foreign Bird Species in Peru and Bolivia as Endangered Throughout Their Range

AGENCY: Fish and Wildlife Service, Interior.

ACTION: Proposed rule.

SUMMARY: We, the U.S. Fish and Wildlife Service (Service), propose to list the following six South American bird species (collectively referred to as “species” for purposes of this proposed rule) as endangered under the Endangered Species Act of 1973, as amended (Act) (16 U.S.C. 1531 *et seq.*): ash-breasted tit-tyrant (*Anairetes alpinus*), Junín grebe (*Podiceps taczanowskii*), Junín rail (*Laterallus tuerosi*), Peruvian plantcutter (*Phytotoma raimondii*), royal cinclodes (*Cinclodes aricomae*), and white-browed tit-spinetail (*Leptasthenura xenothorax*)—all native to Peru. The ash-breasted tit-tyrant and royal cinclodes are also native to Bolivia. This proposal, if made final, would extend the Act’s protection to these species. The Service seeks data and comments from the public on this proposed rule.

DATES: We will accept comments received or postmarked on or before March 8, 2010. We must receive requests for public hearings, in writing, at the address shown in the **FOR FURTHER INFORMATION CONTACT** section by February 19, 2010.

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

- Federal eRulemaking Portal: <http://www.regulations.gov>. Follow the instructions for submitting comments on Docket No. FWS-R9-IA-2009-0059.
- U.S. mail or hand-delivery: Public Comments Processing, Attn: FWS-R9-IA-2009-0059; Division of Policy and Directives Management; U.S. Fish and Wildlife Service; 4401 N. Fairfax Drive, Suite 222; Arlington, VA 22203.

We will not accept comments by e-mail or fax. We will post all comments on <http://www.regulations.gov>. This generally means that we will post any personal information you provide us (see the Public Comments section below for more information).

FOR FURTHER INFORMATION CONTACT: Douglas Krofta, Chief, Branch of Listing, Endangered Species Program, U.S. Fish and Wildlife Service, 4401 N. Fairfax Drive, Room 420, Arlington, VA 22203; telephone 703-358-2105; facsimile 703-358-1735. If you use a telecommunications device for the deaf (TDD), call the Federal Information Relay Service (FIRS) at 800-877-8339.

SUPPLEMENTARY INFORMATION:**Public Comments**

We intend that any final action resulting from this proposal will be as accurate and as effective as possible. Therefore, we request comments or suggestions on this proposed rule. We particularly seek comments concerning:

- (1) Biological, commercial trade, or other relevant data concerning any threats (or lack thereof) to these species and regulations that may be addressing those threats.
- (2) Additional information concerning the taxonomy, range, distribution, and population size of these species, including the locations of any additional populations of these species.
- (3) Additional information on the biological or ecological requirements of these species.
- (4) Current or planned activities in the areas occupied by these species and possible impacts of such activities on these species.
- (5) Any information concerning the effects of climate change on these species or their habitats.

You may submit your comments and materials concerning this proposed rule by one of the methods listed in the **ADDRESSES** section. We will not consider comments sent by e-mail or fax or to an address not listed in the **ADDRESSES** section.

If you submit a comment via <http://www.regulations.gov>, your entire comment—including any personal identifying information—will be posted on the website. If you submit a hardcopy comment that includes personal identifying information, you may request at the top of your document that we withhold this information from public review. However, we cannot guarantee that we will be able to do so. We will post all hardcopy comments on <http://www.regulations.gov>.

Comments and materials we receive, as well as supporting documentation we used in preparing this proposed rule, will be available for public inspection at <http://www.regulations.gov>, or by appointment, during normal business hours, at the U.S. Fish and Wildlife Service, Endangered Species Program, 4401 N. Fairfax Drive, Room 420,

Arlington, VA 22203; telephone 703-358-2171.

Background

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

Previous Federal Actions

On May 6, 1991, we received a petition (the 1991 petition) from the International Council for Bird Preservation (ICBP) to add 53 foreign bird species to the List of Endangered and Threatened Wildlife, including the six Peruvian bird species that are the subject of this proposed rule. In response to the 1991 petition, we published a substantial 90-day finding on December 16, 1991 (56 FR 65207), for all 53 species and initiated a status review. On March 28, 1994 (59 FR 14496), we published a 12-month finding on the 1991 petition, along with a proposed rule to list 30 African birds under the Act (which included 15 species from the 1991 petition). In that

document, we announced our finding that listing the remaining 38 species from the 1991 petition, including the six Peruvian bird species that are the subject of this proposed rule, was warranted but precluded by higher priority listing actions. We made a subsequent warranted-but-precluded finding for all outstanding foreign species from the 1991 petition, including the six Peruvian bird species that are the subject of this proposed rule, as published in our ANOR on May 21, 2004 (69 FR 29354).

Per the Service's listing priority guidelines (September 21, 1983; 48 FR 43098), our 2007 ANOR identified the listing priority numbers (LPNs) (ranging from 1 to 12) for all outstanding foreign species. The six Peruvian bird species that are the subject of this proposed rule were designated with an LPN of 2, and it was determined that their listing continued to be warranted but precluded because of other listing actions. A listing priority of 2 indicates that the subject species face imminent threats of high magnitude. With the exception of the listing priority ranking of 1, which addresses monotypic genera that face imminent threats of high magnitude, categories 2 and 3 represent the Service's highest priorities.

On July 29, 2008 (73 FR 44062), we published in the **Federal Register** a notice announcing our annual petition findings for foreign species. In that notice, we announced listing to be warranted for 30 foreign bird species, including the six Peruvian bird species which are the subject of this proposed rule, and stated that we would "promptly publish proposals to list these 30 taxa." In selecting these six species from the list of warranted-but-precluded species, we took into consideration the magnitude and immediacy of the threats to the species, consistent with the Service's listing priority guidelines.

On September 8, 2008, the Service received a 60-day notice of intent to sue from the Center for Biological Diversity (CBD) and Peter Galvin over violations of section 4 of the Act for the Service's failure to promptly publish listing proposals for the 30 "warranted" species identified in our 2008 ANOR. Under a settlement agreement approved by the U.S. District Court for the Northern District of California on June 15, 2009, (CBD, *et al. v. Salazar*, 09-cv-02578-CRB), the Service must submit to the **Federal Register** proposed listing rules for the ash-breasted tit-tyrant, Junín grebe, Junín rail, Peruvian plantcutter, royal cinclodes, and white-browed tit-spinetail by December 29, 2009.

Species Information and Factors Affecting the Species

Section 4 of the Act (16 U.S.C. 1533), and its implementing regulations at 50 CFR part 424, set forth the procedures for adding species to the Federal Lists of Endangered and Threatened Wildlife and Plants. A species may be determined to be an endangered or threatened species due to one or more of the five factors described in section 4(a)(1) of the Act. The five factors are: (A) The present or threatened destruction, modification, or curtailment of its habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) the inadequacy of existing regulatory mechanisms; and (E) other natural or manmade factors affecting its continued existence. Listing actions may be warranted based on any of the above threat factors, singly or in combination.

Below is a species-by-species analysis of the five factors. The species are considered in alphabetical order, beginning with the ash-breasted tit-tyrant, followed by the Junín grebe, Junín rail, Peruvian plantcutter, royal cinclodes, and the white-browed tit-spinetail.

I. Ash-breasted tit-tyrant (*Anairetes alpinus*)

Species Description

The ash-breasted tit-tyrant, locally known as "*torito pechiceno*," is a small New World tyrant flycatcher in the Tyrannidae family that is native to high-altitude woodlands of the Bolivian and Peruvian Andes (BirdLife International (BLI) 2000, p. 392; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, pp. 170, 281; Fjeldsá and Krabbe 1990, pp. 468-469; InfoNatura 2007, p. 1; Supreme Decree No. 034-2004-AG 2004, p. 276854). The sexes are similar, with adults approximately 5 inches (in) (13 centimeters (cm)) in length, with dark gray, inconspicuously black-streaked upperparts (BLI 2009o, p. 1; del Hoyo *et al.* 2004, p. 281). Two subspecies (see Taxonomy) are distinguished by their underbelly color, which is yellowish-white in the nominate subspecies and white in the other (BLI 2009o, p. 1) (see Taxonomy). Juvenile plumage is duller in appearance, but otherwise similar to the adult coloration (del Hoyo *et al.* 2004, p. 281).

Taxonomy

When the species was first taxonomically described by Carriker (1933, pp. 27-29), it was placed in its own genus, *Yanacea*; this decision was soon questioned by Zimmer (1940, p.

10). It was not until the 1960s that *Yanacea* was merged into *Anairetes* (a genus long-known as *Spizitornis*) by Meyer de Schauensee (1966, p. 376). Some contemporary researchers have suggested retaining the species within *Yanacea* (Fjeldsá and Krabbe 1990, p. 468). Smith (1971, pp. 269, 275) and Roy *et al.* (1999, p. 74) confirmed that the ash-breasted tit-tyrant is a valid species based on its phylogenetic placement and degree of genetic divergence from other species of *Anairetes*, and recent texts continue to place it in *Anairetes* (e.g., del Hoyo *et al.* 2004, p. 281). Therefore, we accept the species as *Anairetes alpinus*, which also follows the Integrated Taxonomic Information System (ITIS 2009, p. 1). Two subspecies are recognized, including, *A. alpinus alpinus* (the nominate subspecies) and *A. alpinus bolivianus*. These subspecies occur in two disjunct (widely separated) areas (see Current Range) (BLI 2000, p. 392; del Hoyo *et al.* 2004, p. 281; ITIS 2009, p. 1) and are distinguished by the color of their underbellies (see Taxonomy) (BLI 2009o, p. 1).

Habitat and Life History

The ash-breasted tit-tyrant is restricted to semihumid *Polylepis* or *Polylepis - Gynoxys* woodlands, where the species is found at elevations between 12,139 and 15,092 feet (ft) (3,700 and 4,600 meters (m)) above sea level (BLI 2000, p. 392; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, p. 281; Fjeldsá and Krabbe 1990, pp. 468-469; InfoNatura 2007, p. 1). The genus *Polylepis* (no English common name; locally referred to as "*queuña*") (Aucca and Ramsay 2005, p. 1), in the Rosaceae family, comprises approximately 20 species of evergreen bushes and trees (De la Via 2004, p. 10; Kessler 1998, p. 1; Kessler and Schmidt-Lebuhn 2006, pp. 1-2), 19 of which occur in Peru (Chutas *et al.* 2008, p. 3). In Bolivia, the ash-breasted tit-tyrant is associated only with *P. pepei* forests, but the bird is found among a greater variety of *Polylepis* species in Peru (Chutas *et al.* 2008, p. 16; I. Gómez, National Museum of National History-Ornithology Section-Bolivian Fauna Collection, La Paz, Bolivia, e-mail to Division of Scientific Authority, in litt. December 4, 2007, p. 1). On average, *Polylepis* species are 10-33 ft (3-10 m) tall, but may grow to a height of 118 ft (36 m) (Purcell *et al.* 2004, p. 455).

Polylepis woodlands occur as dense forests, as open-canopied stands with more arid understories, or as shrubland with scattered trees (De la Via 2004, pp. 10-11; Fjeldsá and Kessler 1996, as cited in Fjeldsá 2002a, p. 113; Lloyd and

Marsden in press, as cited in Lloyd 2008, p. 532). Ash-breasted tit-tyrants prefer dense *Polylepis* forests (Fjeldsá 2002a, p. 114; Smith 1971, p. 269), which often include a mixture of *Gynoxys* trees (no common name), in the Asteraceae family (De la Via 2004, pp. 10; International Plant Names Index (IPNI) 2009, p. 1). Dense *Polylepis* woodlands are characterized by moss- or vine-laden vegetation, with a shaded understory and a rich diversity of insects, making good feeding grounds for insectivorous birds (De la Via 2004, p. 10), such as the ash-breasted tit-tyrant (BLI 2009o, p. 1; Lloyd 2008, p. 535).

There is little information about the ecology and breeding behavior of the ash-breasted tit-tyrant. The species' territory ranges from 2.5–5 acres (ac) (1–2 hectares (ha)) (BLI 2009o, p. 1). The breeding season appears to occur during late dry season (Collar *et al.* 1992, p. 754)—November and December (BLI 2009o, p. 1). Juveniles have been observed in March and July (Collar *et al.* 1992, p. 754; del Hoyo *et al.* 2004, p. 281). Although species-specific information is not available, tit-tyrant nests are generally finely woven, open cups, built in a bush (Fjeldsá and Krabbe 1990, p. 468). The species may share in rearing responsibilities, as pairs of ash-breasted tit-tyrants have been observed feeding young (BLI 2009o, p. 1; Collar *et al.* 1992, p. 754).

The ash-breasted tit-tyrant forages alone, in family groups, and sometimes in mixed-species flocks. The bird takes short flights, either hovering or perching to consume invertebrates near the tops and outer edges of *Polylepis* shrubs and trees (BLI 2009o, p. 1; del Hoyo *et al.* 2004, p. 281; Engblom *et al.* 2002, p. 58; Fjeldsá and Krabbe 1990, p. 468; Lloyd 2008, p. 535). In winter, when invertebrate populations diminish, tit-tyrants may also forage on seeds (Fjeldsá and Krabbe 1990, p. 468).

Historical Range and Distribution

The ash-breasted tit-tyrant may once have been well-distributed throughout previously dense and contiguous *Polylepis* high-Andes woodlands of Peru and Bolivia. Researchers believe that these woodlands were historically contiguous with lower-elevation cloudforests and widespread above 9,843 ft (3,000 m) (Collar *et al.* 1992, p. 753; Fjeldsá 2002a, pp. 111–112, 115; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97–101). Today, *Polylepis* woodlands are found only at elevations of 11,483 to 16,404 ft (3,500 to 5,000 m) (Fjeldsá 1992, p. 10). Researchers consider the reduction in *Polylepis* forest habitat to be the result of historical human activities, including

burning and grazing, which have prevented regeneration of the woodlands and resulted in the fragmented habitat distribution seen today (Fjeldsá and Kessler 1996, Kessler 1995a, Kessler 1995b, and Læggaard 1992, as cited in Fjeldsá 2002a, p. 112; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97–101; Kessler and Herzog 1998, pp. 50–51). Modeling studies by Fjeldsá (2002a, p. 116) indicate that this habitat reduction was accompanied by a loss in species richness. It is estimated that only 2–3 and 10 percent of the original forest cover still remain in Peru and Bolivia, respectively (Fjeldsá and Kessler 1996, as cited in Fjeldsá 2002a, p. 113). Of this amount, only 1 percent of the remaining *Polylepis* woodlands are found in humid areas, where denser stands occur (Fjeldsá and Kessler 1996, as cited in Fjeldsá 2002a, p. 113) and which are preferred by the ash-breasted tit-tyrant (BLI 2009o, p. 1; Fjeldsá 2002a, p. 114; Lloyd 2008, p. 535; Smith 1971, p. 269) (see Factor A).

Current Range and Distribution

The current range of the ash-breasted tit-tyrant is estimated to be 4,595 square miles (mi²) (11,900 square kilometers (km²)) (BLI 2009o, p. 1). However, BirdLife International (2000, pp. 22, 27) defines a species' "Range" as the "Extent of Occurrence," which is "the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred, or projected sites of present occurrence of a species, excluding cases of vagrancy." Given that the species is known to occur in two disjunct locations, this range estimate, therefore, includes a large area of habitat where the species is not known to occur.

The species is restricted to patches of high-elevation semihumid *Polylepis* or *Polylepis*-*Gynoxys* woodlands of Peru and Bolivia, where ash-breasted tit-tyrant is found only at 12,139–15,092 ft (3,700–4,600 m) (BLI 2000, p. 392; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, pp. 170, 281; Fjeldsá and Krabbe 1990, pp. 468–469; InfoNatura 2007, p. 1). The ash-breasted tit-tyrant is known only in two disjunct areas: one location in west-central Peru and another ranging from southern Peru into northern Bolivia (BLI 2000, p. 392; del Hoyo *et al.* 2004, p. 281; ITIS 2009, p. 1).

In west-central Peru, the northern subspecies (*A. alpinus alpinus*) occurs in the Cordilleras Central and Occidental (in the Peruvian Administrative Regions of Ancash, Huánuco, La Libertad, and Lima) (BLI 2007, p. 1; BLI 2009g, p. 1; BLI 2009i,

p. 1; BLI 2009l, p. 1; BLI 2009o, p. 1; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, p. 281). Until 1992, the taxon in this locality was highly localized and known only in Ancash Region (Collar *et al.* 1992, p. 753). The species was subsequently reported in Huánuco Region, in 2003 (BLI 2007, p. 5; BLI 2009i, p. 1); La Libertad Region, in 2004 (del Hoyo *et al.* 2004, p. 281); and Lima Region and again in Huánuco Region, in 2007 (BLI 2007, pp. 1, 5). Also in 2007, the ash-breasted tit-tyrant was also observed in a new locality in Ancash Region, Corredor Conchucos (Chutas 2007, pp. 4, 8), where a *Polylepis* reforestation project is under way to connect two protected areas where ash-breasted tit-tyrants were already known to occur, Parque Nacional Huascarán and Zona Reservada de la Cordillera Huayhuash (Antamina Mine 2006, p. 5; MacLennan 2009, p. 1) (see Factor A).

The second location spans the Peruvian-Bolivian border—in the Peruvian Administrative Regions of Apurímac, Cusco, Puno, and Arequipa (from north to south) and in the Bolivian Department of La Paz. Here, the southern subspecies (*A. alpinus bolivianus*) occurs in Cordillera Oriental (Apurímac and Cusco), Cordilleras Vilcanota and Vilcabamba (Cusco), and Cordillera de Carabaya (Puno)—in Peru—and ranges into Bolivia, where it is found in the Cordillera Real and the Cordillera Apolobamba (La Paz) (BLI 2000, p. 392; BLI 2009a, p. 1; BLI 2009b, p. 1; BLI 2009c, p. 1; BLI 2009d, p. 1; BLI 2009e, p. 1; BLI 2009f, p. 1; BLI 2009k, p. 1; BLI 2009m, p. 1; BLI 2009n, p. 1; BLI 2009o, p. 5; Chutas 2007, p. 8; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, p. 281; Fjeldsá and Krabbe 1990, pp. 468–469; InfoNatura 2007, p. 1). The ash-breasted tit-tyrant was only recently (in 2008) reported in Arequipa Region, Peru (BLI 2009j, p. 1).

The ash-breasted tit-tyrant is highly localized (Collar *et al.* 1992, p. 753) and has been described as "very rare and local, with usually only 1–2 pairs per occupied woodland" (Fjeldsá and Krabbe 1990, p. 469). It exists at such low densities in some places that it goes undetected (Collar *et al.* 1992, p. 753). The species appears to be unable to persist in forest remnants smaller than 2.5 ac (1 ha) (BLI 2009o, p. 1).

Population Estimates

Population information is presented first on the range country level and then in terms of a global population estimate. The range country estimates begin with Peru, where the majority of the population resides.

Local population estimate, Peru: Chutas (2007, p. 8) surveyed five

disjunct *Polylepis* forest patches in Peru and estimated that 461 ash-breasted tit-tyrants were located there. This included 30 birds in Corredor Conchucos (Ancash Region); 181 birds and 33 birds in Cordilleras Vilcanota and Vilcabamba, respectively (Cusco Region); 22 birds in Cordillera de Carabaya (Puno Region); and 195 birds in a study site called "Cordillera del Apurímac" (Apurímac Region) (Chutas 2007, pp. 4, 8), referring to an area within the Runtacocha highlands. Other research in the Runtacocha highlands has indicated that the ash-breasted tit-tyrant is "relatively common" there (BLI 2007, p. 1; BLI 2009o, p. 1), with an estimated 100 pairs of birds found in approximately 40 forest patches (Fjeldsá in litt. 1990, as cited in Collar *et al.* 1992, p. 753). "Small numbers" of birds are reported in La Libertad Region (del Hoyo *et al.* 2004, p. 281). There are no population estimates within Arequipa, Huánuco, or Lima Regions, where the ash-breasted tit-tyrant is also reported to occur (BLI 2009g, p. 1; BLI 2009i, p. 1; BLI 2009j, p. 1; BLI 2009o, p. 1).

Local population estimate, Bolivia: Although BirdLife International reports an estimated population size of 150–300 ash-breasted tit-tyrants in Bolivia (Gómez in litt., 2003 and 2007, as cited in BLI 2009o, p. 1), recent surveys indicate that the estimate is smaller. Over a six-year period, Gómez (in litt. 2007, p. 1) conducted intensive searches throughout 80 percent of the suitable habitat in Bolivia in the Cordillera Real and the Cordillera Apolobamba (La Paz Department), using song playback and point census, to detect the presence of the ash-breasted tit-tyrant. From this work, researchers inferred or observed the presence of 2–10 individuals in each of four forest patches and estimated that approximately 180 ash-breasted tit-tyrants occur in Bolivia.

Global population estimate: BirdLife International, a global organization that consults with and assimilates information from species experts, categorizes the ash-breasted tit-tyrant as having a population size between 250 and 999 individuals, with an estimated actual population size to be in the mid-to upper-hundreds (BLI 2000, p. 392; BLI 2007, p. 1; BLI 2009o, p. 1). Combining the estimated number of ash-breasted tit-tyrants in Peru and Bolivia, the total population consists of at least 641 individuals (Chutas 2007, pp. 4, 8; Gómez in litt. 2007, p. 1), consistent with the BirdLife International category of 250–999 individuals.

However, the total population size, which includes immature individuals, is not an accurate reflection of the species' effective population size (the

number of breeding individuals that contribute to the next generation) (Shaffer 1981, pp. 132-133; Soulé 1980, pp. 160-162). Because the ash-breasted tit-tyrant exists as two subspecies (BLI 2000, p. 392; BLI 2009a, p. 1; BLI 2009b, p. 1; BLI 2009c, p. 1; BLI 2009d, p. 1; BLI 2009e, p. 1; BLI 2009f, p. 1; BLI 2009k, p. 1; BLI 2009m, p. 1; BLI 2009n, p. 1; BLI 2009o, p. 5; Chutas 2007, p. 8; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, p. 281; Fjeldsá and Krabbe 1990, pp. 468-469; InfoNatura 2007, p. 1; ITIS 2009, p. 1), it is reasonable to conclude that there is little or no gene flow between the two subspecies and that the species does not breed as a single population. Therefore, even if all 641 individuals were mature, they would not breed as a single population, such that the species' effective population size is less than 641.

There are also constraints to determining the effective population size on a subspecies level. According to International Union for Conservation of Nature (IUCN) criteria, it is estimated that there are no more than 250 mature individuals in any single subpopulation of the ash-breasted tit-tyrant (IUCN 2001, pp. 8-12). However, the parameters of a subpopulation are not provided in existing research. For instance, while ash-breasted tit-tyrants occupy territories of 2.5–5 ac (1–2 ha) (BLI 2009o, p. 1), there is no information as to the taxon's ability or tendency to travel between territories or forest patches. Research on Bolivian localities indicates that gene flow occurs between some subpopulations, but not all (Gómez 2005, p. 86). In Bolivia, where only 1 subspecies occurs, the birds are distributed in 2 metapopulations, with at least 5 subpopulations in one location and 14 subpopulations in the other (Gómez 2005, p. 86). Peruvian population estimates are incomplete, with no estimates for the ash-breasted tit-tyrants occurring in Arequipa, Huánuco, La Libertad or Lima (BLI 2009g, p. 1; BLI 2009i, p. 1; BLI 2009j, p. 1; BLI 2009o, p. 1; del Hoyo *et al.* 2004, p. 281). Therefore, we can conclude that gene flow occurs at the subspecies level, but there is not sufficient information to determine the extent to which subpopulations interbreed.

The species has experienced a population decline of between 10 and 19 percent in the past 10 years, and this rate of decline is predicted to continue (BLI 2009o, pp. 1, 4). The population is considered to be declining in close association with continued habitat loss and degradation (see Factors A and E) (BLI 2007, pp. 1, 4; BLI 2009o, p. 5).

Conservation Status

The ash-breasted tit-tyrant is considered "endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276855). The IUCN considers the ash-breasted tit-tyrant to be "Endangered" because it has a very small population that is undergoing continued decline in the number of mature individuals and is confined to a habitat that is severely fragmented and is also undergoing a continuing decline in extent, area, and quality (BLI 2008, p. 1; BLI 2009o, p. 4; IUCN 2001, pp. 8-12). The ash-breasted tit-tyrant occurs within the following Peruvian protected areas: Parque Nacional Huascarán, in Ancash, and Santuario Histórico Machu Picchu, in Cusco, and Zona Reservada de la Cordillera Huayhuash, spanning Ancash, Huánuco, and Lima (BLI 2009i, p. 1; BLI 2009l, p. 1; BLI 2009n, p. 1; Chutas *et al.* 2008, p. 16). In La Paz Department, Bolivia, the species is found in Parque Nacional y área Natural de Manejo Integrado Madidi, Parque Nacional y área Natural de Manejo Integrado Cotapata, and the co-located protected areas of Reserva Nacional de Fauna de Apolobamba, área Natural de Manejo Integrado de Apolobamba, and Reserva de la Biosfera de Apolobamba (Auzá and Hennessey 2005, p. 81; BLI 2009a, p. 1; BLI 2009b, p. 1; BLI 2009c, p. 1; BLI 2009e, p. 1; BLI 2009i, p. 1; Chutas *et al.* 2008, p. 16).

Summary of Factors Affecting the Ash-breasted Tit-tyrant

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

The ash-breasted tit-tyrant is dependent upon high-elevation semihumid *Polylepis* or *Polylepis* - *Gynoxys* woodlands (BLI 2000, p. 392; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, pp. 281; Fjeldsá and Krabbe 1990, pp. 468-469; InfoNatura 2007, p. 1). Researchers believe that this habitat was historically contiguous with lower-elevation cloudforests and widespread above 9,843 ft (3,000 m) (Collar *et al.* 1992, p. 753; Fjeldsá 2002a, pp. 111, 115), but *Polylepis* woodlands occur today only between 11,483–16,404 ft (3,500–5,000 m) (Fjeldsá 1992, p. 10). As described above (see Habitat and Life History), the species prefers dense woodlands (Fjeldsá 2002a, p. 114; Smith 1971, p. 269), where the best foraging habitat occurs (De la Via 2004, p. 10), and ash-breasted tit-tyrant occurs at altitudes of 12,139–15,092 ft (3,700–4,600 m) (BLI 2000, p. 392; Collar *et al.* 1992, p. 753; del Hoyo *et al.* 2004, pp.

170, 281; Fjeldså and Krabbe 1990, pp. 468-469; InfoNatura 2007, p. 1).

High-Andean *Polylepis* woodlands are considered by experts to be the most threatened habitat in Peru and Bolivia (Purcell *et al.* 2004, p. 457), throughout the Andean region (BLI 2009a, p. 2), and one of the most threatened woodland ecosystems in the world (Renison *et al.* 2005, as cited in Lloyd 2009, p. 10). The IUCN has listed several *Polylepis* species as "Vulnerable," including two species that occur within the ash-breasted tit-tyrant's range, *Polylepis incana* and *P. pepeí* (WCMC 1998a, p. 1; WCMC 1998b, p. 1). Peruvian and Bolivian *Polylepis* woodlands today are highly fragmented. In the late 1990s, Fjeldså and Kessler (1996, as cited in Fjeldså 2002a, p. 113) conducted comprehensive ground surveys and analyzed maps and satellite images of the area. They estimated that the current range of *Polylepis* woodlands had been reduced from historical levels by 97–98% in Peru and 90 percent in Bolivia. Contemporary *Polylepis* woodlands are dispersed and sparse, covering an estimated area of 386 mi² (1,000 km²) and 1,931 mi² (5,000 km²) in Peru and Bolivia, respectively (Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113). Of the remaining *Polylepis* woodlands, only 1 percent are found in humid areas, where denser *Polylepis* forests tend to occur (Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113). Dense habitat is preferred by the ash-breasted tit-tyrant (BLI 2009o, p. 1; Fjeldså 2002a, p. 114; Lloyd 2008, p. 535; Smith 1971, p. 269).

Habitat loss, conversion, and degradation throughout the ash-breasted tit-tyrant's range have been and continue to occur as a result of ongoing human activity, including (1) clearcutting and burning; (2) extractive activities; (3) human encroachment; and (4) unpredictable climate fluctuations that exacerbate the effects of habitat fragmentation. Clearcutting and burning are among the most destructive activities, and are a leading cause for *Polylepis* habitat loss (WCMC 1998a, p. 1; WCMC 1998b, p. 1). Forested areas are cleared for agriculture and to create pastureland for cattle, sheep, and camels (BLI 2009a, p. 2; BLI 2009c, pp. 1-2; BLI 2009d, pp. 1-2; BLI 2009e, pp. 1, 5; BLI 2009h, p. 1; BLI 2009m, p. 1; BLI 2009n, p. 4). Grazing lands situated amongst remaining forest patches are regularly burned in order to maintain the grassland vegetation (locally known as, "chaqueo"). Regular burning prevents regeneration of native forests and is considered the key factor limiting the distribution of *Polylepis* forests (BLI 2009f, p. 1; BLI 2009n, p. 4; Fjeldså

2002b, p. 8; WCMC 1998a, p. 1; WCMC 1998b, p. 1). In some areas, the burns escape control, causing further habitat destruction (BLI 2009a, p. 2; BLI 2009e, pp. 1, 5). Burning and clearcutting occur throughout the ash-breasted tit-tyrant's range, including in Ancash (BLI 2009h, p. 1), Apurímac (BLI 2009m, p. 1), and Cusco (BLI 2009n, p. 4), in Peru; and La Paz, Bolivia (BLI 2009a, p. 2; BLI 2009c, pp. 1-2; BLI 2009d, pp. 1-2; BLI 2009e, pp. 1, 5). These activities are also ongoing within protected areas, including Parque Nacional Huascarán, Santuario Histórico Machu Picchu, and Zona Reservada de la Cordillera Huayhuash (Barrio 2005, p. 564; BLI 2009l, p. 4; BLI 2009n, p. 2) (see Factor D).

As a result of the intensity of burning and grazing, *Polylepis* species are restricted to areas where fires cannot spread, and where cattle and sheep do not normally roam—in stream ravines and on boulders, rock ledges, and sandy ridges (Fjeldså 2002a, p. 112; Fjeldså 2002b, p. 8). Grazing and trampling by domesticated animals further limit forest regeneration (Fjeldså 2002a, p. 120) and 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 (Engblom *et al.* 2002, p. 56; Purcell *et al.* 2004, p. 458). The loss of nutrient-rich soils leads to habitat degradation, which reduces the ability of the habitat to support dense stands of *Polylepis* woodlands (Fjeldså 2002b, p. 8; Jameson and Ramsay 2007, p. 42; Purcell *et al.* 2004, p. 458).

Ash-breasted tit-tyrant habitat is also subject to conversion, degradation, or destruction caused by extractive activities, such as firewood collection, timber harvest, and mining. Cutting wood for fuel has a consistent and ongoing impact throughout the species' range (BLI 2009a, p. 2; BLI 2009b, pp. 1-2; BLI 2009c, pp. 1-2; BLI 2009d, pp. 1-2; BLI 2009f, p. 1; BLI 2009l, p. 1; WCMC 1998a, p. 1). The high-altitude zones where *Polylepis* occurs have long been inhabited by subsistence farmers, who rely on *Polylepis* wood for firewood and charcoal production (Aucca and Ramsay 2005, p. 287). Many locals manage woodlands for firewood extraction (Engblom *et al.* 2002, p. 56), and community-based *Polylepis* conservation programs fostered by the Peruvian nongovernmental organization Asociación Ecosistemas Andinos (ECOAN) have been under way in Peru and Bolivia since 2004, encompassing Cordilleras Vilcanota and Vilcabamba (Cusco Region), highlands of the Apurímac Region (Aucca and Ramsey 2005, p. 287; ECOAN no date (n.d.), p.

1; Lloyd 2009, p. 10), and in the Ancash Region (MacLennan 2009, p. 2). Known as the "Vilcanota Project" or ECOAN Projects (Aucca and Ramsey 2005, p. 287; ECOAN n.d., p. 1), local communities enter into and enforce management agreements aimed at the primary causes for *Polylepis* deforestation: burning, grazing, and wood-cutting. These projects foster local, sustainable use of resources (Aucca and Ramsay 2005, p. 287; ECOAN n.d., p. 1; Engblom *et al.* 2002, p. 56), such as the use of more fuel-efficient wood-burning stoves that require half the amount of wood fuel (MacLennan 2009, p. 2) (see the Factor A analyses for royal cinclodes and white-browed tit-spinetail for additional examples).

Polylepis wood is also harvested for local commercial use, including within protected areas (BLI 2009a, p. 2; WCMC 1998a, p. 1) (Factor D). At one site, near Abra Málaga (Cusco Region), wood is harvested for sale to local hotels in the towns of Urubamba and Ollantaytambo to support tourism activity (Engblom 2000, p. 1). Engblom (2000, p. 1) documented felling for firewood at this site in Cusco over a 2-day period that significantly reduced the size and quality of the forest patch. Purcell *et al.* (2004, p. 458) noted a positive correlation between habitat destruction and increased demand for (and the concomitant rise in the price of) fuel. *Polylepis* is also harvested for construction, fencing, and tool-making (Aucca and Ramsey 2005, p. 287; BLI 2009a, p. 2). Commercial-scale activities, such as clearcutting, logging, tourism, and infrastructure development, are ongoing throughout this species' range, and alter otherwise sustainable resource use practices (Aucca and Ramsay 2005, p. 287; Engblom 2000, p. 2; Engblom *et al.* 2002, p. 56; MacLennan 2009, p. 2; Purcell and Brelford 2004, pp. 156-157; Purcell *et al.* 2004, pp. 458-459; WCMC 1998a, p. 1). Commercial-scale resource use is exacerbated by ongoing human encroachment, discussed below.

Mining in *Polylepis* habitat occurs in the Peruvian Regions of Ancash and Huánaco and in the Bolivian Department of La Paz (BLI 2009b, p. 1; BLI 2009d, p. 1; BLI 2009g, p. 1). Ancash (Peru) is home to the largest zinc and copper mine "in the world," with a monthly average production rate of 231,485 pounds (105,000 metric tons) of minerals per day and a 186-mile (mi) (300 kilometer (km)) pipeline that stretches from the mine to the port of Punta Lobitos along the coast (Antamina Mine 2006, pp. 4, 9; www.antamina.com/02_operacion/, p.

En_puerto.html). Mining occurs in ash-breasted tit-tyrant habitat in La Paz, Bolivia, where there are active gold, tin, silver, and tungsten mines, in addition to gravel excavation for cement production (USGS Minerals Yearbook 2005, pp. 4-7). Antamina Mine has undertaken habitat conservation programs within the areas affected by mineral extraction, similar to the ECOAN *Polylepis* conservation programs, investing millions of dollars in programs ranging from education and tourism, to organic agriculture and sustainable development. However, tourism has had negative effects in other areas where the ash-breasted tit-tyrant occurs, including Ancash, Huánuco, and Lima, Peru, and La Paz, Bolivia (Barrio 2005, p. 564; BLI 2009e, p. 5) (see below). The Antamina Mining Company conservation program also supports the planned reforestation of 123,552 ac (50,000 ha) of *Polylepis* forest that will connect two protected areas, Parque Nacional Huascarán and Zona Reservada de la Cordillera Huayhuash (Antamina Mine 2006, p. 5). To date, the project has succeeded in restoring 371 ac (150 ha) of forest, with a 95 percent survival rate (MacLennan 2009, p. 1). Known as Corredor Conchucos, at least 30 ash-breasted tit-tyrants have recently been observed there (Chutas 2007, p. 8).

Human encroachment and concomitant increasing human population pressures exacerbate the destructive effects of ongoing human activities throughout the ash-breasted tit-tyrant's habitat. Mining and hydroelectric projects open new areas to exploitation and attract people seeking employment (Purcell *et al.* 2004, p. 458). Increased urbanization and mining have led to increased infrastructure development. Road building and mining projects further facilitate human access to remaining *Polylepis* forest fragments, throughout the ash-breasted tit-tyrant's range (Purcell *et al.* 2004, pp. 458-459; Purcell and Brelford, 2004, pp. 156-157), including protected areas. In the Bolivian Department of La Paz, one of the most transited highways in the country is located a short distance from the Parque Nacional y área Natural de Manejo Integrado Cotapata (BLI 2009b, p. 2) (see Factor D). Road building, mining, and other large-scale resource exploitations are considered to have major impacts on the habitat, as compared to exploitation by local communities (Purcell and Brelford 2004, p. 157).

Ecotourism within protected areas where the ash-breasted tit-tyrant occurs (such as in the Zona Reservada de la Cordillera Huayhuash in Peru, and in

the Apolobamba protected areas in Bolivia) is considered a growing problem (Barrio 2005, p. 564; BLI 2009e, p. 5) (see Factor D). In the Department of La Paz, Bolivia, which encompasses Bolivia's largest urban area, most of the *Polylepis* forest had been eliminated prior to the late 1990s (Purcell and Brelford 2004, p. 157). Recently, an accelerated rate of *Polylepis* forest destruction has been attributed to clearcutting for road building and industrialization projects, such as mining and construction of hydroelectric power stations (Purcell and Brelford 2004, pp. 156-157). Between 1991 and 2003, approximately 494 ac (200 ha) of *Polylepis* habitat was destroyed. Thus, nearly two-thirds of the forest cover that existed in the 1990s no longer existed in 2003 (Purcell and Brelford 2004, p. 155). With this research, it was estimated that only 1,285 ac (520 ha) of *Polylepis* forest remain in the Bolivian Department of La Paz, representing approximately a 40 percent rate of habitat loss in just over one decade. The researchers inferred that this rate of destruction could result in extirpation of the remaining *Polylepis* forest in La Paz within the next 30 years (Purcell and Brelford 2004, pp. 157).

Larger concentrations of people put greater demand on the natural resources in the area. Increasing demand for firewood upsets informal and otherwise sustainable community-based forest management traditions (Purcell and Brelford, 2004, p. 157). Increasing human populations in the high-Andes of Bolivia and Peru have also resulted in a scarcity of arable land. This has led many farmers to burn down additional patches of *Polylepis* forests to plant crops, even on steep hillsides not suitable for cultivation (BLI 2009b, p. 2; BLI 2009h, p. 1; Hensen 2002, p. 199). These ongoing farming practices result in the rapid loss of *Polylepis* forests from Bolivia to Peru.

Habitat destruction is often caused by a combination of human activities that promote habitat degradation. In the Cordillera de Vilcanota (Cusco, Peru), where an estimated 181 birds are found (Chutas 2007, pp. 4, 8), the rate of habitat loss was studied by comparing forest cover between 1956 and 2005. This study revealed a rate of habitat loss averaging only 1 percent. However, remaining patches of *Polylepis* woodland were small, with a mean patch size of 7.4 ac (3 ha); four forest patches had disappeared completely; and no new patches were located within the study area (Jameson and Ramsay 2007, p. 42). Lloyd (2008, p. 532) studied bird foraging habits at three *Polylepis* woodland sites in the

Cordillera Vilcanota during 2003–2005. The sites were described as highly fragmented, consisting of many small remnant patches (less than 2.5 ac (1 ha)) and scattered trees, separated from larger woodland tracts (greater than 25 ac (10 ha)) by distances of 98–4,921 ft (30–1,500 m) (Lloyd and Marsden in press, as cited in Lloyd 2008, p. 532). Given that the species territory ranges from 2.5–5 ac (1–2 ha) and that the ash-breasted tit-tyrant appears to be unable to persist in forest remnants smaller than 2.5 ac (1 ha) (BLI 2009o, p. 1), these patch sizes have met or are approaching the lower threshold of the species' ecological requirements. Moreover, 10 percent of the remaining forest patches showed a decline in forest density over this time-period.

Thus, habitat degradation also has serious impacts in *Polylepis* woodlands (Jameson and Ramsay 2007, p. 42), especially given the species' preference for dense woodlands (Fjeldsá 2002a, p. 114; Smith 1971, p. 269). The fact that no new *Polylepis* forest patches had become established between 1956 and 2005 underscores the long-term ramifications of ongoing burning, clearing, grazing, and other habitat-altering human activities that are pervasive throughout the ash-breasted tit-tyrant's range (BLI 2009f, p. 1; BLI 2009n, p. 4; Fjeldsá 2002b, p. 8; WCMC 1998a, p. 1; WCMC 1998b, p. 1). These activities are considered to be key factors both in preventing regeneration of *Polylepis* woodlands (Fjeldsá 2002a, p. 112, 120; Fjeldsá 2002b, p. 8) and in the historical decline of *Polylepis*-dependent bird species, including the ash-breasted tit-tyrant (Fjeldsá 2002a, p. 116). Researchers consider the species' population to be declining in close association with the continued habitat loss and degradation (BLI 2007, pp. 1, 4; BLI 2008, p. 1; BLI 2009o, p. 1). Therefore, further habitat loss will continue to impact the species' already small population size (see Factor E).

Peru is subject to unpredictable climate fluctuations that exacerbate the effects of habitat fragmentation, such as those that are related to the El Niño Southern Oscillation (ENSO). According to the U.S. Dept. of Commerce–National Oceanic and Atmospheric Administration (NOAA), the term ENSO refers to a range of variability associated with the southern trade winds in the eastern and central equatorial Pacific Ocean. El Niño events are characterized by unusual warming of the ocean, while La Niña events bring cooler ocean temperatures (Tropical Atmosphere Ocean (TAO) Project no date (n.d.), p. 1). Generally speaking, extreme ENSO events alter weather patterns, so that

precipitation increases in normally dry areas, and decreases in normally wet areas. During an El Niño event, rainfall dramatically increases, whereas a La Niña event brings near-drought conditions (Holmgren *et al.* 2001, p. 89).

If we consider that wildlife habitat is not only defined by substrates (vegetation, soil, water), but also atmospheric conditions, then changes in air temperature and moisture can effectively change a species' habitat. Climate change is characterized by variations in the earth's temperature and precipitation, causing changes in atmospheric, oceanic, and terrestrial conditions (Parmesan and Mathews 2005, p. 334). Global climate change and other periodic climatic patterns (e.g., El Niño and La Niña) can cause or exacerbate such negative impacts on a broad range of terrestrial ecosystems and Neotropical bird populations (England 2000, p. 86; Holmgren *et al.* 2001, p. 89; Plumart 2007, pp. 1-2; Timmermann 1999, p. 694).

Unpredictable climate fluctuations negatively impact populations undergoing habitat fragmentation. In the face of an unpredictable climate, the risk of population decline due to habitat fragmentation is heightened. Mora *et al.* (2007, p. 1027) found that the combined effects of habitat fragmentation and climate change (in this case, warming) had a synergistic effect, rather than additive. In other words, the interactive effects of both climate fluctuation and habitat fragmentation led to a greater population decline than if either climate change or habitat fragmentation were acting alone on populations. Jetz *et al.* (2007, p. 1211) investigated the effects of climate change on 8,750 land bird species, including the ash-breasted tit-tyrant, that are exposed to ongoing manmade land cover changes (i.e., habitat loss). They determined that a narrow endemic, such as the ash-breasted tit-tyrant, is likely to suffer greater impacts from climate change, especially where projected range contractions are driven by manmade land conversion activities (Jetz *et al.* 2007, p. 1213). This is due to the species' already small population size, specialized habitat requirements, and heightened risk of extinction from stochastic demographic processes (see also Factor E). According to this study, by 2050, up to 18 percent of the ash-breasted tit-tyrant's current remaining range is likely to be unsuitable for this species due to climate change; and, by 2100, it is predicted that about 18 to 42 percent of the species' range is likely to be lost as a direct result of global climate change (Jetz *et al.* 2007, Supplementary Table 2, p. 73).

In 1999, Timmermann (1999, p. 694) predicted that climate change events would increase the periodicity and severity of droughts and excessive rainfalls, such as those caused by El Niño and La Niña events. Evidence suggests that this is the case in Peru (Richter 2005, p. 24-25). Over the past decade, there have been four El Niño events (1997-1998, 2002-2003, 2004-2005, and 2006-2007) and three La Niña events (1998-2000, 2000-2001, and 2007-2008) (National Weather Service (NWS) 2009, p. 2). In Peru, the Andean highlands, and *Polylepis* species in particular, are strongly influenced by ENSO events (Christie *et al.* 2008, p. 1). Christie *et al.* (2008, p. 1) found that tree growth in *P. tarapacana* is highly influenced by ENSO events because ENSO cycles on the Peruvian Coast are strongest during the growing season (December-February). ENSO-related droughts can increase tree mortality and dramatically alter age structure within tree populations, especially where woodlands have undergone disturbance, such as fire and grazing (Villalba and Veblen 1997, pp. 121-123; Villalba and Veblen 1998, pp. 2624, 2637).

With years of extremely high rainfall followed by years of extremely dry weather (Block and Richter 2007, p. 1), fire hazard is increased from the biomass accumulated during the wet period that dries and adds to the fuel load in the dry season (Block and Richter 2007, p. 1; Power *et al.* 2007, p. 898). Evidence suggests that the fire cycle in Peru has shortened, particularly in coastal Peru and west of the Andes (Power *et al.* 2007, pp. 897-898). Changes in the fire-regime can have broad ecological consequences (Block and Richter 2007, p. 1; Power *et al.* 2007, p. 898). In the case of the ash-breasted tit-tyrant, burning is considered to be a key factor preventing *Polylepis* regeneration (Fjeldså 2002a, p. 112, 120; Fjeldså 2002b, p. 8). Research in Ecuadorian *Polylepis - Gynoxys* mixed woodlands indicated a strong reduction in *P. incana* adult and seedling survival following a single fire. This indicates that the species does not recover well from even a single fire event (Cierjacks *et al.* 2007, p. 176). An accelerated fire cycle would exacerbate this situation.

Activities that destroy and alter habitat are ongoing within protected areas. This is further discussed under Factor D.

Summary of Factor A

The ash-breasted tit-tyrant is dependent on *Polylepis* habitat, with a preference for dense woodlands. *Polylepis* habitat throughout the ash-

breasted tit-tyrant's range has been and continues to be altered and destroyed as a result of human activities, including clearcutting and burning for agriculture, grazing lands, tourism, and industrialization; extractive activities, including firewood, timber, and mineral extraction; and human encroachment and concomitant increased pressure on natural resources. Researchers estimate that 1 percent of the dense woodlands preferred by the species remains, and that all remaining habitat is fragmented and degraded. The ash-breasted tit-tyrant currently occupies an area of approximately 4,595 mi² (11,900 km²) and appears to be unable to persist in forest remnants smaller than 2.5 ac (1 ha). Forest fragments in some portions of the ash-breasted tit-tyrant's range are approaching the lower threshold of the species' ecological requirements. The historical decline of habitat suitable for this species is attributed to the same human activities that are causing habitat loss today. Ongoing and accelerated habitat destruction of the remaining *Polylepis* forest fragments in Peru and Bolivia continues to reduce the quantity, quality, distribution, and regeneration of remaining patches. Human activities that degrade, alter, and destroy habitat are ongoing throughout the species' range, including within protected areas (see also Factor D). Current research indicates that climate fluctuations exacerbate the risks to species that are already undergoing range reduction due to habitat alteration. Climate models predict that this species' habitat will continue to decline. Experts consider the species' population decline to be commensurate with the declining habitat (Factor E). Therefore, we find that destruction and modification of habitat are threats to the continued existence of the ash-breasted tit-tyrant throughout its range.

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

We are not aware of any information currently available that indicates that overutilization of the species for commercial, recreational, scientific, or educational purposes has occurred or is occurring at this time. As a result, we are not considering overutilization to be a threat to the continued existence of the ash-breasted tit-tyrant.

C. Disease or Predation

We are not aware of any scientific or commercial information that indicate disease or predation poses a threat to this species. As a result, we are not considering disease or predation to be a

threat to the continued existence of the ash-breasted tit-tyrant.

D. Inadequacy of Existing Regulatory Mechanisms

This analysis of regulatory mechanisms is discussed on a country-by-country basis, beginning with Peru.

Peru: The ash-breasted tit-tyrant is considered “endangered” by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276855). This Decree prohibits hunting, take, transport, and trade of protected species, except as permitted by regulation. As hunting, take, transport, and trade are not currently threats to the ash-breasted tit-tyrant (Factor B), this regulation does not mitigate any current threats to this species.

The Peruvian national protected area system includes several categories of habitat protection. Habitat may be designated as any of the following: (1) Parque Nacional (National Park, an area managed mainly for ecosystem conservation and recreation); (2) Santuario (Sanctuary, for the preservation of sites of notable natural or historical importance); (3) Reserva Nacional (National Reserve, for sustainable extraction of certain biological resources); (4) Bosque de Protección (Protection Forest, to safeguard soils and forests, especially for watershed conservation); (5) Zona Reservada (Reserved Zone, for temporary protection while further study is under way to determine their importance); (6) Bosque Nacional (National Forest, to be managed for utilization); (7) Reserva Comunal (Communal Reserve, for local area use and management, with national oversight); and (8) Cotos de Caza (Hunting Reserve, for local use and management, with national oversight) (BLI 2008, p. 1; Rodríguez and Young 2000, p. 330). National reserves, national forests, communal reserves, and hunting reserves are managed for the sustainable use of resources (IUCN 1994, p. 2). The designations of National Parks, Sanctuaries, and Protection Forests, are established by supreme decree that supersedes all other legal claim to the land and, thus, these areas tend to provide more habitat protection. All other protected areas are established by supreme resolution, which is viewed as a less powerful form of protection (Rodríguez and Young 2000, p. 330).

Protected areas have been established through regulation in at least three sites occupied by the ash-breasted tit-tyrant in Peru: Parque Nacional Huascarán (Ancash), and Santuario Histórico Machu Picchu (Cusco); and Zona Reservada de la Cordillera Huayhuash

(spanning Ancash, Huánuco, and Lima) (Barrio 2005, p. 563; BLI 2009i, p. 1; BLI 2009l, p. 1; BLI 2009n, p. 1). Habitat destruction and alteration, including burning, cutting, and grazing are ongoing within Parque Nacional Huascarán and Santuario Histórico Machu Picchu (BLI 2009l, p. 4; BLI 2009n, p. 2; Engblom *et al.* 2002, p. 58), where resources are supposed to be managed for conservation (Rodríguez and Young 2000, p. 330). Reserved zones are intended to be protected pending further study (Rodríguez and Young 2000, p. 330). However, burning for habitat conversion and maintenance of pastures for grazing and increasing ecotourism are ongoing within Zona Reservada de la Cordillera Huayhuash (Barrio 2005, p. 564). Therefore, the occurrence of the ash-breasted tit-tyrant within protected areas in Peru does not protect the species, nor does it mitigate the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

Recent studies by the Peruvian Society for Environmental Law (SPDA) have concluded that there are approximately 5,000 laws and regulations directly or indirectly related to environmental protection and natural resource conservation in Peru. However, many of these are not effective because of limited implementation and/or enforcement capability (Muller 2001, pp. 1-2).

Bolivia: The 1975 Law on Wildlife, National Parks, Hunting and Fishing (Decree Law No. 12,301 1975, pp. 1-34) has the fundamental objective of protecting the country’s natural resources. This law governs the protection, management, utilization, transportation, and selling of wildlife and their products; the protection of endangered species; habitat conservation of fauna and flora; and the declaration of national parks, biological reserves, refuges, and wildlife sanctuaries, tending to the preservation, promotion, and rational use of these resources (Decree Law No. 12,301 1975, pp. 1-34; eLAW 2003, p. 2). Although this law designates national protection for all wildlife, there is no information as to the actual protections this confers to ash-breasted tit-tyrants. Law No. 12,301 (1975, pp. 1-34) also placed into public trust all national parks, reserves, refuges, and wildlife sanctuaries. However, there is no specific information as to the governmental protections afforded within the protected areas to either the ash-breasted tit-tyrant or its habitat. Given the ongoing habitat destruction throughout the species’ range in Bolivia, this law does not protect the species,

nor does it mitigate the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

Bolivia passed an overarching environmental law in 1992 (Law No. 1,333 1992), with the intent of protecting and conserving the environment and natural resources. However, there is no specific legislation to implement these laws (eLAW 2003, p. 1). Therefore, we cannot determine that this law protects the species or mitigates the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

In Bolivia, habitat is protected either on the national or departmental level through the following designations: (1) Parque (Park, for strict and permanent protection of representative of ecosystems and provincial habitats, as well as plant and animal resources, along with the geographical, scenic and natural landscapes that contain them); (2) Santuario (Sanctuary, for the strict and permanent protection of sites that house endemic plants and animals that are threatened or in danger of extinction); (3) Monumento Natural (Natural Monument, to preserve areas such as those with distinctive natural landscapes or geologic formations, and to conserve the biological diversity contained therein); (4) Reserva de Vida Silvestre (Wildlife Reserve, for protection, management, sustainable use and monitoring of wildlife); (5) Área Natural de Manejo Integrado (Natural Area of Integrated Management, where conservation of biological diversity is balanced with sustainable development of the local population; and (6) Reserva Natural de Inmovilización (“Immobilized” Natural Reserve, a temporary (5–year) designation for an area that requires further research before any official designations can be made and during which time no natural resource concessions can be made within the area) (Supreme Decree No. 24,781 1997, p. 3). Within parks, sanctuaries and natural monuments, extraction or consumption of all resources are prohibited, except for “scientific research, eco-tourism, environmental education, and activities of subsistence of original towns, properly described and authorized.” National protected areas are under the management of the national government, while departmental protected areas are managed at the department level (eLAW 2003, p. 3; Supreme Decree No. 24,781 1997, p. 3).

The ash-breasted tit-tyrant occurs within several protected areas in the Department of La Paz, Bolivia: Parque Nacional y área Natural de Manejo

Integrado Madidi, Parque Nacional y área Natural de Manejo Integrado Cotapata, and the co-located protected areas of Reserva Nacional de Fauna de Apolobamba, área Natural de Manejo Integrado de Apolobamba, and Reserva de la Biosfera de Apolobamba (Auza and Hennessey 2005, p. 81; BLI 2009a, p. 1; BLI 2009b, p. 1; BLI 2009c, p. 1; BLI 2009e, p. 1). Although national parks are intended to be strictly protected, the two parks in which the species occurs (Parque Nacional y área Natural de Manejo Integrado Madidi and Parque Nacional y área Natural de Manejo Integrado Cotapata) are also designated as areas of integrated management, which are managed for the biological conservation balanced with the sustainable development of the local population (Supreme Decree No. 24,781 1997, p. 3). Grazing and firewood extraction are ongoing within Parque Nacional y área Natural de Manejo Integrado Cotapata (BLI 2009b, p. 2; BLI 2009c, p. 2). Commercial logging has occurred within Parque Nacional y área Natural de Manejo Integrado Madidi (BLI 2009a, p. 2; WCMC 1998a, p. 1). Uncontrolled clearing, extensive agriculture, grazing, and “irresponsible” tourism are ongoing within the Apolobamba protected areas (Auza and Hennessey 2005, p. 81; BLI 2009e, p. 5). Habitat degradation and destruction from grazing, forest fires, and timber extraction are ongoing in other protected areas, such as Tunari National Park (Department of Cochabamba, Bolivia), where suitable habitat exists for this species (De la Vie 2004, p. 7). Therefore, the occurrence of the ash-breasted tit-tyrant within protected areas in Bolivia does not protect the species, nor does it mitigate the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

Summary of Factor D

Peru and Bolivia have enacted various laws and regulatory mechanisms to protect and manage wildlife and their habitats. The ash-breasted tit-tyrant is “endangered” under Peruvian law and occurs within several protected areas in Peru and Bolivia. As discussed under Factor A, the ash-breasted tit-tyrant prefers dense woodlands, which have been reduced by an estimated 99 percent in Peru and Bolivia, and the remaining habitat is fragmented and degraded. Habitat throughout the species’ range has been and continues to be altered as a result of human activities, including clearcutting and burning for agriculture, grazing lands, and industrialization; extractive activities, including, firewood, timber,

and mineral extraction; and human encroachment and concomitant increased pressure on natural resources. Despite the species’ “endangered” status in Peru and Bolivian laws intended to protect all wildlife and natural resources, these activities are ongoing within protected areas, indicating that the laws governing wildlife and habitat protection in both countries are either inadequate or inadequately enforced to protect the species or to mitigate ongoing habitat loss (Factor A) and population declines (Factor E). Therefore, we find that the existing regulatory mechanisms are inadequate to mitigate the current threats to the continued existence of the ash-breasted tit-tyrant throughout its range.

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

An additional factor that affects the continued existence of the ash-breasted tit-tyrant is the species’ small population size. As discussed above (see Population Estimates), the global population estimate is not an accurate reflection of the species’ effective population size because gene flow does not occur between the subspecies. At the same time, there is insufficient information on the subspecies or subpopulation level (in terms of numbers of individuals and breeding structure) to estimate the effective population size at the subspecies level. However, with an estimated global population size in the mid- to upper-hundreds (BLI 2000, p. 392; BLI 2007, p. 1; BLI 2009o, p. 1), and the most recent estimate of 641 individuals (Chutas 2007, pp. 4, 8; Gómez in litt. 2007, p. 1), the ash-breasted tit-tyrant is considered to have a “very small population” size (BLI 2000, p. 392; BLI 2008, p. 1; BLI 2009o, p. 1).

Small population size renders a species vulnerable to any of several risks, including inbreeding depression, loss of genetic variation, and accumulation of new mutations. Inbreeding can have individual or population-level consequences either by increasing the phenotypic expression (the outward appearance or observable structure, function, or behavior of a living organism) of recessive, deleterious alleles (harmful gene sequences) or by reducing the overall fitness of individuals in the population (Charlesworth and Charlesworth 1987, p. 231; Shaffer 1981, p. 131). Small, isolated wildlife populations are also more susceptible to environmental fluctuations and demographic shifts (Pimm *et al.* 1988, pp. 757, 773-775;

Shaffer 1981, p. 131), such as reduced reproductive success of individuals and chance disequilibrium of sex ratios. Species tend to have a higher risk of extinction if they occupy a small geographic range and occur at low density (Purvis *et al.* 2000, p. 1949).

The ash-breasted tit-tyrant population declined at a rate between 10 and 19 percent in the past 10 years, and this decline is expected to continue in close association with continued habitat loss and degradation (see Factor A) (BLI 2009o, p. 1). Extinction risk is heightened in small, declining populations by an increased vulnerability to the loss of genetic variation due to inbreeding depression and genetic drift (changes in relative frequency of gene sequences). This, in turn, compromises a species’ ability to adapt genetically to changing environments (Frankham 1996, p. 1507) and reduces fitness, thus increasing extinction risk (Reed and Frankham 2003, pp. 233-234). Once a population is reduced below a certain number of individuals, it tends to rapidly decline towards extinction (Franklin 1980, pp. 147-148; Gilpin and Soulé 1986, p. 25; Holsinger 2000, pp. 64-65; Soulé 1987, p. 181).

Complications arising from the species’ small population size are exacerbated by the species’ fragmented distribution. The ash-breasted tit-tyrant is currently confined to restricted and severely fragmented forest patches in the high Andes of Peru and Bolivia (BLI 2000, p. 392; BLI 2007, pp. 1; BLI 2008, p. 1; BLI 2009o, p. 1; Collar *et al.* 1992, p. 753; Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113; del Hoyo *et al.* 2004, p. 281; InfoNatura 2007, p. 1), where it is estimated that only 1 percent of the dense woodlands preferred by the species remain (Fjeldså 2002a, p. 114; Smith 1971, p. 269) (see Habitat and Life History). Habitat fragmentation can cause genetic isolation and heighten the risks to the species associated with short-term genetic viability. Species with a small population size, combined with a restricted and severely fragmented range, are exposed to increased vulnerability to adverse natural events and manmade activities that destroy individuals and their habitat (Holsinger 2000, pp. 64-65; Primack 1998, pp. 279-308; Young and Clarke 2000, pp. 361-366).

Summary of Factor E

The ash-breasted tit-tyrant has a small population size that renders it vulnerable to genetic risks that negatively impact the species’ viability. The species occurs in two disjunct

populations, where habitat is highly fragmented and continues to be altered by human activities (Factor A). The restricted and fragmented range, as well as its small population size, increases the species' vulnerability to extinction, through demographic or environmental fluctuations. Based on its small population size and fragmented distribution, we have determined that the ash-breasted tit-tyrant is particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., deforestation, habitat alteration, and infrastructure development) that destroy individuals and their habitat. The genetic and demographic risks associated with small population sizes are exacerbated by ongoing human activities that continue to curtail the species' habitat throughout its range. The species' population has declined and is predicted to continue declining commensurate with ongoing habitat loss (Factor A). Therefore, we find that the species' small population size, in concert with its fragmented distribution and its heightened vulnerability to adverse natural events and manmade activities, are threats to the continued existence of the ash-breasted tit-tyrant throughout its range.

Status Determination for the Ash-Breasted Tit-Tyrant

The ash-breasted tit-tyrant, a small New World tyrant flycatcher, exists in two disjunct areas in Peru and Bolivia. Preferring dense, semihumid *Polylepis* or *Polylepis*-mixed woodlands, the ash-breasted tit-tyrant occupies a narrow range of distribution, at elevations between 12,139 and 15,092 ft (3,700 and 4,600 m). The species has a highly restricted and severely fragmented range (approximately 4,595 mi² (11,900 km²)), and is known only in two disjunct areas: one location in west-central Peru (in the Peruvian Administrative Regions of Ancash, Huánuco, La Libertad, and Lima) and another location ranging from southern Peru (Apurímac, Cusco, Puno, and Arequipa Regions) into northern Bolivia (in the Department of La Paz). The known population of the ash-breasted tit-tyrant is estimated to be 641 individuals.

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the ash-breasted tit-tyrant and have concluded that there are three primary factors that threaten the continued existence of the ash-breasted tit-tyrant: (1) Habitat destruction, fragmentation, and degradation; (2) limited size and

isolation of remaining populations; and (3) inadequate regulatory mechanisms.

Human activities that degrade, alter, and destroy habitat are ongoing throughout the ash-breasted tit-tyrant's range. Widespread deforestation and the conversion of forests for grazing, agriculture, and human settlement have led to the fragmentation and degradation of habitat throughout the range of the ash-breasted tit-tyrant (Factor A). Researchers estimate that only 1 percent of the dense *Polylepis* woodlands preferred by the species remain extant. Limited by the availability of suitable habitat, the species occurs today only in some of these fragmented and disjunct locations. Ash-breasted tit-tyrant habitat continues to be altered by human activities, such as burning, grazing, extractive activities, and human encroachment, which result in the continued degradation, conversion, and destruction of habitat and reduce the quantity, quality, distribution, and regeneration of remaining forest patches.

The ash-breasted tit-tyrant population is small, rendering the species particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., deforestation and habitat alteration) that destroy individuals and their habitat. Ongoing human activities that curtail the species' habitat throughout its range exacerbate the genetic and demographic risks associated with small population sizes (Factor E). The population has declined 10–19 percent in the past 10 years (see Population Estimates), and is predicted to continue declining commensurate with ongoing habitat loss (Factor A). Habitat loss was a factor in the ash-breasted tit-tyrant's historical population decline (see Historical Range and Distribution), and the species is considered to be declining today in association with the continued reduction in habitat (Factors A and E). Moreover, current research indicates that narrow endemics, such as the ash-breasted tit-tyrant, are especially susceptible to climate fluctuations, because of the synergistic effect these fluctuations have on declining populations that are also experiencing range reductions due to human activities (Factor A).

Despite the species' "endangered" status in Peru and its occurrence within several protected areas in Peru and Bolivia (Factor D), human activities that degrade, alter, and destroy habitat are ongoing throughout the species' range, including within protected areas. Therefore, regulatory mechanisms are either inadequate or ineffective at

curbing the threats to the ash-breasted tit-tyrant of habitat loss (Factor A) and corresponding population decline (Factor E).

Section 3 of the Act defines an "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range" and a "threatened species" as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Based on the immediate and ongoing threats to the ash-breasted tit-tyrant throughout its entire range, as described above, we determine that the ash-breasted tit-tyrant is in danger of extinction throughout all of its range. Therefore, on the basis of the best available scientific and commercial information, we are proposing to list the ash-breasted tit-tyrant as an endangered species throughout all of its range.

II. Junín grebe (*Podiceps taczanowskii*)

Species Description

The Junín grebe is a highly social, flightless water bird in the *Podicipedidae* family that is endemic to a single location (Lake Junín) in Peru. Other common names for the species (in English) are: Junín flightless grebe, puna grebe, and Taczanowski's grebe. This species is also known by two Spanish names: "*zampullín del Junín*" or "*zambullidor de Junín*" (del Hoyo *et al.* 1992, p. 195; Fjeldså 2004, p. 199; Instituto Nacional de Recursos Naturales (INRENA) 1996, p. 3; Ramsen *et al.* 2007, p. 18; Supreme Decree 034-2004-AG 2004, p. 276854).

A slim, long-necked bird, the Junín grebe is about 13.78 in (35 cm) in length, and its weight ranges from 0.66 to 1.04 pounds (0.30 to 0.47 kilograms) (BLI 2009b, p. 1; UNEP-WCMC 2009, p. 1). The Junín grebe has a pointed head, with dark feathers on its back, a white throat, and mottled, dusky-colored underparts. This grebe is distinguished by its slender gray bill, red iris, and dull yellow-orange colored feet. Immature birds are darker gray on the flanks than mature birds (BLI 2009b, p. 1).

Taxonomy

The Junín grebe was taxonomically described by Berlepsch and Stolzmann in 1894 (ITIS 2009, p. 1). It is one of nine species of grebes in the genus *Podiceps* worldwide (Dickinson 2003, p.80). The species' taxonomic status as *Podiceps taczanowskii* is valid (ITIS 2009, p. 1).

Habitat and Life History

The Junín grebe is endemic to the open waters and marshlands of Lake Junín, located at 13,390 ft (4,080 m) above sea level in the Peruvian Administrative Region of Junín (BLI 2003, p. 1; BLI 2009b, p. 1). The 57-mi² (147-km²) lake, also known as “Chinchaycocha” or “Lago de Junín,” is large but fairly shallow (BLI 2003, p. 1; BLI 2009a, p. 1; BLI 2009b, p. 1; ParksWatch 2009, p. 1; Tello 2007, p. 1). Situated within “puna” habitat, the climate is seasonal and can be “bitterly cold” in the dry season (Fjeldsá 1981, p. 240). Local vegetation is characterized by tall dense grasslands and scrubland with open, rocky areas, all interspersed with wetlands and woodlands (BLI 2003, p. 1; ParksWatch 2009, pp. 1, 4). The dominant terrestrial plant species surrounding the lake includes 43 species of grass (*Poaceae* family), 15 species of asters (*Asteraceae* family), and 10 species of legumes (*Fabaceae* family) (ParksWatch 2009, p. 1). Aquatic vegetation includes Andean watermilfoil (*Myriophyllum quitense*), several species of pondweed (including *Elodea potamogeton*, *Potamogeton ferrugineus*, and *P. filliformis*), and bladderwort (*Utricularia* spp.). Floating plants, such as duckweed (*Lemna* species (spp.)), large duckweed (*Spiodela* spp.), and water fern (*Azolla filiculoides*), also occur on the lake (ParksWatch 2009, p. 2). The Lake is surrounded by extensive marshland along the lake shore (BLI 2009a, p. 1; BLI 2009b, p. 1) that extends into the lake up to 1–3 mi (2–5 km) from shore (O’Donnel and Fjeldsá 1997, p. 29). The marshes are dominated by two robust species of cattails, giant bulrush (*Schoenoplectus californicus* var. *totara*) and totorilla (*Juncus articus* var. *andicola*) (Fjeldsá 1981, pp. 244, 246). Both cattail species can reach nearly 6.6 ft (2 m) in height. These plant communities, or “*tortoras*,” grow so densely that stands are often impenetrable (ParksWatch 2009, p. 1). In shallow water, during low lake levels, “*tortora*” communities can become partially or completely dry (BLI 2009b, p. 1; ParksWatch 2009, p. 2).

Lake Junín supports one of the richest and most diverse arrays of bird species of all Peruvian high Andean wetlands (ParksWatch 2009, p. 3). These bird species include migratory birds, birds that nest at high altitude, aquatic birds, and local endemic species, such as the Junín grebe and the Junín rail (*Laterallus tuerosi*; also the subject of this proposed rule), the giant coot (*Fulica ardesiaca*), and the Chilean flamingo (*Phoenicopterus chilensis*)

(BLI 2009a, pp. 2-3; ParksWatch 2009, p. 3; Tello 2007, p. 2). Mammals are relatively scarce in the area, although there are some predators (ParksWatch 2009, p. 4) (see Factor C).

Breeding season for this species occurs annually from November to March (Fjeldsá 1981, pp. 44, 246; O’Donnel and Fjeldsá 1997, p. 29). The Junín grebe nests in the protective cover of the marshlands during the breeding season (Fjeldsá 1981, p. 247; Tello 2007, p. 3), particularly in stands of giant bulrush (ParksWatch 2009, p. 4). Under natural conditions, winter rains increase the lake water level during the breeding season, allowing the grebes to venture into local bays and canals, although they are never found nesting on the lake’s shore (Tello 2007, p. 3). The species nests in the giant bulrush marshlands (ParksWatch 2009, p. 4). Well-hidden floating nests can contain up to three eggs, with an average of two eggs, laid during November and December (Fjeldsá 1981, p. 245). The species is believed to have a deferred sexual maturation (Fjeldsá 2004, p. 201) and exhibits low breeding potential, perhaps as a reflection to adaptation to a “highly predictable, stable environment” (del Hoyo *et al.* 1992, p. 195), laying one clutch during the breeding season (ParksWatch 2009, p. 4). Junín grebes occasionally produce a replacement clutch if their original nest is disturbed (Fjeldsá 2004, pp. 199, 201). After the eggs hatch, the male grebe cares for the chicks, and does not leave the nest to feed. The female grebe is responsible for feeding the male and chicks until the chicks can leave the nest (Tello 2007, p. 3). The Junín grebe is likely a long-lived species (Fjeldsá 2004, p. 201), and its breeding success and population size are highly influenced by the climate (BLI 2008, pp. 1, 3-4; BLI 2009b, p. 2; Elton 2000, p. 3; Fjeldsá 2004, p. 200; Hirshfeld 2007, p. 107) (see Factor A).

The Junín grebe feeds in the open waters of the lake and around the marsh edges, moving into the open waters of the lake to feed where it is easier to dive for food during the winter (Fjeldsá 1981, pp. 247-248; Tello 2007, p. 3). Fish (primarily pupfish (*Orestias* spp.)) account for over 90 percent of the grebe’s diet (Fjeldsá 1981, pp. 251-252). Pupfish become scarce when the marshlands dry during periods of reduced water levels, and the Junín grebe is then known to vary its diet with midges (Order *Diptera*), corixid bugs (*Trichocorixa reticulata*), amphipods (*Hyalella simplex*), and shore fly maggots and pupa (*Ephydriid* spp.).

Historical Range and Distribution

The Junín grebe was historically known to be endemic to Lake Junín, in the Peruvian Administrative Region of Junín (Fjeldsá 1981, p. 238; Fjeldsá 2004, p. 200; Fjeldsá and Krabbe 1990, p. 70; INRENA 1996, p. 1). Experts believe that the species was previously distributed throughout the entire 57-mi² (147-km²) lake (BLI 2003, p. 1; BLI 2009a, p. 1; Fjeldsá 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldsá 2004, p. 200). In 1938, the Junín grebe was encountered throughout the entire lake (Morrison 1939, p. 645). The Junín grebe is now absent from the northwest portion of Lake Junín due to mine waste contamination and a severe decline in population (Fjeldsá 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldsá 2004, p. 200).

Current Range and Distribution

The Junín grebe continues to be endemic to the 57-mi² (147-km²) Lake Junín, located at 13,390 ft (4,080 m) above sea level in the Peruvian High Andes (BLI 2003, p. 1; BLI 2009a, p. 1; BLI 2009b, p. 1). Although BirdLife International (2009b, p. 1) reports the current estimated range of the species as 55 mi² (143 km²), their definition of a species’ range is the total area within its extent of occurrence (see Current Range and Distribution of the ash-breasted tit-tyrant) (BLI 2000, pp. 22, 27). Noting that Lake Junín is only a 57-mi² (147-km²) lake (BLI 2003, p. 1; BLI 2009a, p. 1) and that the Junín grebe is restricted to the southern portion of the lake (Fjeldsá 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldsá 2004, p. 200), its current range is actually smaller than the figure reported by BirdLife International. The entire population of this species is located within a protected area, the Junín National Reserve (BLI 2008, p. 2; BLI 2009a, p. 1; BLI 2009b, p. 1; ParksWatch 2009, p. 4).

Population Estimates

The current population of the Junín grebe is estimated to be 100–300 individuals (BLI 2009b, p. 3), having undergone a severe population decline in the latter half of the 20th century, with extreme population fluctuations during this time (Fjeldsá 1981, p. 254). Field studies in 1938 indicated that the Junín grebe was “extremely abundant” throughout Lake Junín (Morrison 1939, p. 645). Between 1961 and 1979 the population fell from greater than 1,000 individuals to an estimated 250–300 birds (BLI 2009b, p. 2; Collar *et al.* 1992, p. 43; Harris 1981, as cited in O’Donnell and Fjeldsá 1997, p. 30; Fjeldsá 1981, p.

254). Surveys during the mid-1980s estimated a total of 250 individuals inhabiting the southern portion of Lake Junín (BLI 2009b, p. 2; Collar *et al.* 1992, p. 43). In 1992, only 100 birds were observed and, by 1993, the population had declined to 50 birds, of which fewer than half were breeding adults (BLI 2008, p. 3; BLI 2009b, p. 2). In 1995, an estimated 205 Junín grebes were present on Lake Junín (O'Donnell and Fjeldsá 1997, p. 30). Breeding and fledging were apparently unsuccessful from 1995 to 1997. However, there were two successful broods fledged during the 1997 and 1998 breeding seasons (BLI 2008, p. 3; T. Valqui in litt., as cited in BLI 2009b, p. 2). In 1998, more than 250 Junín grebes were counted in a 1.5-mi² (4-km²) area in the southern portion of Lake Junín, suggesting a total population of 350 to 400 birds (T. Valqui in litt., as cited in BLI 2009b, p. 2). In 2001, field surveys indicated that there may have been a total population of 300 birds, but that estimate has been considered overly optimistic (Fjeldsá in litt. 2003, as cited in BLI 2009b, p. 2). Fjeldsá (in litt. 2003, as cited in BLI 2009b, p. 2) postulated that perhaps only half that number would have been mature individuals. These population fluctuations are strongly linked to precipitation (see Factor A).

The species has experienced a population decline of 14 percent in the past 10 years, and is expected to continue to decline (BLI 2009b, pp. 1, 6-7). The species' decline is associated with continued habitat loss and degradation (Fjeldsá 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldsá 2004, p. 200) (See Factors, A, C, and E).

Conservation Status

The Junín grebe is considered "critically endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276853). The IUCN categorizes the Junín grebe as "Critically Endangered" because it is endemic to one location and has undergone "significant population declines, such that an extremely small number of adults remain" (BLI 2008, p. 1; BLI 2009b, pp. 1, 3). The single known population of the Junín grebe occurs wholly within one protected area in Peru, the Junín National Reserve (BLI 2009b, pp. 1-2).

Summary of Factors Affecting the Junín Grebe

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

The Junín grebe is endemic to Lake Junín, where it resides year-round. The species is completely dependent on the open waters and marshland margins of the lake for feeding and on the protective cover of the marshlands during the breeding season (BLI 2008, p. 1; BLI 2009a, p. 1; Fjeldsá 1981, p. 247; Tello 2007, p. 3). The current estimated range of the species is 55 mi² (143 km²) (BLI 2009b, p. 1). However, as described under Current Range and Distribution, its actual range is smaller, because the species is restricted to the southern portion of the lake (BLI 2009b, p. 1; Fjeldsá 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldsá 2004, p. 200). Breeding season begins in November (Fjeldsá 1981, pp. 44, 246; O'Donnell and Fjeldsá 1997, p. 29). Junín grebes build their nests (BLI 2008, p. 1; Fjeldsá 1981, p. 247; Tello 2007, p. 3) and obtain their primary prey, pupfish, in the expansive offshore flooded marshlands that may extend into the lake up to 1–3 mi (2–5 km) from shore (Fjeldsá 2004, p. 200; O'Donnell and Fjeldsá 1997, pp. 29-30).

The quality of Junín grebe habitat and the species' reproductive success is highly influenced by water levels and the water quality of the lake. Water levels in the lake are affected by manmade activities (such as hydropower generation) that are exacerbated by unpredictable climate fluctuations (such as drought or excessive rain). Water quality in Lake Junín has been compromised by contamination.

The Upamayo Dam, located at the northwest end of the lake, has been in operation since 1936, and lake water is used to power the 54-megawatt Malpaso hydroelectric plant (Martin *et al.* 2001, p. 178; ParksWatch 2006, p. 5). Dam operations have caused seasonal water level fluctuations of as much as 6 ft (2 m) in Lake Junín (Martin and McNee 1999, p. 659). Under normal conditions, water levels are lower in the dry season and the marshlands can become partially or completely dry (BLI 2009b, p. 1; ParksWatch 2009, p. 2). The floodgates of the dam are often opened during the dry season (June to November) (BLI 2009b, p. 1; ParksWatch 2009, p. 2), and water offtake for hydropower generation further drains the lake, such that by the end of the dry season, in November, the marshlands encircling the lake are more apt to

become completely desiccated (Fjeldsá 2004, p. 123).

Reduced water levels directly impact the Junín grebe's breeding success, by reducing the amount of available nesting habitat (BLI 2008, p. 1; Fjeldsá 2004, p. 200). The giant bulrush marshlands, upon which the Junín grebe relies for nesting and foraging habitat, have virtually disappeared from some sections of the lake (O'Donnell and Fjeldsá 1997, p. 29). When the marshlands are completely desiccated, the Junín grebe does not breed at all (Fjeldsá 2004, p. 123).

Reduced water levels also impact the species by reducing the Junín grebe's primary prey, pupfish (Fjeldsá 2004, p. 200) (see Habitat and Life History). The perimeter of the flooded marshlands provides the primary recruitment habitat for fish in the lake during extremely dry years, including 1983–1987, 1991, and 1994–1997 (Fjeldsá 2004, p. 200; O'Donnell and Fjeldsá 1997, p. 29). Submerged aquatic vegetation, habitat for pupfish, has become very patchy, further triggering declines in the prey population. Few marshlands are permanently inundated now, due to the power generation requirements of the Upamayo Dam, and the giant bulrushes that previously grew tall and provided extensive cover for this species for breeding and feeding have virtually disappeared, reducing both nesting and foraging habitat for the Junín grebe. The reduction in nesting and foraging habitat are believed to contribute to mass mortality of Junín grebes during extreme drought years (O'Donnell and Fjeldsá 1997, p. 30).

Manipulation of the Lake Junín's water levels also results in competition between the white-tufted grebe (*Rollandia rolland*) and the Junín grebe for food resources during the Junín grebe's breeding season (Fjeldsá 2004, p. 200). Under normal conditions, the expansive offshore marshlands may extend into the lake up to 1–3 mi (2–5 km) from shore (O'Donnell and Fjeldsá 1997, p. 29). In years when water levels remain high, the Junín grebe and white-tufted grebe are spatially segregated during the breeding season, with white-tufted grebes utilizing the interior of the reed marsh and Junín grebes remaining at the edges of the marshlands, closer to the center of the lake (Fjeldsá 1981, pp. 245, 255). Near the end of the dry season, as early as October, when water levels are lower in the lake and the marshlands can partially or completely dry out (BLI 2009b, p. 1; ParksWatch 2009, p. 2), thousands of white-tufted grebes move from the interior of the marshlands to the edges, where they compete with the Junín grebe for food

(Fjeldsá 1984, pp. 413-414). As the breeding season for the Junín grebe begins in November (Fjeldsá 1981, pp. 44, 246; O'Donnel and Fjeldsá 1997, p. 29), Junín grebes build floating nests and breed on the margins of marshlands (BLI 2008, p. 1; Fjeldsá 1981, p. 247; Tello 2007, p. 3), and a plentiful supply of fish becomes more important (O'Donnel and Fjeldsá 1997, p. 29). Competition becomes more critical the longer the water level remains low at the end of the dry season, and activities that further reduce low water levels only accentuate this competition (Fjeldsá 1981, pp. 252-253).

Water quality affects the availability of habitat for the endemic Junín grebe. The water in Lake Junín has been contaminated from mining waste, agricultural runoff, and organic matter from the land surrounding the lake. There are several mining operations (lead, copper, and zinc) north of Lake Junín, and wastewater from the mines flows untreated into the lake via the Río San Juan (Fjeldsá 1981, p. 255; Martin and McNee 1999, pp. 660-661; ParksWatch 2006, p. 2; Shoobridge 2006, p. 3). Agricultural insecticides wash into Lake Junín from surrounding fields and through drainage systems from villages around the lake (ParksWatch 2006, pp. 5, 19). Organic matter originating from local communities is piped untreated into the lake, resulting in eutrophication (a process whereby excess nutrients facilitate excessive plant growth, which ultimately reduces the amount of dissolved oxygen in the water, harming oxygen-dependent organisms) (ParksWatch 2006, p. 5; Shoobridge 2006, p. 3).

Lake Junín is a sink for several streams that transport mining wastes and other pollution downstream and into the lake (ParksWatch 2006, p. 19). The Río San Juan is the primary source of water for Lake Junín and feeds into the lake from the northern end (Fjeldsá 1981, p. 255; Martin and McNee 1999, pp. 660-661; Shoobridge 2006, p. 3). Tests indicate that the Río San Juan contains trace metals, including copper, lead, mercury, and zinc, in excess of currently accepted aquatic life thresholds (Martin and McNee 1999, pp. 660-661). Non-point source pollutants from agricultural fertilizers, such as ammonium and nitrate concentrations, are also suspended in the water column (Martin and McNee 1999, pp. 660-661). Iron oxide contamination is visible near the outflow of the Río San Juan because iron oxide produces a reddish tinge to the water and reed borders. Vegetation near the river's outflow is completely absent (Fjeldsá 2004, p. 124;

ParksWatch 2006, pp. 20-21), and this portion of the lake has been rendered lifeless by the precipitation of iron oxide from mining wastewaters (BLI 2008, p. 4). The giant bulrush marshlands, which once existed in great expanses around the entire perimeter of the lake and upon which the Junín grebe relies for nesting and foraging habitat, have virtually disappeared and at least one species of catfish (*Pygidium oroyae*) may have been extirpated from the lake (O'Donnel and Fjeldsá 1997, p. 29).

Heavy metal contamination is not limited to the northern end of the lake (ParksWatch 2006, p. 20), but extends throughout the southern end (Martin and McNee 1999, p. 662), where the Junín grebe is now restricted (BLI 2003, p. 1; BLI 2009b, p. 1; Fjeldsá 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldsá 2004, p. 200). Near the center of the lake, the bottom has been described as "lifeless," due to sedimentation of iron oxides (Fjeldsá 1981, pp. 255-256; Fjeldsá 2004, p. 124). Martin *et al.* (2001, p. 180) determined that sediments at the lake's center are enriched with copper, zinc, and lead and are anoxic (having low levels of dissolved oxygen). High concentrations of dissolved copper, lead, and zinc have damaged an estimated one-third of the lake, increasing turbidity of the lake, and exceeding established aquatic life thresholds (Martin and McNee 1999, pp. 660-661; ParksWatch 2006, pp. 2, 20; Shoobridge 2006, p. 3). This has severely affected animal and plant populations in the area, contributing to mortality of species, including the Junín grebe, around the lake (ParksWatch 2006, pp. 3, 20) (see Factor C).

In 2009, conservation organizations and civil society groups demanded action to reverse the deterioration of Lake Junín and requested an independent environmental audit and continuous monitoring of the lake (BLI 2009b p. 4; BLI 2009c, p. 1). The conservation groups BLI, American Bird Conservancy (ABC), Asociación Ecosistemas Andinos (ECOAN), and INRENA adopted the Junín grebe as the symbol of wetland conservation for the high Andes (BLI 2009c, p. 1). Although translocation has been a consideration for the conservation of the Junín grebe since the mid-1990s, no suitable habitat for the species has been located (O'Donnel and Fjeldsá 1997, pp. 30, 35; BLI 2008, p. 5; BLI 2009b, p. 2). None of these conservation organization's activities have been effective at curbing the ongoing habitat degradation (see also Factor D).

The effects of habitat alteration and destruction (such as those caused by artificially reduced water levels and

water contamination) are accentuated by unpredictable climate fluctuations (such as droughts or excessive rains) (Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027). Peru is subject to unpredictable climate fluctuations, such as those that are related to the El Niño Southern Oscillation (ENSO). Changes in weather patterns, such as ENSO cycles (El Niño and La Niña events), tend to increase precipitation in normally dry areas, and decrease precipitation in normally wet areas (Holmgren *et al.* 2001, p. 89; TAO Project n.d., p. 1); thereby exacerbating the effects of habitat reduction and alteration on the decline of a species (England 2000, p. 86; Holmgren *et al.* 2001, p. 89; Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027; Parmesan and Mathews 2005, p. 334; Plumart 2007, pp. 1-2; Timmermann 1999, p. 694), especially for narrow endemics (Jetz *et al.* 2007, p. 1213) such as the Junín grebe (see also Factor E). Moreover, the Junín grebe's low breeding potential is considered to be a reflection of its adaptation to being in a "highly predictable, stable environment" (del Hoyo *et al.* 1992, p. 195).

The Junín grebe's breeding success and population size are highly influenced by the climate, with population declines occurring during dry years, population increases during rainy years, and mortality during extreme cold weather events. Several times during the last two decades (e.g., 1983-1987, 1991-1992, 1994-1997), the population has declined to 100 birds or less following particularly dry years (BLI 2008, pp. 1, 3-4; BLI 2009b, p. 2; Elton 2000, p. 3; Fjeldsá 2004, p. 200). There have been short-term population increases of 200 to 300 birds in years with higher rainfall amounts following El Niño events (such as the 1997-1998 and 2001-2002 breeding seasons) (PROFONANPE 2002, as cited in Fjeldsá 2004, p. 133; T. Valqui pers. comm., as cited in BLI 2009b, p. 2). However, excessive rains also increase contamination in Lake Junín, which decreases the amount of suitable habitat for the species (as described above) and has adverse effects on the species' health (see Factor C). Many Junín grebes died during extremely cold conditions in 1982 (BLI 2008, p. 4). In 2007, the population declined again following another cold weather event (Hirshfeld 2007, p. 107). ENSO cycles are ongoing, having occurred several times within the last decade (NWS 2009, p. 2), and evidence suggests that ENSO cycles have already increased in periodicity and severity (Richter 2005, pp. 24-25;

Timmermann 1999, p. 694), which will exacerbate the negative impacts of habitat destruction on a species.

Habitat degradation and alteration caused by fluctuating water levels and environmental contamination are considered key factors in the species' historical decline (Fjeldsá 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldsá 2004, p. 200). The species has experienced a population decline of 14 percent in the past 10 years, and is expected to continue to decline as a result of deteriorating habitat and water quality (BLI 2009b, pp. 1, 6-7). Therefore, further habitat degradation is expected to continue impacting this species' already small population size (see Factor E).

Summary of Factor A

The habitat in and around Lake Junín, where the Junín grebe is endemic, has been and continues to be altered and degraded as a result of human activities, including artificial water level fluctuations to generate hydropower and water contamination caused by mining waste, agricultural and organic runoff from surrounding lands, and wastewater from local communities. The Junín grebe is dependent on the quantity and quality of lake water for breeding and feeding. Water levels in Lake Junín are manipulated to generate electricity, which leads to dramatic fluctuations in water levels of up to 6 ft (1.8 m). The Junín grebe relies on the protective cover of flooded marshlands for nesting. As water drawdown occurs near the end of the dry season and the inception of the Junín grebe's mating season, portions of the marshlands may dry out completely. Reductions in water levels decrease the availability of suitable breeding and foraging habitat, and decrease the availability of the Junín grebe's primary prey, forcing competition with the white-tufted grebe for food. Drought years have a negative impact on the Junín grebe, resulting in severe population fluctuations due to poor breeding success and limited recruitment of juveniles into the adult population. Severed dry conditions can cause total breeding failure (see also Factor E).

Although the population appears to rebound during wetter years (i.e., following El Niño events) (see Habitat and Life History and Population Estimate), excessive rain decreases the suitable habitat for the species, as pollution washes into the water from around the lake and the upstream rivers that feed the lake, increasing contamination levels in Lake Junín. This increased contamination also affects the Junín grebe's health and has resulted in

mortality of the species (see Factor C). Severe water contamination has rendered the northwest portion of the lake lifeless, devoid of aquatic and terrestrial species. Experts believe that the Junín grebe once inhabited the entire Lake, but the species is now confined to the southern portion of the lake due to water contamination (Historical Range and Distribution). Elevated levels of heavy metals may reduce the fitness and overall viability of the Junín grebe (Factor C), which would heighten risks associated with short- and long-term genetic viability (Factor E). Therefore, we find that destruction and modification of habitat are threats to the continued existence of the Junín grebe throughout its range.

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

We are not aware of any information currently available that indicates that overutilization of Junín grebe for commercial, recreation, scientific, or education purposes has occurred or is occurring at this time. Fjeldsá (1981, pp. 254-255) notes that local hunters are not interested in grebes as food because they have "too little meat." As a result, we are not considering overutilization to be a threat to the continued existence of the Junín grebe.

C. Disease or Predation

Disease: Although no specific disease threat has been identified for the Junín grebe, contamination of Lake Junín has contributed directly and indirectly to Junín grebe mortality and has potentially reduced the overall fitness and health of the species. As discussed under Factor A, lead, copper, and zinc mining residues (Fjeldsá 1981, p. 255; Martin and McNee 1999, pp. 660-661; Shoobridge 2006, p. 3), agricultural runoff, organic matter, and wastewater are discharged directly into Lake Junín (ParksWatch 2006, pp. 5, 19; Shoobridge 2006, p. 3). High concentrations of environmental contaminants (including ammonium, copper, iron oxide, lead, mercury, nitrate, and zinc) have been detected throughout the lake (Fjeldsá 1981, pp. 255-256; Fjeldsá 2004, p. 124; Martin and McNee 1999, pp. 660-662; ParksWatch 2006, pp. 20-21) and exceed established thresholds for aquatic life (Martin and McNee 1999, pp. 660-661; ParksWatch 2006, p. 20). Chemical waste has rendered the northern portion of the lake lifeless due to eutrophication (BLI 2008, p. 4; Shoobridge 2006, p. 3) and the sediments in the center of the lake anoxic (containing no dissolved oxygen) (Martin *et al.* 2001, p. 180). High concentrations of suspended

particulate matter increases the turbidity of the water, making it less penetrable to sunlight and resulting in die-off of aquatic plants and algae (ParksWatch 2006, p. 20). Chemical waste has damaged at least one third of the lake and has severely affected animal and plant populations in the area (O'Donnel and Fjeldsá 1997, p. 29; ParksWatch 2006, pp. 3, 20; Shoobridge 2006, p. 3). The northern portion of the lake is completely devoid of vegetation (Fjeldsá 2004, p. 124; ParksWatch 2006, pp. 20-21), and the giant bulrush marshlands, which once existed in great expanses around the entire perimeter of the lake and upon which the Junín grebe relies for nesting and foraging habitat, have virtually disappeared. At least one species of catfish (*Pygidium oroyae*) may have been extirpated from the lake (O'Donnel and Fjeldsá 1997, p. 29). During years of heavy rainfall, the lake is filled, and the lakeshore becomes polluted with "toxic acid gray sediment" that has caused large-scale mortality of cattle (approximately 2,000 in 1994) and birds, apparently due to lead poisoning (O'Donnel and Fjeldsá 1997, p. 30). Lead poisoning from the presence of mine wastes is a common cause of mortality in waterbirds, and is medically described as an intoxication resulting from absorption of hazardous levels of lead into body tissues (Friend and Franson 1999, p. 317).

Water contamination has directly affected the health of the Junín grebe population. As predators of aquatic organisms, the Junín grebe occupies a mid-tertiary level position in the food chain and is prone to bioaccumulation of pesticides, heavy metals, and other contaminants that are absorbed or ingested by its prey (Fjeldsá 1981, pp. 255-256; Fjeldsá 2004, p. 123). Green plants form the first trophic, or feeding, level; they are the primary producers. Herbivores form the second trophic level, while carnivores form the third and even fourth trophic levels (The University of the Western Cape 2009, p. 1). Moreover, species such as the Junín grebe, which inhabit high trophic levels, are strictly dependent upon the functioning of a multitude of ecosystem processes. The loss or absence of species at lower trophic levels can result in cascading ecosystem effects, causing imbalances in the food web at all higher trophic levels (The University of the Western Cape 2009, p. 1). In parts of the lake, increased turbidity has caused die-off of aquatic plants and algae, disrupting the food chain (ParksWatch 2006, p. 20). Studies indicate that lead mining effluents severely reduce or eliminate primary prey populations of

fish and aquatic invertebrates, either directly through lethal toxicity, or indirectly through toxicity to their prey species (Demayo *et al.* 1982, as cited in Eisler 1988, p. 5). Analysis of feathers and bone tissue of Junín grebes and of pupfish, the species' primary prey, indicate that both the grebe and its prey contain elevated lead levels (Fjeldsá 1981, pp. 255-256).

Drought conditions exacerbate the effects of water contamination and bioaccumulation of contaminants in aquatic species. From 1989 to 1992, an extensive drought occurred in the Lake Junín area. During that time, many dead Junín grebes and other waterbirds were found along the edges of the lakeshore (T. Valqui and J. Barrio in litt. 1992, as cited in Collar *et al.* 1992, p. 45, 190). In 1992, one of the driest years in decades, up to 10 dead grebes per month were reported around the lake. Three Junín grebe carcasses were found along 1.2 mi (2 km) of shoreline in one month alone (T. Valqui and J. Barrio in litt. 1992, as cited in Collar *et al.* 1992, p. 45). Experts consider the cause of death to have been either heavy metal contamination, which increased in concentration as water levels decreased (T. Valqui and J. Barrio in litt. 1992, as cited in Collar *et al.* 1992, p. 45), or reduced prey availability (Fjeldsá 2004, p. 124). Reduced prey availability is exacerbated by manmade activities that are reducing the water levels of the lake, increasing competition among sympatric grebe species (different species that occupy the same range) and decreasing the marshlands that provide primary spawning habitat for the pupfish, the grebe's primary prey species (Factor A).

Persistent exposure to contaminants can contribute to a decline in fitness for long-lived, mid-trophic level species, which is inherited by offspring and can impact embryonic development, juvenile health, or viability (Rose 2008, p. 624). The excessive contaminant load in Lake Junín could also allow opportunistic bacterial and viral infections to overcome individuals. According to Fjeldsá (1981, p. 254), the Junín grebe bears a heavy infestation of stomach nematodes (parasitic roundworms), especially as compared to other grebe species. Stomach contents of Junín grebes that have been examined had an average of 16.7 nematodes, compared with no nematodes in silver grebes and 1.6 nematodes in white-tufted grebes. Fjeldsá (1981, p. 254) postulates that the higher nematode infestation in Junín grebes may be an indicator of poor health.

Predation: Predators around Lake Junín include the Andean fox

(*Pseudalopex culpaeus*), the long-tailed weasel (*Mustela frenata*), Pampas cat (*Onicifelis colocolo*), and hog-nosed skunk (*Conepatus chinga*) (ParksWatch 2009, p. 4). However, nest sites of the Junín grebe are generally inaccessible to mammalian predators (Fjeldsá 1981, p. 254). The only raptor likely to take a grebe on Lake Junín is the Cinereous harrier (*Circus cinereus*), which primarily feeds in white-tufted grebe habitats. Moorhens (*Gallinula chloropus*), which also inhabit the lake (ParksWatch 2009, p. 3; Tello 2007, p. 2), are egg stealers and may steal Junín grebe eggs for food (Fjeldsá 1981, p. 254). However, there is no direct evidence of predation upon the Junín grebe.

Summary of Factor C

Environmental contamination poses direct and indirect threats to the Junín grebe's overall health and survival. The species' trophic level also exposes it to bioaccumulation of toxins accumulated in the tissue of prey species. Research indicates that the species has elevated lead levels and carries a high load of nematodes, a possible indicator of overall poor health. Junín grebes have died as a direct result of contaminant poisoning or reduction in the pupfish, which has also been found to carry elevated lead levels. Therefore, we find that disease is a threat to the continued existence of the Junín grebe. However, there is no available evidence to indicate that predation is causing declines in Junín grebe populations or otherwise contributing to the species' risk of extinction. Therefore, we find that predation is not a threat to the Junín grebe.

D. Inadequacy of Existing Regulatory Mechanisms

The Junín grebe is listed as "critically endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276853). This Decree prohibits hunting, take, transport, and trade of protected species, except as permitted by regulation. As hunting, take, transport, and trade do not currently threaten the Junín grebe, this regulation does not mitigate any current threats to this species.

Peru has several categories of national habitat protection, which were described above as part of the Factor D analysis for the ash-breasted tit-tyrant (BLI 2008, p. 1; IUCN 1994, p. 2; Rodríguez and Young 2000, p. 330). The Junín grebe population occurs wholly within one protected area: the Junín National Reserve (Junín, Peru) (BLI 2009b, pp. 1-2). The Junín National

Reserve has an area of 133,437 ac (53,000 ha), bordering Lake Junín and its adjacent territories (Wege and Long 1995, p. 264). In Peru, National Reserves are also created for the sustainable extraction of certain biological resources (BLI 2008, p. 1; Rodríguez and Young 2000, p. 330). Established in 1974, through Supreme Decree No. 0750-74-AG, the stated objectives of the Junín National Reserve include: integrated conservation of the local ecosystem, its associated flora and wildlife; preservation of the scenic beauty of the lake; and support of socioeconomic development in the area through the sustainable use of its renewable natural resources (BLI 2009a, p. 2; Hirshfeld 2007, p. 107). Most of the lakeshore is designated a "Direct Use Zone," which allows fishing, grazing, and other educational, research, and recreational activities (ParksWatch 2006, p. 12). Although designation of this reserve has heightened awareness of the ecological problems at Lake Junín (BLI 2009c, p. 1), it has not reduced or eliminated the primary threats to the Junín grebe: water fluctuations and contamination (Factor A), contamination resulting in poor health (Factor C), and small population size (Factor E). Therefore, the existence of this species within a protected area has not reduced or mitigated the threats to the species.

The Junín National Reserve was designated a Ramsar site under the Convention on Wetlands of International Importance (Ramsar Convention) in 1997 (BLI 2009a, p. 2; Hirshfeld 2007, p. 107; INRENA 1996, pp. 1-14). The Ramsar Convention, signed in Ramsar, Iran, in 1971, is an intergovernmental treaty which provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources. There are presently 159 Contracting Parties to the Convention (including Lake Junín), with 1,874 wetland sites, totaling more than 457 million ac (185 million ha), designated for inclusion in the Ramsar List of Wetlands of International Importance (Ramsar 2009, p. 1). Peru acceded to Ramsar in 1992. It has 13 sites on the Ramsar list, comprising 16.8 million ac (6.8 million ha) (Ramsar 2009, p. 5). In an examination of 5 Ramsar sites, experts noted that Ramsar designation may provide nominal protection (protection in name only) by increasing both international awareness of a site's ecological value and stakeholder involvement in conservation (Jellison *et al.* 2004, pp. 1, 4, 19). However, activities that negatively impact the Junín grebe are

ongoing within this Ramsar wetland, including water fluctuations and contamination (Factor A), contamination resulting in poor health (Factor C), and small population size (Factor E). Therefore, the Ramsar designation has not mitigated the impact of threats on the Junín grebe.

In 2002, the Peruvian Government passed an emergency law to protect Lake Junín. This law makes provisions for the cleanup of Lake Junín, and places greater restrictions on extraction of water for hydropower and mining activities (J. Fjeldså in litt. 2003, as cited in BLI 2007, p. 3). However, this law has not been effectively implemented, and conditions around the lake may even have worsened after passage of this law (BLI 2009c, p. 1). The Ministry of Energy and Mining has implemented a series of Environmental Mitigation Programs (PAMA) to combat mine waste pollution in the Junín National Reserve (ParksWatch 2006, p. 21; ParksWatch 2009 p. 3). The PAMAs were scheduled to have been completed by 2002, but extensions have been granted, indicating that many of the mines currently in operation are still functioning without a valid PAMA. Reductions in pollution are reported because some mine companies have begun to utilize drainage fields and recycle residual water. However, analysis of existing PAMAs indicate that they do not address specific responsibilities for mining waste discharged into the Río San Juan and delta, nor do they address deposition of heavy metal-laced sediments in Lake Junín (ParksWatch 2006, p. 21; ParksWatch 2009, p. 3). Recent information indicates that mining waste contamination in the lake continues to be a source of pollution (ParksWatch 2006, pp. 20-21; Fjeldså 2004, p. 124; BLI 2009b, p. 1). Therefore, this law is not effective at mitigating the threat of habitat degradation (Factor A), health issues associated with contamination (Factor C) and small population size of the species (Factor E).

There are approximately 5,000 laws and regulations directly or indirectly related to environmental protection and natural resource conservation in Peru. Recent studies by the Peruvian Society for Environmental Law (SPDA) have concluded that many of these are not effective because of limited implementation and/or enforcement capability (Muller 2001, pp. 1-2).

Summary of Factor D

Peru has enacted various laws and regulatory mechanisms for the protection and management of wildlife and their habitats. The Junín grebe is

“critically endangered” under Peruvian law, and its entire population occurs within one protected area. As discussed under Factor A, the Junín grebe’s distribution, breeding success and recruitment, and food availability on Lake Junín has been curtailed, and are negatively impacted due to habitat destruction that is caused by artificial water fluctuations and water contamination from human activities. These habitat-altering activities are ongoing throughout the species’ range, which is wholly encompassed within one protected area. Thus, despite the species’ critically endangered status and presence within a designated protected area, laws governing wildlife and habitat protection in Peru are inadequately enforced or ineffective at protecting the species or mitigating ongoing habitat degradation (Factor A), impacts from contaminants, and concomitant population declines (Factor E). Therefore, we find that the existing regulatory mechanisms are inadequate to mitigate the threats to the continued existence of the Junín grebe throughout its range.

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

An additional factor that affects the continued existence of the Junín grebe is the species’ small population size. The current population of the Junín grebe is estimated to be 100–300 individuals, however, only an “extremely small number of adults remain” (BLI 2008, p. 1; BLI 2009b, pp. 1, 3). The number of adults in a population are important because these individuals contribute to the next generation (Shaffer 1981, pp. 132-133; Soulé 1980, pp. 160-162). The Junín grebe underwent a severe population decline in the latter half of the 20th century, with extreme population fluctuations (Fjeldså 1981, p. 254) (see Population Estimates). For example, in 1993, the population size declined to below 50 individuals, of which fewer than half were breeding adults (BLI 2008, p. 3; BLI 2009b, p. 2). Even if the estimate of 100–300 individuals is correct, the number of mature individuals is likely to be far smaller, perhaps only half (Fjeldså in litt. 2003, as cited in BLI 2009b, p. 2). Therefore, 100–300 individuals overestimates the species’ effective population size (the number of breeding individuals that contribute to the next generation) (Shaffer 1981, pp. 132-133; Soulé 1980, pp. 160-162).

Small population size renders species vulnerable to genetic risks that can have individual or population-level genetic

consequences, such as inbreeding depression, loss of genetic variation, and accumulation of new mutations, and may affect the species’ viability by increasing its susceptibility to demographic shifts or environmental fluctuations, as explained in more detail above in the Factor E analysis for the ash-breasted tit-tyrant (Charlesworth and Charlesworth 1987, p. 238; Pimm *et al.* 1988, pp. 757, 773-775; Shaffer 1981, p. 131). Small population size also leads to a higher risk of extinction and, once a population is reduced below a certain number of individuals, it tends to rapidly decline towards extinction (Frankham 1996, p. 1507; Franklin 1980, pp. 147-148; Gilpin and Soulé 1986, p. 25; Holsinger 2000, pp. 64-65; Purvis *et al.* 2000, p. 1949; Reed and Frankham 2003, pp. 233-234; Soulé 1987, p. 181). In addition, species that inhabit a small geographic range, occur at low density, occupy a high trophic level, and exhibit low reproductive rates tend to have a higher risk of extinction than species that are not limited by the same risk factors (Purvis *et al.* 2000, p. 1949).

Complications arising from the species’ small population size are exacerbated by the species’ restricted range and threat of disease (Factor C). The Junín grebe is known only from a single Andean lake, Lake Junín, in central Peru (BLI 2000, p. 45; BLI 2009b, p. 1; Collar *et al.* 1992, p. 43). Although the species was believed to have been previously distributed throughout the entire 57-mi² (147-km²) lake (Fjeldså 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldså 2004, p. 200; Morrison 1939, p. 645), it is now restricted to the southern portion of Lake Junín (BLI 2009b, p. 1; Fjeldså 1981, p. 254; F. Gill and R.W. Storer, pers. comm. as cited in Fjeldså 2004, p. 200). The population has declined by at least 14 percent in the last 10 years and is expected to continue to decline, as a result of declining water quality and extreme water level fluctuations (BLI 2009b, pp. 1, 4, 6-7) (Factor A). We consider that the risks associated with small population size will continue to impact this species and may accelerate, if habitat destruction continues unabated. Environmental contamination poses direct and indirect threats to the Junín grebe’s overall health and survival, including the presence of toxins in both the Junín grebe and its primary prey species and mass die-offs that are linked to contamination or reduction in prey species (Factor C). A species’ small population size, combined with its restricted range and threat of disease, increases the species’

vulnerability to adverse natural events and manmade activities that destroy individuals and their habitat (Holsinger 2000, pp. 64-65; Primack 1998, pp. 279-308; Young and Clarke 2000, pp. 361-366).

Summary of Factor E

The Junín grebe has a small population size that renders it vulnerable to genetic risks that negatively impact the species' long-term viability and, possibly, its short-term viability. The species has a restricted range and occurs in habitat that continues to undergo degradation and curtailment due to human activities (Factor A). Environmental contaminants have caused die-offs of the species and have likely reduced the overall general health of the Junín grebe population (Factor C). The small population size, as well as its restricted range and health issues associated with contamination, increases the species' vulnerability to extinction, through demographic or environmental fluctuations. Based on its small population size, restricted range, and threat of disease, we have determined that the Junín grebe is particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., water level manipulation) that destroy individuals and their habitat. The genetic and demographic risks associated with small population sizes are exacerbated by ongoing human activities that continue to curtail the species' habitat throughout its range. The species' population has declined and is predicted to continue declining due to an ongoing reduction in water quality and extreme water level fluctuations (Factor A). Therefore, we find that the species' small population size, in concert with its restricted range, threat of disease, and its heightened vulnerability to adverse natural events and manmade activities, are threats to the continued existence of the Junín grebe throughout its range.

Status Determination for the Junín Grebe

The Junín grebe, a flightless grebe, is endemic to Lake Junín, found at 13,390 ft (4,080 m) above sea level in Peru, where it resides year-round. The species relies on the open waters and marshland margins of the lake for feeding and on the protective cover of the marshland margins for nesting during the breeding season. The species has a highly restricted range (approximately 55 mi² (143 km²)) and is currently known only in one location in central Peru. The species' population size is estimated as 100–300 individuals, although the

number of mature individuals may be limited to half this amount.

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the Junín grebe and have concluded that there are four primary factors that threaten the continued existence of the Junín grebe: (1) Habitat destruction, fragmentation, and degradation; (2) disease; (3) limited size and isolation of remaining populations; and (4) inadequate regulatory mechanisms.

Human activities that degrade, alter, and destroy habitat are ongoing throughout the Junín grebe's range. Lake waters are artificially manipulated to produce hydropower, resulting in reductions in water levels that impact the species' nesting and foraging sites. Manipulation of water levels for hydropower production reduces prey populations, causes increased food competition with white-tufted grebes, and results in the abandonment of breeding in drought years. Reduced water levels have permanently destroyed segments of giant bulrush communities, compromising the amount of suitable flooded marshland available for nesting (Factor A). Mining, agricultural, and organic materials have contaminated the water, causing eutrophication and anoxia in portions of the lake and the accumulation of trace minerals in lake bottom sediments. This has had direct effects on the Junín grebe, destroying habitat in the northwest portion of the lake so that the species' range is restricted to only the southern portion of Lake Junín and causing Junín grebe mortality during times of drought (Factors A and C). Contaminants have also reduced or eliminated submerged and emergent vegetation throughout the lake, decreasing pupfish spawning habitat and reducing prey availability (Factor A).

Junín grebe habitat continues to be altered by human activities, conversion, and destruction of habitat, which reduce the quantity, quality, distribution, and regeneration of habitat available for the Junín grebe on Lake Junín. Habitat loss was a factor in the Junín grebe's historical population decline (see Historical Range and Distribution). Population declines have been correlated with water availability, and droughts have caused severe population fluctuations that have likely compromised the species' long-term viability. The Junín grebe population is small, rendering the species vulnerable to the threat of adverse natural (e.g., genetic, demographic) and human activity (e.g., water extraction and contaminants from mining) events that

destroy individuals and their habitat. (Factor E). The Junín grebe's reproductive success and life cycle relies on the availability of sufficient water in Lake Junín. During drought years, nesting and reproduction decline.

Although the population appears to rebound during wetter years (such as following excessive rains from El Niño events (see Population Estimate and Factor A), excessive rains also bring additional contaminants into the lake as runoff from lands surrounding the lake and upstream rivers. Research indicates that both the Junín grebe and its primary prey species, the pupfish, have accumulated toxins resulting in elevated lead levels. Environmental contaminants have caused die-offs of the species and have likely reduced the general health of the Junín grebe population (Factor C). The population has declined 14 percent in the past 10 years (see Population Estimates), and this decline is predicted to continue commensurate with ongoing threats from habitat destruction and water contamination (Factor A).

Despite the species' "critically endangered" status in Peru and its occurrence entirely within a protected area, the lake continues to be destroyed and degraded as a result of human activities that alter the lake's water levels and compromise water quality (Factors A and C). Therefore, regulatory mechanisms are either inadequate or ineffective at mitigating the existing threats to the Junín grebe and its habitat (Factor D).

Section 3 of the Act defines an "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range" and a "threatened species" as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Based on the immediate and ongoing threats to the Junín grebe throughout its entire range, as described above, we determine that the Junín grebe is in danger of extinction throughout all of its range. Therefore, on the basis of the best available scientific and commercial information, we are proposing to list the Junín grebe as an endangered species throughout all of its range.

III. Junín rail (*Laterallus tuerosi*)

Species Description

The Junín rail is a secretive bird of the *Rallidae* family that is endemic to a single lake (Lake Junín) in Peru. The species is also referred to as the Junín black rail (Fjeldså 1983, p. 281) and is locally known as "*gallinetita de Junín*"

(Supreme Decree 034-2004-AG 2004, p. 27684). This rail measures 4.7–5.1 in (12–13 cm) in length, and has a dark slate-colored head, throat and underparts. Its belly and vent (anal aperture) are black, with heavily-barred white primary feathers along the back and side. The under-tail coverts (feathers on the underside of the base of the tail) are buff in color, with a dull rufous-brown back. The remaining underparts are dark brown and boldly barred in white, and the legs are greenish-yellow (BLI 2009b, p. 1).

Taxonomy

The Junín rail was first taxonomically described as *Laterallus spilonotus* after Gould's 1841 description of the Galapagos crake (Fjeldsá 1983, p. 281; ITIS 2009b, p. 1). Fjeldsá (1983) later described it as a subspecies of the black rail (*Laterallus jamaicensis tuerosi*) (Fjeldsá 1983, pp. 277-282; Fjeldsá and Krabbe 1990, p. 146; ITIS 2009a, p. 1), noting that the bird's morphology may suggest that it is a distinct species (Fjeldsá 1983, p. 281). *Laterallus jamaicensis* has since been split into *L. jamaicensis* and *L. tuerosi* (BLI 2009b, p. 1). ITIS continues to list the species as the subspecies *L. j. tuerosi* based on Fjeldsá's work in 1983 (ITIS 2009a, p. 1). BirdLife International considers this rail a full species based on morphological differences (BLI 2009b p. 1). Furthermore, the black rail, *Laterallus jamaicensis* occurs at much lower elevations (i.e., 0 to 4,429 ft (0 to 1,350 m) above sea level) (Collar *et al.* 1992, p. 190; BLI 2000, p. 170; BLI 2007, p. 1). Based on the morphological differences and the species' distinct and disjunct ranges, we consider the Junín rail to be a discrete species and recognize it as *L. tuerosi*.

It should be noted that it appears that only 2 specimens of the Junín rail have ever been collected (near Ondores) (Fjeldsá 1983, pp. 278-279) and that all expert accounts of this species rely solely on that collection and a subsequent observation of the species in Pari (Fjeldsá in litt., 1992, as cited in Collar *et al.* 1992, p. 190).

Habitat and Life History

The Junín rail inhabits Lake Junín at 13,390 ft (4,080 m) above sea level in the Andean highlands of Peru (Junín Region). The Junín rail occurs in the dense, interior marshlands where rushes (*Juncus* spp.) predominate or in more open mosaics of rushes, mosses (division Bryophyta), and low herbs (Fjeldsá 1983, p. 281). Lake Junín, home to the Junín grebe that is also the subject of this proposed rule, is located in the seasonally climatic "puna" habitat, with

a variety of species of grasses, asters, and trees of the bean family forming tall, dense grasslands and open scrubland, interspersed with wetlands and woodlands (ParksWatch 2006, p. 2; ParksWatch 2009, pp. 1, 4). Giant bulrushes and totorilla dominate the extensive marshlands surrounding the lake (BLI 2009b, p. 1; Fjeldsá 1983, p. 281; ParksWatch 2009, p. 1). In shallow water, during low lake levels, "tortora" communities can become partially or completely dry (ParksWatch 2009, p. 2). The lake supports a wide variety of bird species and aquatic vegetation (BLI 2003, p. 1; BLI 2009a, pp. 2-3; ParksWatch 2009, p. 3; Tello 2007, p. 2). Mammals are relatively scarce in the area, although there are some predators (ParksWatch 2009, p. 4) (see Factor C). A more detailed discussion of the flora and fauna of the lake are provided above as part of the analysis of the Habitat and Life History of the Junín grebe.

There is little information regarding the ecology of the Junín rail. The species appears to be completely dependent on the wide marshlands located around the southeast shoreline of the lake for nesting, foraging, and year-round residence (BLI 2009b, p. 2; Collar *et al.* 1992, p. 190; Fjeldsá 1983, p. 281) (see also Current Range and Distribution). Similar to all rails, the Junín rail is furtive and remains well-hidden in the marshes surrounding the lake (BLI 2009b, p. 2). The Junín rail reportedly nests at the end of the dry season, in September and October. Nests are built on the ground within dense vegetative cover, and the species' clutch size is two eggs (BLI 2009b, p. 2; Collar *et al.* 1992, p. 190). The diet of the Junín rail has not been studied specifically, but other black rail species feed primarily on small aquatic and terrestrial invertebrates and seeds (Eddleman *et al.* 1994, p. 1).

Historical Range and Distribution

The Junín rail is endemic to Lake Junín (Fjeldsá 1983, p. 278; BLI 2009b, p. 2). The species may have been historically common in the rush-dominated marshlands surrounding the entire lake (Fjeldsá 1983, p. 281). In addition to the species' specific habitat preferences (see Current Range and Distribution), it is believed that the Junín rail is now restricted to the marshes at the southwest corner of the lake because of the high level of water contamination that flows into the northwest margins of the lake via the Río San Juan (Martin and McNee 1999, p. 662).

Current Range and Distribution

The Junín rail is restricted to the southwest shore of the lake, and is currently known in only two localities (near the towns of Ondores and Pari) (BLI 2009b, p. 2; Collar *et al.* 1992, p. 190; Fjeldsá 1983, p. 281). The current estimated range of the species is 62 mi² (160 km²) (BLI 2009b, p. 1). However, this is likely an overestimate of the species' actual range for several reasons. First, BirdLife International's definition of a species' range results in an overestimate of the actual range (see Current Range and Distribution of the ash-breasted tit-tyrant) (BLI 2000, pp. 22, 27). Second, the species' range was calculated based on the availability of presumed suitable habitat for the Junín rail. It has long been assumed that the rail potentially occupies the entire marshland area surrounding Lake Junín (Fjeldsá 1983, p. 281). The total marshland area has been estimated by BirdLife International to be 58 mi² (150 km²) (BLI 2000, p. 170; BLI 2007, p. 1; BLI 2008, p. 3; BLI 2009b, p. 1). However, the species has never been confirmed outside the two known localities on the southwest shore of the lake. Moreover, a better understanding of the "puna" habitat, as well as the habitat specificity (the specific habitat needs) of other rail species, indicates that these may be the only two localities for this species.

Despite the apparently uniform appearance of the "puna," the habitat provides a complex mosaic of niches that leads to the patchy distribution of many bird species throughout the region, indicating that the species have specialized habitat requirements that are only satisfied locally (Fjeldsá and Krabbe 1990, p. 32). The species' distribution is highly localized around the lake. The Junín rail apparently prefers the dense, interior marshlands comprised primarily of rushes (*Juncus* spp.) and mosaics of rushes, mosses (division Bryophyta), and low herbs in more open marsh areas (Fjeldsá 1983, p. 281). High habitat specificity is consistent with related rail species. Studies of the closely related California black rail (*Laterallus jamaicensis coturniculus*) indicate that this species is a habitat specialist, whereby the emergent vegetation used for cover, water depth, and access to upland vegetation, are all important factors in the black rail's habitat use (Flores and Eddleman 1995, p. 362). Therefore, the Junín rail's actual range is clearly smaller than the figure that continues to be reported by BirdLife International since 2000 (BLI 2000, p. 170; BLI 2007, p. 1; BLI 2008, p. 3; BLI 2009b, p. 1).

Population Estimates

Rigorous population estimates have not been made (Fjelds  1983, p. 281), and the species' elusiveness makes it difficult to locate (BLI 2009b, p. 2). In 1983, the Jun n rail was characterized as possibly common, based on local fishers' sightings of groups of up to a dozen birds at a time (Fjelds  1983, p. 281). The species continues to be reported as "fairly common," assuming that it occurs throughout the marshland surrounding the lake (BLI 2007, p. 1; BLI 2009b, p. 1). The BirdLife International estimate that this species' population size falls within the population range category of 1,000–2,499 (BLI 2000, p. 170; BLI 2007, p. 1; BLI 2009b, p. 1). This estimate is an extrapolation that continues to be based on the assumption that the species "may be fairly common in the entire c. 58 mi² (150 km²) of available marshland" around Lake Jun n (BLI 2000, p. 170; BLI 2007, p. 1; BLI 2008, p. 3; BLI 2009b, p. 1). As indicated in the analysis of this species' Current Range and Distribution, the species has never been confirmed outside its two known localities and, therefore, it is possible that the species is locally common, but not widely distributed. If the Jun n rail is not common throughout Lake Jun n's marshland, the actual population size may be much lower.

The species has experienced a population decline of between 10 and 19 percent in the past 10 years (BLI 2009b, p. 2). The population is considered to be declining in close association with continued habitat loss and degradation (see Factors A, C, and E) (BLI 2008, p. 1).

Conservation Status

The Jun n rail is considered "endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276855). The IUCN categorizes the Jun n rail as "Endangered" because it is known only from a small area of marshland (i.e., near Ondores and Pari) around a single lake, where habitat quality is declining (BLI 2008, p. 3). The single known population of the Jun n rail occurs wholly within one protected area in Peru, the Jun n National Reserve (BLI 2008, p. 1; BLI 2009b, pp. 1-2).

Summary of Factors Affecting the Jun n Rail

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

The Jun n rail is endemic to Lake Jun n, where it resides year-round and

is restricted to two localities within the shallow marshlands encircling Lake Jun n (BLI 2008, p. 3; BLI 2009b, p. 2; Fjelds  1983, p. 278). The current estimated range of the species, 62 mi² (160 km²) (BLI 2009b, p. 1), is an overestimate of this species' range for the reasons outlined above as part of the analysis of this species' Current Range and Distribution. The species is known only from two discrete locations, near Ondores and Pari, on the southwest shore of the lake. Breeding occurs near the end of the dry season, in September and October, and the birds build their nests on the ground within the dense vegetative cover of the rushes that make up the marshland perimeter of the lake (BLI 2000, p. 170; BLI 2007, p. 1; BLI 2009b, p. 2).

The habitat in and around Lake Jun n is subjected to manmade activities that have altered, destroyed, and degraded the quantity and quality of habitat available to the Jun n rail. These activities include: (1) artificial manipulation of water levels; (2) water contamination; and (3) plant harvesting in the species' breeding grounds. The negative impacts of these activities are accentuated by unpredictable climate fluctuations (such as droughts or excessive rains) (Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027). The Upamayo Dam went into operation at the northwest end of Lake Jun n in 1936 to generate electricity using hydropower (Martin *et al.* 2001, p. 178; ParksWatch 2006, p. 5). Under normal conditions, water levels are lower during the dry season, and the marshlands can become partially or completely desiccated (BLI 2009b, p. 1; ParksWatch 2009, p. 2). The dam is often opened during the dry season, to generate power (June to November) (BLI 2009b, p. 1; ParksWatch 2009, p. 2), leading to further drawdown of the lake. Lake drawdown has been known to cause water levels to fluctuate seasonally up to 6 ft (2 m) (Martin and McNee 1999, p. 659) and has at times caused complete desiccation of the marshlands by the end of the dry season (Fjelds  2004, p. 123). The ground nesting Jun n rail breeds near the end of the dry season, in September and October, and the species' relies on the dense vegetative cover of the rushes on the lake perimeter in which to build their nests (BLI 2009b, p. 2). A similar species, the California black rail, may tolerate decreases in water depth, but only if the substrate remains moist enough to support sufficient wetland vegetation (Flores and Eddleman 1995, p. 362). Eddleman *et al.* (1988, p. 463) noted that water drawdown before

nesting season disrupts nest initiation by rails. Therefore, water drawdown near the end of the dry season that results in complete desiccation of the shallow marshlands (BLI 2009b, p. 1; ParksWatch 2009, p. 2) is likely to disrupt Jun n rail nest initiation.

Experts believe that the Jun n rail is restricted to the marshes at the southwest corner of the lake because of the high level of contamination at the northwest margins of the lake (Martin and McNee 1999, p. 662). Experts also believe that pollution and artificial water level fluctuations will continue to have adverse consequences for the vegetation surrounding the lake and, therefore, the Jun n rail (BLI 2000, p. 170; BLI 2007, p. 1; J. Fjelds  in litt., 1987, as cited in Collar *et al.* 1992, p. 190). Indeed, in some places, the tall marshlands, which rely on inundated soils to thrive, have virtually disappeared because the reed-beds are no longer permanently inundated (O'Donnell and Fjelds  1997, p. 30). Moreover, as the marshes dry, livestock (primarily sheep (*Ovis aries*), but also cattle (*Bos taurus*), and some llamas (*Llama glama*) and alpacas (*Llama pacos*)) move into the desiccated wetlands surrounding the lake to graze. Overgrazing is a year-round problem around Lake Jun n because the entire lakeshore is zoned for grazing by a large number of livestock (approximately 60,000–70,000 head) (ParksWatch 2006, pp. 12, 19). During the dry season, the hoofed stock moves into the marshlands to graze, compacting the soil and trampling the vegetation (ParksWatch 2006, p. 31). Increased access to the wetlands during the end of the dry season, which coincides with the inception of the Jun n rail's nesting season, likely disrupts the rail's nesting activities or leads to nest trampling. Therefore, activities that increase lakeshore access, such as water drawdown, decrease the amount of available habitat for the Jun n rail (for nesting and feeding) and are likely to negatively impact the Jun n rail's reproduction (through trampling) and mating habits (through disturbance) (BLI 2009b, p. 1).

Water quality is another factor influencing the quality of habitat available to the Jun n rail. The degraded water quality in Lake Jun n was fully discussed as part of the Factor A analysis for the Jun n grebe and is summarized here. The water in Lake Jun n has been contaminated from mining (Martin and McNee 1999, pp. 660-661; ParksWatch 2006, p. 2; Shoobridge 2006, p. 3), agricultural activities (Martin and McNee 1999, pp. 660-661; ParksWatch 2006, p. 2;

Shoobridge 2006, p. 3), and from organic matter and wastewater runoff from local communities around the lake (ParksWatch 2006, pp. 5, 19; Shoobridge 2006, p. 3). Water pollution has resulted in heavy metal contamination throughout the lake, exceeding established thresholds for aquatic life throughout at least one-third of the lake (Martin and McNee 1999, pp. 660-661; O'Donnel and Fjelds  1997, p. 29; ParksWatch 2006, pp. 3, 20; Shoobridge 2006, p. 3), and rendering the northern portion of the lake lifeless (BLI 2008, p. 4; Shoobridge 2006, p. 3; Fjelds  2004, p. 124; Martin and McNee 1999, pp. 660-662; ParksWatch 2006, pp. 20-21). At the lake's center, lake bottom sediments are lifeless and anoxic due to contaminants (Fjelds  2004, p. 124; Martin *et al.* 2001, p. 180), and the lakeshore has become polluted with "toxic acid gray sediment" (O'Donnel and Fjelds  1997, p. 30). There is no vegetation at the northern end of the lake (Fjelds  2004, p. 124; ParksWatch 2006, pp. 20-21), and ongoing contamination has the potential to reduce vegetative cover in other areas of the lake, including the marshlands where the Jun n rail occurs. In addition, these pollutants have severely affected animal and plant populations in the area, contributing to mortality of species around the lake (ParksWatch 2006, pp. 3, 20) and have the potential to reduce the health and fitness of the Jun n rail (see Factor C).

Local residents also harvest and burn cattails from the marshland habitat, which the Jun n rail depends upon. Cattails are harvested for use in construction (i.e., to assemble rafts, baskets, and mats) and as forage for livestock (ParksWatch 2006, p. 23). Cattails are also burned to encourage shoot renewal (ParksWatch 2006, p. 23) and for hunting the montane guinea pig (*Cavia tschudii*), which seeks cover in the cattail marshes and is part of the local diet. Burning cattail communities has a negative and long-lasting impact on species that use the cattails as permanent habitat (INRENA 2000, as cited in ParksWatch 2006, p. 22; Eddleman *et al.* 1988, p. 464), including the Jun n rail, which relies on the dense vegetative cover of the marshlands for year-round residence and nesting (BLI 2000, p. 170; BLI 2007, p. 1; BLI 2009b, p. 2).

The negative impacts of habitat alteration and destruction (such as artificially reduced water levels, water contamination, and cattail harvesting and burning) are accentuated by unpredictable climate fluctuations (such as droughts or excessive rains) (Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.*

2007, p. 1027). Peru is subject to unpredictable climate fluctuations, such as those that are related to the El Ni o Southern Oscillation (ENSO). Changes in weather patterns, such as ENSO cycles (El Ni o and La Ni a events), tend to increase precipitation in normally dry areas, and decrease precipitation in normally wet areas (Holmgren *et al.* 2001, p. 89; TAO Project n.d., p. 1). ENSO events exacerbate the effects of habitat reduction and alteration on the decline of a species (England 2000, p. 86; Holmgren *et al.* 2001, p. 89; Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027; Parmesan and Mathews 2005, p. 334; Plumart 2007, pp. 1-2; Timmermann 1999, p. 694), particularly for narrow endemics (Jetz *et al.* 2007, p. 1213) such as the Jun n rail (see also Factor E). As discussed above, droughts increase access to the wetlands where Jun n rails live and breed. Excessive rain increases contamination in the water and causes soil toxicity (see Factor C). ENSO cycles are ongoing, having occurred several times within the last decade (NWS 2009, p. 2). Evidence suggests that ENSO cycles have already increased in periodicity and severity (Richter 2005, pp. 24-25; Timmermann 1999, p. 694), which will, thus, exacerbate the negative impacts on a species.

Summary of Factor A

The habitat in and around Lake Jun n, where the Jun n rail is endemic, has been and continues to be altered and degraded as a result of human activities, including artificial water level fluctuations to generate hydropower, water contamination caused by mining waste, agricultural and organic runoff from surrounding lands, and wastewater from local area communities. The Jun n rail is dependent on the marshland habitat surrounding the lake for breeding and feeding. Water levels in Lake Jun n are manipulated to generate electricity, which leads to dramatic fluctuations in water levels of up to 6 ft (1.8 m). The Jun n rail nests on the ground, within the protective cover of the marshlands. As water drawdown occurs near the end of the dry season and during the inception of the Jun n rail's mating season, portions of the marshlands may dry out completely, affecting the availability of suitable breeding and foraging habitat. This species' population decline has been linked to deteriorating habitat quality (see also Factor E). Overgrazing, cattail harvest, and burning are ongoing around the lakeshore, and water drawdown increases access to the marshlands. Severe water contamination in the

northwest portion of the lake has rendered it lifeless, and experts believe that water contamination limits the Jun n rail's foraging and breeding activities to the southern portion of the lake. The effects of artificially reduced water levels and water contamination are accentuated by droughts or excessive rains caused by El Ni o events. Reduced water levels near the end of the dry season (during Jun n rail nesting season) expose the species to greater vulnerability to predation (see Factor C), which also heightens the risks to the species that are associated with short- and long-term genetic viability (Factor E). Therefore, we find that destruction and modification of habitat are threats to the continued existence of the Jun n rail throughout its range.

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

We are not aware of any information currently available that indicates that overutilization of Jun n rail for commercial, recreation, scientific, or education purposes has occurred or is occurring at this time. As a result, we are not considering overutilization to be a threat to the continued existence of the Jun n rail.

C. Disease or Predation

Disease: Although no specific disease threat has been identified for the Jun n rail, contamination of Lake Jun n exposes the Jun n rail to mortality and a reduction the overall fitness and health of the species. The effects of water contamination on the health of species inhabiting Lake Jun n were discussed as part of the analysis of Factor C for the Jun n grebe and are summarized here. In Lake Jun n, mining activities (Martin and McNee 1999, pp. 660-661; Shoobridge 2006, p. 3), and agricultural runoff, organic matter, and wastewater (ParksWatch 2006, pp. 5, 19; Shoobridge 2006, p. 3) have contaminated the entire lake with high concentrations of dissolved chemicals (Fjelds  2004, p. 124; Martin and McNee 1999, pp. 660-662; ParksWatch 2006, pp. 20-21). Environmental contaminants exceed current established thresholds for aquatic life (Martin and McNee 1999, pp. 660-661; ParksWatch 2006, p. 20) and have rendered the northern portion of the lake lifeless from eutrophication (BLI 2008, p. 4; Shoobridge 2006, p. 3). Due to severe contamination, the sediments in the center of the lake are anoxic (Martin *et al.* 2001, p. 180), and the lake's turbidity has increased (ParksWatch 2006, p. 20). Chemical waste has damaged at least one third of the lake, severely affecting animal and

plant populations in the area (O'Donnel and Fjelds  1997, p. 29; ParksWatch 2006, pp. 3, 20; Shoobridge 2006, p. 3) and completely eliminating vegetation from the northern portion of the lake (Fjelds  2004, p. 124; ParksWatch 2006, pp. 20-21). It is also believed that contamination may, in fact, be responsible for the possible extirpation of at least one fish species (a catfish) (O'Donnel and Fjelds  1997, p. 29).

Contamination from mining waste may have direct and indirect impacts on the fitness and health of the Jun n rail. As described above as part of the Factor C analysis for the Jun n grebe, a waterbird that is sympatric with the Jun n rail, mine waste contamination may have caused heavy metal poisoning (T. Valqui and J. Barrio in litt. 1992, as cited in Collar *et al.* 1992, pp. 45, 190) or reduced prey availability (Fjelds  2004, p. 124), leading to Jun n grebe mortality during an extensive drought from 1989 to 1992. Large-scale bird mortality has occurred on the lake, apparently due to lead poisoning from mining effluents—a common cause of mortality in waterbirds (Friend and Franson 1999, p. 317; O'Donnel and Fjelds  1997, p. 30). Heavy metals in the water column and the lake's sediments, where this species feeds, would have negative health consequences for the Jun n rail, as in the case for the Jun n grebe and other waterbirds that inhabit the lake. Excessive contaminant load can contribute to a decline in fitness and vigor for long-lived, mid-trophic level species (Rowe 2008, p. 624), such as the Jun n rail. Increased turbidity of the water has resulted in die-offs of aquatic plants and algae, which disrupts the food chain (ParksWatch 2006, p. 20). Higher trophic level species (discussed in more detail as part of the Factor C analysis for the Jun n grebe), such as the Jun n rail, are more susceptible to disruptions in the food chain at lower trophic levels (Fjelds  2004, p. 123; The University of the Western Cape 2009, p. 1) and prone to bioaccumulation because they ingest pesticides, heavy metals, and other contaminants that are present in their prey (Demayo *et al.* 1982, as cited in Eisler 1988, p. 5; Fjelds  2004, p. 123). Drought conditions exacerbate the effects of water contamination and bioaccumulation for species at higher trophic levels (Demayo *et al.* 1982, as cited in Eisler 1988, p. 5; Fjelds  2004, p. 123).

Predation: Predators around Lake Jun n include the Andean fox (*Pseudalopex culpaeus*), the long-tailed weasel (*Mustela frenata*), Pampas cat (*Onicifelis colocolo*), and hog-nosed skunk (*Conepatus chinga*) (ParksWatch

2009, p. 4). Jun n rails are preyed upon by pampas cats (BLI 2008, p. 4; BLI 2009b, p. 2). Under normal conditions, water levels are lower in the dry season and the marshlands can become partially or completely dry (BLI 2009b, p. 1; ParksWatch 2009, p. 2) reducing protective cover and allowing predators to more easily locate the rail. When the floodgates of the Upumayo Dam are opened during the dry season (June to November) (BLI 2009b, p. 1; ParksWatch 2009, p. 2), drawdown has led to complete desiccation of the marshlands by the end of the dry season (Fjelds  2004, p. 123). The ground nesting Jun n rail breeds near the end of the dry season, in September and October, and builds their nests in the dense vegetative cover of the rushes on the lake perimeter (BLI 2009b, p. 2). Water drawdown and periods of drought increases the bird's vulnerability to predation because nesting grounds become exposed and larger areas of the marsh are accessible to predators (ParksWatch 2006, p. 23). Predation increases the risk of extirpation due to the species' already small population size. In addition, species that inhabit a small geographic range, occur at low density, occupy a high trophic level, and exhibit low reproductive rates tend to have a higher risk of extinction than species that are not limited by the same risk factors (Purvis *et al.* 2000, p. 1949) (Factor E).

Summary of Factor C

Environmental contaminants (Factor A) in Lake Jun n may have negative consequences on the health of the Jun n rail, given that extensive environmental contamination in Lake Jun n has resulted in mortality of flora and fauna that inhabit the lake and its margins. The species' trophic level also exposes it to bioaccumulation of toxins accumulated in the tissue of prey species. There is documented evidence that other waterbirds occupying the same habitat have died as a direct result of contaminant poisoning or reduction of the availability of prey species. Therefore, we find that disease is a threat to the continued existence of the Jun n rail.

Predation by the pampas cat results in the direct removal of individuals from the population and can remove potentially reproductive adults from the breeding pool. Ongoing habitat destruction (through reduced water levels and contamination) continues to degrade the quality of habitat available to the Jun n rail (Factor A) and the species' habitat becomes more accessible to predators during droughts and water drawdowns. Predation

renders the species particularly vulnerable to local extirpation due to its small population size (Factor E). Therefore, we find that predation, exacerbated by ongoing habitat destruction (Factor A), are threats to the continued existence of the Jun n rail throughout its range.

D. Inadequacy of Existing Regulatory Mechanisms

The Jun n rail is listed as "endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276855). This Decree prohibits hunting, take, transport, and trade of protected species, except as permitted by regulation. As hunting, take, transport, and trade do not currently threaten the Jun n rail, this regulation does not mitigate any current threats to this species.

Peru has several categories of national habitat protection, which were described above as part of the Factor D analysis for the ash-breasted tit-tyrant (BLI 2008, p. 1; IUCN 1994, p. 2; Rodr guez and Young 2000, p. 330). The single Jun n rail population occurs wholly within the Jun n National Reserve (Jun n, Peru) (BLI 2009b, pp. 1-2), which encompasses the lake and surrounding land and was established in 1974 by Supreme Decree 0750-74-AG (BLI 2009a, p. 2; Wege and Long, p. 264). Peruvian National Reserves are created for the sustainable extraction of certain biological resources (BLI 2008, p. 1; Rodr guez and Young 2000, p. 330), and most of the lakeshore is designated a "Direct Use Zone," allowing fishing, grazing, and other educational, research, and recreational activities (ParksWatch 2006, p. 12). Habitat destruction and alteration (through artificial water level fluctuations, contamination (BLI 2009b, p. 1; Fjelds  2004, p. 124; ParksWatch 2006, pp. 20-21; Wege and Long 1995, p. 264)), overgrazing, and cattail harvest and burning (ParksWatch 2006, pp. 22-23) are ongoing throughout the Reserve (Factor A), increasing the species' susceptibility to predation (ParksWatch 2006, p. 23) (Factor C), and jeopardizing the continued existence of the species, given its already small population size (Factor E). Therefore, the existence of this species within a protected area has not reduced or mitigated the threats to the Jun n rail.

The Jun n National Reserve was designated a Ramsar site in 1997 (BLI 2009a, p. 2; INRENA 1996, pp. 1-14; Ramsar 2009, p. 2). As more fully described for the Jun n grebe, this designation provides only nominal protection of wetland habitat (Jellison *et*

al. 2004, p. 19). Activities that negatively impact the Junín rail are ongoing throughout this wetland, including water fluctuations and contamination (Factor A), water fluctuations that increase the species' risk of predation (Factor C), and small population size (Factor E). Therefore, the Ramsar designation has not mitigated the impact of threats on the Junín rail.

There are approximately 5,000 laws and regulations directly or indirectly related to environmental protection and natural resource conservation in Peru. Recent studies by the Peruvian Society for Environmental Law (SPDA) have concluded that many of these are not effective because of limited implementation or enforcement capability (Muller 2001, pp. 1-2).

Summary of Factor D

Peru has enacted various laws and regulatory mechanisms for the protection and management of wildlife and their habitats. The Junín rail is "endangered" under Peruvian law, and its entire population occurs within a protected area. As discussed under Factor A, habitat destruction and alteration have curtailed the species' range and threaten the continued existence of the species. Ongoing habitat destruction (including water level manipulation, contamination, overgrazing, and cattail harvest and burning (Factor A)), predation (Factor C), and predators' increased access due to habitat destruction intensify the risks to the species from its already small population size (Factor E). These activities are ongoing throughout the species' range, which is entirely encompassed within a protected area. Thus, despite its endangered status and its presence within a designated protected area, laws governing wildlife and habitat protection in Peru are inadequately enforced or ineffective at protecting the species or mitigating ongoing habitat degradation (Factor A) and concomitant population declines (Factor E). Therefore, we find that the existing regulatory mechanisms are inadequate to mitigate the threats to the continued existence of the Junín rail throughout its range.

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

An additional factor that affects the continued existence of the Junín rail is the species' small population size. As discussed above (see Population Estimates), BirdLife International has placed the Junín rail in the population category of between 1,000 and 2,499

individuals (BLI 2009b, p. 2), and considers the population to be likely "very small and presumably declining" (BLI 2000, p. 170; BLI 2009b, p. 1).

Small population size renders species vulnerable to genetic risks that can have individual or population-level genetic consequences, such as inbreeding depression, loss of genetic variation, and accumulation of new mutations, and may affect the species' viability by increasing its susceptibility to demographic shifts or environmental fluctuations, as explained in more detail above in the Factor E analysis for the ash-breasted tit-tyrant (Charlesworth and Charlesworth 1987, p. 238; Pimm *et al.* 1988, pp. 757, 773-775; Shaffer 1981, p. 131). Small population size leads to a higher risk of extinction and, once a population is reduced below a certain number of individuals, it tends to rapidly decline towards extinction (Frankham 1996, p. 1507; Franklin 1980, pp. 147-148; Gilpin and Soulé 1986, p. 25; Holsinger 2000, pp. 64-65; Purvis *et al.* 2000, p. 1949; Reed and Frankham 2003, pp. 233-234; Soulé 1987, p. 181). In addition, species that inhabit a small geographic range, occur at low density, occupy a high trophic level, and exhibit low reproductive rates tend to have a higher risk of extinction than species that are not limited by the same risk factors (Purvis *et al.* 2000, p. 1949). We consider that the risks associated with small population size will continue to impact this species and may accelerate, if habitat destruction continues unabated.

Complications arising from the species' small population size are exacerbated by its restricted range and the threat of predation (Factor C). The Junín rail is known from two localities (Ondores and Pari) on the southwest shore of a single Andean lake, Lake Junín, in central Peru (BLI 2000, p. 170; BLI 2009b, pp. 1-2; Fjeldså 1983, p. 281; Fjeldså in litt. 1992, as cited in Collar *et al.* 1992, p. 190) (see Current Range and Distribution). The population has declined at a rate between 10 and 19 percent in the past 10 years, and this decline is expected to continue, as a result of the declining quality of habitat within its small, restricted range (BLI 2009b, pp. 4-5) (Factor A). The pampas cat is a known predator of Junín rails (BLI 2008, p. 4; BLI 2009b, p. 2). The ground nesting Junín rail is particularly vulnerable to predation near the end of the dry season, when nesting begins and the vegetative cover of the marshlands within which they build their nests (BLI 2009b, p. 2) is more exposed and accessible to predators (ParksWatch 2006, p. 23). The Junín rail's small population size, combined with its

restricted range and threat of predation, increases the species' vulnerability to adverse natural events and manmade activities that destroy individuals and their habitat (Holsinger 2000, pp. 64-65; Primack 1998, pp. 279-308; Young and Clarke 2000, pp. 361-366).

Summary of Factor E

The Junín rail has a small population size that renders it vulnerable to genetic risks that negatively impact the species' viability. The species occupies a restricted range and occurs in habitat that continues to be altered and destroyed due to human activities (Factor A). Predation jeopardizes the species' already small population size because it results in the direct removal of Junín rail individuals from the population, can remove potentially reproductive adults from the breeding pool, and could lead to extirpation (Factor C). The small population size, as well as its restricted range and threat of predation, increases the species' vulnerability to extinction through demographic or environmental fluctuations. Based on the species' small population size, restricted range, and threat of predation, we have determined that the Junín rail is particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., water level manipulation, contamination, cattail harvest, and overgrazing) that destroy individuals and their habitat. The genetic and demographic risks associated with small population sizes are exacerbated by ongoing human activities that continue to curtail the species' habitat throughout its range. The species' population has declined and is predicted to continue declining due to an ongoing reduction in the quality of its habitat (Factor A). Therefore, we find that the species' small population size, in concert with its restricted range, threat of predation, and its heightened vulnerability to adverse natural events and manmade activities, are threats to the continued existence of the Junín rail throughout its range.

Status Determination for the Junín Rail

The Junín rail is a ground nesting bird endemic to Lake Junín, found at 13,390 ft (4,080 m) above sea level in Peru, where it resides year-round. The species has high habitat specificity and occurs only in two localities within the marshland mosaic habitat that surrounds the lake. The current estimated range of the species is 62 mi² (160 km²), and its population size is estimated to be 1,000–2,499. However,

both of these figures are likely to be overestimates; despite suggestions that the species inhabits the entire area of marshland surrounding the lake, the species has only been confirmed in two localities.

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the Junín rail and have concluded that there are four primary factors that threaten the continued existence of the rail: (1) Habitat destruction, fragmentation, and degradation; (2) disease and predation; (3) limited size and isolation of remaining populations; and (4) inadequate regulatory mechanisms.

Human activities that degrade, alter, and destroy habitat are ongoing throughout the Junín rail's range. Lake waters are artificially manipulated to produce hydropower, resulting in reductions in water levels that impact the species' nesting and foraging sites, especially near the end of the dry season when this ground nesting bird begins breeding. Reduced water levels and aquatic contaminants have permanently destroyed segments of cattail communities, compromising the amount of suitable flooded marshland available for nesting. Climate fluctuations exacerbate the effects of habitat alteration, as droughts further decrease available nesting grounds and excessive rains increase contaminant runoff into the lake (Factor A). Environmental contaminants have caused mortality in other waterbirds and have likely reduced the overall general health of the Junín rail population (Factor C). Overgrazing and cattail harvest and burning further alter and destroy the marshland upon which the rails depend (Factor A). Water drawdown exposes the birds and their marshland nesting areas to greater access by grazing animals, cattail harvesters, and predators (Factors A and C).

Junín rail habitat continues to be altered by human activities, which results in the continued degradation and destruction of habitat and reduces the quality and distribution of remaining suitable habitat. Habitat loss was a factor in the Junín rail's historical decline (see Historical Range and Distribution), and the species is considered to be declining today due to continued reduction of available habitat (Factors A and E). The Junín rail population is small, increasing the species' vulnerability to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., water contamination, water level manipulation, cattail harvest, and

overgrazing) that destroy individuals and their habitat. Human activities that continue to curtail the species' habitat throughout its range exacerbate the genetic and demographic risks associated with small population sizes (Factor E). Predation jeopardizes the species' already small population size because it results in the direct removal of Junín rail individuals from the population, can remove potentially reproductive adults from the breeding pool, and could lead to extirpation (Factor C). The Junín rail population has declined at a rate between 10 and 19 percent during the past 10 years (see Population Estimates), and this decline is predicted to continue commensurate with ongoing threats from habitat destruction, water contamination, overgrazing, and cattail harvest and burning (Factor A).

Despite the species' endangered status in Peru and its occurrence entirely within one protected area (Factor D), habitat destruction and degradation continue as a result of human activities that alter lake levels and compromise water quality and increase the species' susceptibility to overgrazing and predation (Factors A and C). Therefore, regulatory mechanisms are either inadequate or ineffective at mitigating the existing threats to the species and its habitat (Factor D).

Section 3 of the Act defines an "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range" and a "threatened species" as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Based on the immediate and ongoing threats to the Junín rail throughout its entire range, as described above, we determine that the Junín rail is in danger of extinction throughout all of its range. Therefore, on the basis of the best available scientific and commercial information, we are proposing to list the Junín rail as an endangered species throughout all of its range.

IV. Peruvian plantcutter (*Phytotoma raimondii*)

Species Description

The Peruvian plantcutter, locally known as "*cortarrama Peruana*," is a small finch-like bird endemic to the dry forests of coastal northwest Peru (Collar *et al.* 1992, p. 805; Goodall 1965, p. 636; Ridgely and Tudor 1994, p. 733; Sibley and Monroe 1990, p. 371; Schulenberg *et al.* 2007, p. 488; Walther 2004, p. 73). The Peruvian plantcutter is an herbivore with a predominantly leaf-eating diet

(Bucher *et al.* 2003, p. 211; Schulenberg *et al.* 2007, p. 488; Walther 2004, p. 73).

Plantcutters have bright yellow eyes, short wings and rather long tails, and their crown feathers form a slight crest (Goodall 1965, p. 635; Ridgely and Tudor 1994, p. 732). Adult birds are 7.28 to 7.48 in (18.5 to 9 cm) in length and weigh approximately 1.26 to 1.55 ounces (oz) (36 to 44 grams (g)) (Schulenberg *et al.* 2007, p. 488; Walther 2004, p. 73). Males are pale ashy gray, except a broad cinnamon-rufous color band on the belly and above the bill, and white colored bands on their wings (BLI 2000, p. 401; BLI 2009a, p. 1; Goodall 1965, p. 636; Ridgely and Tudor 1994, p. 733). Females are buff-brown with broad, dark brown stripes above, and white with heavy black-striped underparts (BLI 2009a, p. 1; Collar *et al.* 1992, p. 805). Juvenile birds have not been described (Walther 2004, p. 73). The Peruvian plantcutter's bill is stout, short, conical, and finely serrated with sharp tooth-like projections that run the length of the beak on both sides, and which are well suited for plucking buds, leaves, shoots, and fruits (Goodall 1965, p. 635; Ridgely and Tudor 1994, p. 732; Schulenberg *et al.* 2007, p. 488) (see Habitat and Life History).

Taxonomy

The Peruvian plantcutter was first taxonomically described as *Phytotoma raimondii* by Taczanowski in 1883 (ITIS 2009, p. 1; Sibley and Monroe 1990, p. 371). The type-specimen of the Peruvian plantcutter (the specimen that was described by Taczanowski) was collected by the ornithologist Konstanty Jelski, who recorded the specimen as being collected in the Tumbes Department of Peru (Flanagan *et al.* in litt. 2009, p. 2). However, the reported collection location may have been inaccurate (see Historical range and Distribution, below).

The genus *Phytotoma* contains three species of plantcutters, all endemic to South America (Dickinson 2003, p. 346; Goodall 1965, p. 635; Sibley and Monroe 1990, p. 371; Walther 2004, p. 73). Ornithologists have long debated to which family this genus belongs. Some ornithologists have recommended that the genus be placed in its own family, Phytotomidae (Lanyon and Lanyon 1989, p. 422), while others placed the genus within the Tyrannidae family (Sibley and Monroe 1990, p. 371). Molecular research using DNA sequencing supports the inclusion of *Phytotoma* in the Cotingidae family (Irestedt *et al.* 2001, p. 23; Johansson *et al.* 2002, p. 993; Ohlson *et al.* 2006, p. 10). The Cotingidae family includes a

wide diversity of species that are found in Central America and tropical South America (Snow 2004, p. 32). Therefore, based on the information currently available to us, we accept that the Peruvian plantcutter belongs to the Cotingidae family, which also follows the Integrated Taxonomic Information System (ITIS 2009, p. 1).

Habitat and Life History

The Peruvian plantcutter is reportedly selective in its habitat preference and requires a variety of arid tree and shrub species with dense low-hanging branches to the ground (Collar *et al.* 1992, p. 805; Flanagan and More 2003, p. 5; Flanagan *et al.* in litt. 2009, p. 7; Williams 2005, p. 2). The primary habitat for the Peruvian plantcutter is seasonally dry tropical forest, which is also referred to equatorial dry tropical forest, and which occurs in the semiarid lowlands of northwest Peru (Linares-Palomino 2006, pp. 260, 263-266; Schulenberg *et al.* 2007, p. 21; Walther 2004, p. 73). The Peruvian plantcutter also uses arid lowland scrub (dense and open) and dense riparian shrub communities (BLI 2009a, p. 2; Collar *et al.* 1992, p. 805; Schulenberg *et al.* 2007, pp. 21, 488; Stotz *et al.* 1996, p. 19; Walther 2004, p. 73). Stotz *et al.* (1996, pp. 19, 428) identified the Peruvian plantcutter as a key indicator species for Equatorial Pacific Coast arid lowland scrub. The lowland dry tropical forest and scrub are characterized as small and heavily fragmented patches of plant species adapted to the arid conditions of the prolonged dry season of northwest Peru (Best and Kessler 1995, p. 40; Bridgewater *et al.* 2003, pp. 132, 140; Ridgely and Tudor 1994, p. 734).

The lowland dry forest or woodlands in northwest Peru are open canopied, with trees occurring in scattered clumps or individually (Flanagan and More 2003, p. 4). The dominant tree species of the lowland dry forest is *Prosopis pallida* (common name “kiawe;” also locally referred to as “algarrobo”) in the Fabaceae family (legume family) (Lopez *et al.* 2005, p. 542; More 2002, p. 39). *Prosopis pallida* is a wide-spreading tree or large shrub, 26–65 ft (8–20 m) tall, with dense branches; spines can be present or absent (Pasicznik *et al.* 2001, p. 36). This deep-rooted drought-tolerant species, related to mesquite species of southwest United States and Mexico, provides an important ecological function by improving and stabilizing soil conditions (Brewbaker 1987, p. 1; Pasicznik *et al.* 2001, pp. 101-102). Typical of legumes, *P. pallida* is able to “fix” atmospheric nitrogen for plant utilization and growth (Brewbaker 1987, p. 1; Pasicznik *et al.* 2001, p. 3).

Three of the most common tree species associated with *P. pallida* dry forest habitat used by the Peruvian plantcutter are *Capparis scabrida* (no common name, but locally known as “sapote”), in the Capparaceae (caper) family, and *Acacia macracantha* (long-spine acacia, locally known as “faique”) and *Parkinsonia aculeata* (Jerusalem thorn, locally known as “palo verde”), both in the Fabaceae (legume) family (More 2002, pp. 17-23). Associated flowering shrubs in dry forest habitat include *Capparis avicennifolia* (no common name, but locally known as “bichayo”) and *C. crotonoides* (no common name, but locally known as “guayabito de gentil”), both in the Capparaceae (caper) family; *Cordia lutea* (no common name, but locally known as “overall”) in the Boraginaceae (borage) family; and *Maytenus octogona* (no common name, but locally known as “realengo”) in the Celastraceae (bittersweet) family. Other commonly occurring dry forest vegetation includes vines (e.g., Convolvulaceae (morning-glory) and Cucurbitaceae (gourd) families), *Psittacanthus chanduyensis* (tropical mistletoe; locally known as “suela con suela”) in the Loranthaceae (mistletoe) family, scattered herbaceous species (e.g., Asteraceae (sunflower), Scrophulariaceae (figwort), and Solanaceae (nightshade) families), and grasses (e.g., Poaceae (grass) family) (Elton 2004, p. 2; Ferreyra 1983, pp. 248-250; More 2002, pp. 14-17; Walther 2004, p. 73). Riparian vegetation includes dense shrub and small trees of *P. pallida*, *A. macracantha*, *Capparis* spp., and *Salix* spp. (willow spp.) (Lanyon 1975, p. 443).

The arid climate of northwest Peru is due to the influence of the cold Humboldt Current that flows north, parallel to the Peruvian Coast (UNEP 2006, p. 16; Linares-Palomino 2006, p. 260; Rodriguez *et al.* 2005, p. 2). The Humboldt Current has a cooling influence on the climate of coastal Peru, as the marine air is cooled by the cold current and, thus, is not conducive to generating rain. To the east, the Andean Mountains prevent humid air from the Amazon from reaching the western lowlands (Lanyon 1975, p. 443; Linares-Palomino 2006, p. 260).

Coastal northwest Peru experiences a short rainy season during the summer months (January–April) (Linares-Palomino 2006, p. 260), which can also include precipitation in the form of mist or fine drizzle along the coast (Lanyon 1975, p. 443). The mean annual precipitation across the range of the Peruvian plantcutter is 0.196 to 3.80 in (5.0 to 99 mm) (hyper-arid to arid)

(Galan de Mera *et al.* 1997, p. 351). The climate is warm and dry with the annual temperature range of 74° to 77° F (23° to 25° C) at elevations below 1,968 ft (600 m) (Linares-Palomino 2006, p. 260). Northwest Peru is strongly influenced by the El Niño Southern Oscillation (ENSO) cycle (Rodriguez *et al.* 2005, p. 1), which can have particularly profound and long-lasting effects on arid terrestrial ecosystems (Holmgren *et al.* 2006a, p. 87; Mooers *et al.* 2007, p. 2) (see Factor A).

Knowledge of the breeding of most species within the Cotingidae family, including the Peruvian plantcutter, is not well known (Walther 2004, p. 73). The Peruvian plantcutter is considered a resident species in Peru, which indicates that it breeds there (Snow 2004, p. 61; Walther 2004, p. 73). Nesting activity of plantcutters appears to occur from March to April (Collar *et al.* 1992, p. 805; Walther 2004, p. 73). Plantcutters build shallow, cup-shaped nests that are made of thin dry twigs and lined with root fibers and other softer material (Snow 2004, p. 55). Nests range from 2.4 to 3.5 in (6 to 9 cm) in height and 3.9 to 7.0 in (10 to 18 cm) in diameter, and can be placed 3.3 to 9.8 ft (1 to 3 m) above the ground inside a thick thorny shrub or higher in the fork of a tree (Elton 2004, p. 2; Flanagan and More 2003, p. 3; Snow 2004, p. 55). Each female lays two to four eggs, and the incubation period lasts about 2 weeks (Snow 2004, p. 56; Goodall 1965, p. 636; Walther 2004, p. 73). The eggs have been described as short oval to sub-elliptical in shape and grayish olive in color with dark brownish olive spots at the larger end (Flanagan and Millen 2008, p. 1). Males assist rearing the chicks, which fledge after 17 days or so (Snow 2004, p. 56).

Plantcutters are herbivores with a predominantly leaf-eating diet (Bucher *et al.* 2003, p. 211; Snow 2004, p. 46). As an herbivore, the Peruvian plantcutter is dependent on year-round availability of high-quality food, particularly during the dry season when plant growth is very limited (Bucher *et al.* 2003, p. 216). Peruvian plantcutters eat buds, leaves, and shoots of *P. pallida* and various other trees and shrubs, as well as some fruits (e.g., mistletoe) (Goodall 1965, p. 635; Schulenberg *et al.* 2007, p. 488; Walther 2004, p. 73). The seeds, green seed pods, leaves, and flowers of *P. pallida* provide a protein-rich food source for animals (Lewis *et al.* 2006, p. 282). Research studies on the two related plantcutter species, the rufous-tailed plantcutter (*P. rara*) and the white-tipped plantcutter (*P. rutila*), showed that the herbivore diet of these

two species did not affect the energy levels of observed birds (Lopez-Calleja and Bozinovic 1999, p. 709; Meynard *et al.* 1999, p. 906; Rezende *et al.* 2001, p. 783). The Peruvian plantcutter appears to prefer to feed while perched in shrubs and trees, although individuals also have been observed foraging on the ground (Snow 2004, p. 50). Birds have been observed in pairs and small groups (Collar *et al.* 1992, p. 804; Flanagan and More 2003, p. 3; Schulenberg *et al.* 2007, p. 488; Walther 2004, p. 73).

Historical Range and Distribution

The Peruvian plantcutter is a restricted-range species that is confined to the mostly flat, narrow desert zone, which is less than 31 mi (50 km) in width (Lanyon 1975, p. 443) and runs along the coast of northwest Peru (Ridgely and Tudor 1994, p. 734; Stattersfield *et al.* 1998, p. 213; Walther 2004, p. 73). The historical range of the Peruvian plantcutter reportedly extended from the town of Tumbes, located in extreme northwest corner of Peru and approximately south to north of Lima within the Regions of Tumbes, Piura, Lambayeque, La Libertad, Ancash, and Lima (from north to south) (Collar *et al.* 1992, pp. 804-805).

The historical distribution of the Peruvian plantcutter was most likely throughout the contiguous lowland *P. pallida* dry forest and riparian vegetation, below 1,804 ft (550 m) (Collar *et al.* 1992; Williams 2005, p. 1). According to Collar *et al.* (1992, pp. 804-805), the Peruvian plantcutter is known from 14 historical sites.

The type-specimen of the Peruvian plantcutter, which was collected and labeled by Konstanty Jelski as being found in Tumbes in the late 1870s, was most likely collected south of the town of Tumbes (Flanagan *et al.* in litt. 2009, pp. 2, 15). It is unknown whether the type specimen was lost or destroyed, or if it was ever returned to Peru (Flanagan *et al.* in litt. 2009, p. 2). Today, there is good indication that the type-specimen was mislabeled as being collected in Tumbes (Flanagan *et al.* in litt. 2009, pp. 2). Although the Tumbes Region has been extensively surveyed for the Peruvian plantcutter, including the North-West Biosphere Reserve, there have never been other collections in or near the vicinity of Tumbes or other evidence to suggest that the Peruvian plantcutter ever occurred in the area (Flanagan *et al.* in litt. 2009, p. 2). Thus, it appears that the Peruvian plantcutter never occurred in the Tumbes Region.

Researchers consider the reduction in dry forest habitat to be the result of historical human activities, including extensive land clearing for agriculture,

timber and firewood extraction, charcoal production, and overgrazing. These activities have led to the reduction and severe fragmentation of dry forest habitat today (BLI 2009a, pp. 2-3; Bridgewater *et al.* 2003, p. 132; Flanagan *et al.* in litt. 2009, pp. 1-9; Lanyon 1975, p. 443; Lopez *et al.* 2006, p. 898; Pasiecznik *et al.* 2001, pp. 10, 75, 78, 95; Ridgely and Tudor 1994, p. 734; Schulenberg *et al.* 2007, p. 488; Stotz *et al.* 1998, p. 52) (see Factor A).

Current Range and Distribution

The current range of the Peruvian plantcutter is approximately 1,892 mi² (4,900 km²) (BLI 2009a, p. 1), which is between 33 and 1,804 ft (10 and 550 m) above sea level and within the Peruvian Regions of Piura, Lambayeque, Cajamarca, La Libertad, and Ancash (from north to south) (Flanagan *et al.* in litt. 2009, pp. 14-15). The species' reported range is an overestimate because BirdLife International defines a species' range as the total area within its extent of occurrence (see Current Range and Distribution of the ash-breasted tit-tyrant) (BLI 2000, pp. 22, 27). The Peruvian plantcutter's current distribution is severely fragmented and distributed amongst small, widely separated remnant patches of *P. pallida* dry forest (BLI 2009a, pp. 2-3; Flanagan *et al.* in litt. 2009, pp. 1-9; Ridgely and Tudor 1994, p. 18), which are usually heavily disturbed fragments of forest (Bridgewater *et al.* 2003, p. 132). Therefore, the species' actual range is smaller than this figure.

The Peruvian plantcutter is extirpated from 11 of its 14 historical sites due to loss of habitat or degradation of habitat (Elton 2004, p. 1; Flanagan and More 2003, p. 5; Hinze 2004, p. 1). Depending on habitat quality, it is estimated that the Peruvian plantcutter requires approximately 2.5 ac (1 ha) of habitat for suitable food and nesting sites (Flanagan and More 2003, p. 3; Flanagan *et al.* in litt. 2009, p. 7). Although the Peruvian plantcutter has been found in patches of *P. pallida* dry forest habitat that are in close proximity to agricultural lands, tracks or roads, and human settlement (Flanagan *et al.* in litt. 2009, pp. 2-7), much of the available *P. pallida* dry forest habitat is unoccupied (BLI 2000, p. 401; Schulenberg *et al.* 2007, p. 488; Snow 2004, p. 69; Walther 2004, p. 73).

Flanagan *et al.* (in litt. 2009, pp. 1-15) recently completed a comprehensive review of 53 locations where there have been documented sightings of the Peruvian plantcutter, of which the authors determined 29 sites were extant. Flanagan *et al.* (in litt. 2009, pp. 2-4, 14) reported that 17 of the 22 documented sites of the Peruvian plantcutter in the

Piura Region are extant. In this region, Talara Province contains the largest concentration of intact *P. pallida* dry forest habitat in northwest Peru and the largest subpopulation of the Peruvian plantcutter (BLI 2009a, p. 2; Flanagan *et al.* in litt. 2009, p. 3; Flanagan and More 2003, p. 5; Walther 2004, p. 73). Additionally, there are several other documented sites of the Peruvian plantcutter in the Piura Region (e.g., Manglares de San Pedro, Illescas Peninsula, and Cerro Illescas) (BLI 2009c, p. 1; Flanagan *et al.* in litt. 2009, pp. 4, 14).

Flanagan *et al.* (in litt. 2009, pp. 4-5, 14) reported a total of 13 locations of the Peruvian plantcutter in the Lambayeque Region, of which 5 are considered extant. Within the Region, there are four important areas for the Peruvian plantcutter:

(1) The Pómac Forest Historical Sanctuary (Santuario Histórico de Bosque de Pómac), designated as a protected archeological site in 2001, is comprised of 14,547 ac (5,887 ha) of *P. pallida* dry forest (BLI 2009e, p. 1; Flanagan *et al.* in litt. 2009, p. 4). The Sanctuary includes the archeological site Batán Grande, an area comprised of 1,235 ac (500 ha) of *P. pallida* dry forest (BLI 2009e, p. 1; Flanagan *et al.* in litt. 2009, p. 4).

(2) Near the small town of Rafan are remnant patches of *P. pallida* dry forest, encompassing approximately 3,706 ac (1,500 ha) (BLI 2009f, p. 1). The Rafan area has become a popular birding site for the Peruvian plantcutter (BLI 2009f, p. 1; Engblom 1998, p. 1).

(3) Murales Forest (*Bosque de Murales*), comprised of *P. pallida* dry forest, is a designated Archeological Reserved Zone (BLI 2009a, p. 3; Stattersfield *et al.* 2000, p. 402).

(4) Chaparri Ecological Reserve, comprised of 85,033 ac (34,412 ha) with *P. pallida* dry forest, is a community-owned and managed protected area (Walther 2004, p. 73). The remaining sites in the Lambayeque Region are small remnant patches of *P. pallida* dry forest and comprised of a few acres (Flanagan *et al.* in litt. 2009, pp. 4-5; Walther 2004, p. 73). The protected areas are further discussed under Factors A and D.

Flanagan *et al.* (in litt. 2009, pp. 5, 14) reported one occupied site of the Peruvian plantcutter in the Cajamarca Region, consisting of approximately 14.8 ac (6 ha) of remnant *P. pallida* dry forest in the Río Chicama Valley. Six of the 12 known sites of the Peruvian plantcutter in the La Libertad Region are considered extant (Flanagan *et al.* in litt. 2009, pp. 5-6, 14). Each of these sites consists of small patches of remnant *P.*

pallida dry forest habitat (Walther 2004, p. 73; Flanagan *et al.* in litt. 2009, pp. 5-6). Of the three known sites of the Peruvian plantcutter in the Ancash Region, only one is reported to be extant (Flanagan *et al.* in litt. 2009, pp. 6, 14). Additionally, the authors reported that the two historical sites in the Lima Region were also unoccupied in the most recent survey (Flanagan *et al.* in litt. 2009, pp. 7, 15).

In summary, the extant population of the Peruvian plantcutter is comprised of two disjunct subpopulations (BLI 2009g, pp. 1-2; Walther 2004, p. 73), with several smaller sites (Flanagan and More 2003, pp. 5-9; Flanagan *et al.* in litt. 2009, pp. 2-7; Walther 2004, p. 73; Williams 2005, p. 1). Additional surveys are needed to determine if available *P. pallida* dry forest habitat is occupied by the Peruvian plantcutter (Flanagan *et al.* in litt. 2009, p. 7).

Population Estimates

There have been no rigorous quantitative assessments of the Peruvian plantcutter's population size (Williams 2005, p. 1). The estimated extant population size is between 500 and 1,000 individuals, and is comprised of 2 disjunct subpopulations (BLI 2009g, pp. 1-2; Walther 2004, p. 73) and several smaller sites (Flanagan and More 2003, pp. 5-9; Flanagan *et al.* in litt. 2009, pp. 2-7; Walther 2004, p. 73; Williams 2005, p. 1).

The northern subpopulation, located in the Talara Province in Piura Region, reportedly has between 400 and 600 individuals, or approximately 60 to 80 percent of the total population of the Peruvian plantcutter (BLI 2009a, p. 2; Snow 2004, p. 69; Walther 2004, p. 73; Williams 2005, p. 1). The second subpopulation, located at Pómac Forest Historical Sanctuary (Lambayeque Region), reportedly has 20 to 60 individuals (BLI 2009a, p. 2; BLI 2009e 2009, p. 1; Walther 2004, p. 73). The smaller sites are estimated to consist of a few individuals, up to 40 individuals (Flanagan and More 2003, pp. 5-9; Flanagan *et al.* in litt. 2009, pp. 2-7; Walther 2004, p. 73; Williams 2005, p. 1).

The population estimate for the Peruvian plantcutter—that is, the total number of mature individuals—is not the same as the effective population size (i.e., the number of individuals that actually contribute to the next generation). Further, the subpopulation structure and the extent of interbreeding among the occurrences of the Peruvian plantcutter are unknown. Although the two large subpopulations and many of the smaller occurrences of the Peruvian plantcutter are widely separated (BLI

2009a, pp. 2-3; Flanagan *et al.* in litt. 2009, pp. 1-9; Ridgely and Tudor 1994, p. 18), there is insufficient information to determine whether these occurrences function as genetically isolated subpopulations.

The Peruvian plantcutter has experienced a population decline of between 1 and 9 percent in the past 10 years, and this rate of decline is predicted to continue (BLI 2009g, p. 1). The population is considered to be declining in close association with continued habitat loss and degradation of habitat (see Factors A and E) (BLI 2009a, pp. 1-3; BLI 2009g, pp. 1-3; Ridgely and Tudor 1994, p. 18; Snow 2004, p. 69).

Conservation Status

The Peruvian plantcutter is considered “endangered” by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276855). The IUCN considers the Peruvian plantcutter to be “Endangered” because of ongoing habitat destruction and degradation of its small and severely fragmented range (BLI 2000, p. 402; BLI 2009a, pp. 2-3; BLI 2009g 2009, pp. 1-2). From 1996 to 2000, the IUCN considered the Peruvian plantcutter to be “Critically Endangered” (BLI 2009g, p. 1), following changes to the IUCN listing criteria in 2001. Experts have suggested returning the the species to its previous classification of “Critically Endangered,” due to the numerous and immediate threats to the species (Jeremy N. M. Flanagan, Conservation Biologist, Sullana, Peru, in litt. 2009 e-mail to DSA, p. 1; Snow 2004, p. 69; Walther 2004, p. 74).

The Peruvian plantcutter occurs within two protected areas in Peru. It has been documented in the *P. pallida* dry forest within the protected archeological sites of the Pómac Forest Historical Sanctuary (BLI 2009e, p. 1) and Murales Forest (Walther 2004, p. 73).

Summary of Factors Affecting the Peruvian plantcutter

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

The Peruvian plantcutter is dependent upon undisturbed *Prosopis pallida* dry forest with good floristic diversity (Collar *et al.* 1992, p. 805; Engblom 1998, p. 1; Flanagan and More 2003, p. 4). In northwest Peru, *P. pallida* dry forest was historically contiguous, covering approximately 2,703 m² (7,000 km²) of the coastal lowland of northwest

Peru (Ferreyera 1983, p. 248). There were also extensive wooded stands of small to medium trees of *P. pallida*, *Acacia* spp., *Capparis* spp., and willows (*Salix* spp.) along permanent lowland rivers, which have since been cleared for agricultural purposes (Lanyon 1975, p. 443).

Today, with the exception of three relatively large intact dry forests (i.e., Talara Province, Murales Forest, and Pómac Forest Historical Sanctuary), the vast majority of *P. pallida* dry forest, arid lowland scrub, and riparian vegetation has been reduced due to human activities. Seasonally dry tropical forests are considered the most threatened of all major tropical forest types (Janzen 1988, p. 130), with higher threat levels than any other Neotropical habitat (Stotz *et al.* 1996, p. 51). The Peruvian plantcutter has been extirpated from most of its historical sites due to loss or degradation of habitat (Elton 2004, p. 1; Flanagan *et al.* in litt. 2009, pp. 1-15; Flanagan and More 2003, pp. 5-9; Snow 2004, p. 69). Current information indicates that the vast majority of occupied sites of the Peruvian plantcutter are small, remnant, disjunct patches of *P. pallida* dry forest with each a few acres in size (BLI 2000, p. 402; Flanagan *et al.* in litt. 2009, pp. 2-7; Snow 2004, p. 69; Walther 2004, p. 73).

Habitat loss, conversion, and degradation throughout the Peruvian plantcutter's range have been and continue to occur as a result of human activities (BLI 2000, p. 402; BLI 2009a, p. 2; BLI 2009b, p. 1; BLI 2009e, p. 1; BLI 2009f, p. 1; Bridgewater *et al.* 2003, p. 132; Flanagan *et al.* in litt. 2009, p. 15; Lanyon 1975, p. 443; Schulenberg *et al.* 2007, p. 488; Snow 2004, p. 69; Walther 2004, p. 73), including:

(1) Clearcutting and burning of dry forest for agriculture and other purposes (BLI 2000, p. 402; BLI 2009a, p. 2; BLI 2009d, pp. 1-2; BLI 2009f, p. 1; BLI 2009g, p. 2; Bridgewater *et al.* 2003, p. 132; Collar *et al.* 1992, p. 806; Engblom 1998, p. 1; Flanagan *et al.* 2005, p. 244; Ridgely and Tudor 1994, p. 734; Snow 2004, p. 69; Walther 2004, p. 73; Williams 2005, p. 2);

(2) Extraction activities, including cutting for timber, firewood, and charcoal production (Best and Kessler 1995, p. 196; BLI 2000, p. 402; BLI 2009d, pp. 1-2; BLI 2009g, p. 2; Ridgely and Tudor 1994, p. 734; Rodriguez *et al.* 2007, p. 269; Snow 2004, p. 69; Williams 2005, p. 1);

(3) Grazing by goats of *P. pallida* dry forests, and arid scrub and riparian vegetation (*Capra* species) (BLI 2000, p. 402; BLI 2009a, p. 2; BLI 2009b, p. 1; BLI 2009d, pp. 1-2; Best and Kessler

1995, p. 196; More 2002, p. 37; Snow 2004, p. 69);

(4) Human encroachment (Fernandez-Baca *et al.* 2007, p. 45); and

(5) Unpredictable climate fluctuations that exacerbate human activities and encourage further habitat destruction (Block and Richter 2007, p. 1; Jetz *et al.* 2007, p. 1211; Richter 2005, p. 26).

The vast majority of *P. pallida* dry forest habitat has been converted to commercial agricultural production, which is the primary factor in the historical decline of the Peruvian plantcutter (BLI 2000, p. 402; BLI 2009a, p. 2; BLI 2009d, pp. 1-2; BLI 2009f, p. 1; Collar *et al.* 1992, p. 806; Engblom 1998, p. 1; Ridgely and Tudor 1994, p. 734; Snow 2004, p. 69; Walther 2004, p. 73; Williams 2005, p. 2). Agriculture in the coastal lowlands of northwest Peru consists of modern large, privately owned farms and large cooperatives that primarily produce crops (e.g., sugarcane, cotton, rice) for export (Lanyon 1975, p. 443; Roethke 2003, pp. 58-59).

Continual habitat destruction and degradation of the dry forest is mostly due to firewood cutting and charcoal production. *Prosopis pallida* is the dominant tree of the dry forest habitat, and is highly sought after because the wood provides an important source of high-quality cooking fuel (Brewbaker 1987, p. 1; Pasiecznik *et al.* 2001, p. 75). Throughout the Peruvian plantcutter's range, whole trees, branches, and roots of *P. pallida* are cut for firewood and production of charcoal, which is used for cooking fuel in homes, restaurants, and businesses that use brick kilns, both locally and in urban centers (Flanagan *et al.* in litt. 2009, p. 7). Wood of *P. pallida* is also used for construction and fence posts (Pasiecznik *et al.* 2001, p. 78). Additionally, roots of older *P. pallida* trees are used in wooden art crafts (BLI 2009a, p. 2).

Talara Province (in Piura Region) contains the largest remaining intact *P. pallida* dry forest in northwest Peru, encompassing approximately 123,553 ac (50,000 ha) (Flanagan *et al.* in litt. 2009, pp. 2-3; Flanagan and More 2003, p. 5; Walther 2004, p. 73). The Province also has the largest subpopulation of the Peruvian plantcutter, reportedly between 400 and 600 individuals or approximately 60 to 80 percent of the total population (BLI 2009a, p. 2; Elton 2004, pp. 3-4; Snow 2004, p. 69; Walther 2004, p. 73; Williams 2005, p. 1). Until recently, a large portion of the Province, including *P. pallida* dry forest habitat, was owned by the State-owned petroleum company PetroPeru, which prohibited access to approximately 90,000 ac (36,422 ha). Under the

management of PetroPeru, the *P. pallida* dry forest was not subject to the same habitat destruction and degradation activities (e.g., clearing of trees, firewood cutting and charcoal production) as other dry forest habitat areas (Elton 2004, pp. 3-4; Hinze 2004, p. 1). Recently, the land was reverted to the Peruvian Government, and it is unclear whether the government plans to issue private concessions as in other areas of the Province (Elton 2004, p. 4). Consequently there have been recent efforts, including a formal petition to the Peruvian Government, to create a 12,000 to 24,710-ac (4,856- to 10,000-ha) protected reserve for the northern subpopulation of the Peruvian plantcutter (Elton 2004, p. 4; Walther 2004, p. 73). However, the government has not designated such a reserve for the species (Elton 2004, p. 4; Williams 2005, p. 3).

Habitat destruction and degradation of *P. pallida* dry forest, including firewood cutting and charcoal production, is ongoing in the Talara Province, including on the land previously owned by PetroPeru and an area identified as the Talara Important Birding Area by BirdLife International (Flanagan in litt. 2009, p. 1). Since 2005, there has been extensive cutting and clearing of *P. pallida* trees for fuel to cook and dry Humboldt giant squid (*Dosidicus gigas*) carcasses (Flanagan *et al.* in litt. 2009, p. 8). The most important commercial fishery of the Humboldt giant squid occurs along the coast of Peru (UNEP 2006, p. 33; Zeidberg and Robison, 2007, p. 12, 948). Harvested carcasses are transported by truck from the Talara port to recently cleared areas in the dry forest, where they are boiled and dried (Flanagan *et al.* in litt. 2009, p. 8). Therefore, this fishery not only adds to the collection pressure on *Prosopis* species for use as fuel, but also adds to forest clearing in the area. Another relatively new demand for *P. pallida* firewood is associated with the illegal extraction of crude oil from above-ground pipes in the Talara Province. The stolen oil is distilled by heating it with firewood (Flanagan *et al.* in litt. 2009, p. 8). *Capparis scabrada* (no common name, but locally known as "sapote") is a tree that occurs with *P. pallida* and is also a food source for the Peruvian plantcutter. Although the tree is listed as "Critically Endangered" by the Peruvian Government, the highly sought after wood is cut to produce handicrafts for the local, national, and international market, as well as for firewood and charcoal production (Rodriguez *et al.* 2007, p. 269).

Habitat alteration is also caused by grazing goats, which remove or heavily degrade the shrubs and trees (BLI 2000, p. 402; BLI 2009a, p. 2; Elton 2004, pp. 3-4; Snow 2004, p. 69; Williams 2005, p. 2). The seed pods and leaves of *P. pallida* provide highly nutritious fodder for goats (Brewbaker 1987, pp. 1-2; Pasiecznik *et al.* 2001, p. 95). Goats roam freely and graze on trees and shrubs, particularly lower branches close to ground which are preferred by the Peruvian plantcutter for foraging and nesting (Elton 2004, pp. 3-4; Snow 2004, p. 50; Williams 2005, p. 2).

Human encroachment and concomitant increasing human population pressures exacerbate the destructive effects of ongoing human activities (e.g., clearing of *P. pallida* dry forest, firewood cutting, and charcoal production) throughout the Peruvian plantcutter's range. Although the coastal lowlands represent only about 10 percent of country's total territory, more urban centers are located on the coast, which represent approximately 52 percent of the total population of Peru (Fernandez-Baca *et al.* 2007, p. 45). Larger concentrations of people put greater demand on the natural resources in the area, which spurs additional habitat destruction and increases infrastructure development that further facilitates encroachment.

Peruvian plantcutters are also impacted by unpredictable climate fluctuations that exacerbate the effects of habitat fragmentation. Unpredictable climate fluctuations are more fully described under the Factor A analysis of the ash-breasted tit-tyrant and are summarized here. Changes in weather patterns, such as ENSO cycles (El Niño and La Niña events), tend to increase precipitation in normally dry areas, and decrease precipitation in normally wet areas (Holmgren *et al.* 2001, p. 89; TAO Project n.d., p. 1) while intensifying the effects of habitat fragmentation on the decline of a species (England 2000, p. 86; Holmgren *et al.* 2001, p. 89; Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027; Parmesan and Mathews 2005, p. 334; Plumart 2007, pp. 1-2; Timmermann 1999, p. 694), especially for narrow endemics (Jetz *et al.* 2007, p. 1213) such as the Peruvian plantcutter.

The arid terrestrial ecosystem of Northwest Peru, where the Peruvian plantcutter occurs, is strongly influenced by the ENSO cycle (Rodriguez *et al.* 2005, p. 1), which can have profound and long-lasting effects (Holmgren *et al.* 2006a, p. 87; Mooers *et al.* 2007, p. 2). The amount of rainfall during an El Niño year can be more than 25 times greater than during normal years in northern Peru (Holmgren *et al.*

2006a, p. 90; Rodriguez *et al* 2005, p. 2). El Niño events are important triggers for regeneration of plants in semiarid ecosystems, particularly the dry forest of northwest Peru (Holmgren *et al.* 2006a, p. 88; Lopez *et al.* 2006, p. 903; Rodriguez *et al.* 2005, pp. 2-3). During El Niño events, plant communities and barren lands are transformed into lush vegetation, as seeds germinate and grow more quickly in response to increased rainfall (Holmgren *et al.* 2006a, p. 88; Holmgren *et al.* 2006b, pp. 2-8; Rodriguez *et al.* 2005, pp. 1-6). Over the last 20 years, recruitment of *P. pallida* in northwest Peru doubled during El Niño years, when compared to non-El Niño years (Holmgren *et al.* 2006b, p. 7). However, the abundant supply of vegetation encourages locals to expand goat breeding operations, which results in overgrazing by goats and further land degradation (Richter 2005, p. 26).

ENSO cycles increase the risk of fire because El Niño events are often followed by years of extremely dry weather (Block and Richter 2007, p. 1), and accumulated biomass dries and adds to the fuel load in the dry season (Block and Richter 2007, p. 1; Power *et al.* 2007, p. 898). Evidence suggests that the fire cycle in Peru has shortened, particularly coastal Peru and west of the Andes (Power *et al.* 2007, pp. 897-898), which can have broad ecological consequences (Block and Richter 2007, p. 1; Power *et al.* 2007, p. 898). According to Block and Richter (2007, p. 1), *P. pallida* dry forest and *Capparis* spp. scrublands in northwest Peru would likely experience a long-term change in plant species composition that favor aggressive, annual, non-native weedy plant species (Richter 2005, p. 26). An accelerated fire cycle would further exacerbate changes in species composition that hinder long-lived perennial, native plant species, such as *Prosopis* species, upon which the Peruvian plantcutter relies.

ENSO cycles are ongoing, having occurred several times within the last decade (NWS 2009, p. 2), and evidence suggests that ENSO cycles have already increased in periodicity and severity (Richter 2005, pp. 24-25; Timmermann 1999, p. 694), which will exacerbate the negative impacts of habitat destruction on a species. It is predicted that, by 2050, approximately 11 to 16 percent of existing land is likely to be unsuitable for this species due to climate change; and, by 2100, it is predicted that about 24 to 35 percent of the species' range is likely to be lost as a direct result of global climate change (Jetz *et al.* 2007, p. 81).

Habitat destruction is often caused by a combination of human activities that

promote habitat degradation. In Lambayeque Region, a 3,706-ac (1,500-ha) section of remnant *P. pallida* dry forest is under continual threat from human activities, including conversion to agriculture, firewood cutting and charcoal production, and grazing by goats. This area may support between 20 and 40 Peruvian plantcutters (BLI 2009f, p. 1; Walther 2004, p. 73). In the 1990s, a significant portion of this dry forest was converted to sugarcane fields (Engblom in litt. 1998, p. 1; Snow 2004, p. 69; Walther 2004, p. 73; Williams 2005, p. 2). Within Piura and Lambayeque Regions, threats to the dry forest habitat include conversion to agriculture, firewood and timber cutting, and grazing by goats (BLI 2009d, pp. 1-2). Habitat destruction and alteration also occurs within two protected areas where the Peruvian plantcutter occurs (in Lambayeque Region), Pómac Forest Historical Sanctuary (Andean Air Mail and Peruvian Times 2009, p. 1; Flanagan *et al.* in litt. 2009, pp. 7-8; Williams 2005, p. 1), and the Murales Forest (BLI 2000, p. 402; BLI 2009a, p. 3; Stattersfield *et al.* 2000, p. 402; Walther 2004, p. 73). Habitat destruction and alteration activities within these protected areas are discussed under Factor D.

Experts consider the population of this range-restricted endemic species to be declining in close association with the continued habitat loss and degradation (BLI 2000, p. 401; BLI 2009a, pp. 1-2; BLI 2009g, pp. 1-3), and that the effects are higher in dry forest habitat than in any other Neotropical habitat (Stotz *et al.* 1998, p. 51).

Summary of Factor A

The Peruvian plantcutter is dependent upon intact *P. pallida* dry forest with low-hanging branches and high floristic diversity, and associated arid lowland scrub and riparian vegetation. *Prosopis pallida* dry forest habitat, as well as arid lowland scrub and riparian shrub habitats, throughout Peruvian plantcutter's range have been and continue to be altered and destroyed as a result of human activities, including conversion to agriculture; timber and firewood cutting and charcoal production; grazing of goats; and human encroachment. Extant *P. pallida* dry forest today consists of remnant, disjunct patches of woodlands, which are heavily disturbed and under continued threat of degradation by human activities. Although observations suggest that this dry forest-dependent species is able to occupy very small remnant patches of dry forest with low-hanging branches and floristic diversity, and is able to persist to some degree

near developed lands, many of these sites are approaching the lower threshold of the species' ecological requirements. This species has been extirpated from most of its historical sites due to loss or degradation of habitat. Additionally, many of the extant occupied sites are separated by great distances, which may lead to genetic isolation of the species (See Factor E). The same activities that caused the historical decline in this species are ongoing today. These habitat-altering activities are compounded by unexpected climate fluctuations, especially for narrow endemics, such as the Peruvian plantcutter. Excessive rains that are accompanied by El Niño events induce further habitat destruction, as people take advantage of better grazing and growing climate conditions. Climate models predict that this species' habitat will continue to decline. Destruction of the remaining *P. pallida* dry forest fragments in Peru continues to reduce the quantity, quality, distribution, and regeneration of remaining patches of dry forest. Human activities that degrade, alter, and destroy habitat are ongoing throughout the species' range, including within the one protected area (Factor D). Therefore, we find that destruction and modification of habitat are threats to the continued existence of Peruvian plantcutter throughout its range.

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

We are not aware of any information currently available that indicates that overutilization of Peruvian plantcutter for commercial, recreation, scientific, or education purposes has occurred or is occurring at this time. As a result, we are not considering overutilization to be a threat to the continued existence of the Peruvian plantcutter.

C. Disease or Predation

We are not aware of any scientific or commercial information that indicates disease or predation poses a threat to the Peruvian plantcutter. As a result, we do not consider disease or predation to be a threat to the continued existence of the Peruvian plantcutter.

D. The Inadequacy of Existing Regulatory Mechanisms

The Peruvian plantcutter is considered "endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276854). This Decree prohibits hunting, take, transport, and trade of protected species, except as permitted by regulation. As hunting, taking, or trade

do not currently threaten the Peruvian plantcutter (Factor B), this regulation does not mitigate any current threats to the species.

Peru has several categories of national habitat protection, which were described above as part of Factor D for the ash-breasted tit-tyrant (BLI 2008, p. 1; IUCN 1994, p. 2; Rodríguez and Young 2000, p. 330). The Peruvian plantcutter is known to occur within two Peruvian nationally protected areas, the Pómac Forest Historical Sanctuary and the Murales Forest (both Lambayeque Region). The Pómac Forest Historical Sanctuary supports an estimated 20 to 60 Peruvian plantcutters (BLI 2009a, p. 2; BLI 2009e, p. 1; Walther 2004, p. 73). Resources within the Pómac Forest Historical Sanctuary are managed for the preservation of the archeological site, *P. pallida* dry forest, and wildlife species. However, habitat destruction and alteration, including illegal forest clearing for farming, timber and firewood cutting, and grazing, continually threaten the Sanctuary (Williams 2005, p. 1). For 8 years, more than 250 families illegally occupied and farmed land in the Sanctuary. During the illegal occupancy, the inhabitants logged 4,942 ac (2,000 ha) of *P. pallida* trees for firewood and burned many other trees for charcoal production (Andean Air Mail and Peruvian Times 2009, p. 1). The logged forest was subsequently converted to agricultural crops, while remaining forest habitat was continually degraded by firewood cutting, charcoal production, and grazing of goats (Flanagan *et al.* in litt. 2009, p. 8.). In January 2009, the government forcibly removed the inhabitants, but it is too soon to determine the effect that habitat destruction has had on the suitability of the habitat for the Peruvian plantcutter. There is insufficient information to conclude that recent efforts to stop the illegal human occupancy of the area will have a positive impact on the species or remaining habitat within the protected area. Therefore, any protections afforded by this Sanctuary has not mitigated the threats to the species from ongoing habitat loss and associated population decline (Factor A).

The Murales Forest is a designated Archeological Reserved Zone (BLI 2000, p. 401; BLI 2009a, p. 3; Stattersfield *et al.* 2000, p. 402), and supports a declining population of Peruvian plantcutters. According to Peruvian law, designation as a Reserved Zone allows for temporary protection while further study is under way to determine the area's importance (BLI 2008, p. 1; Rodríguez and Young 2000, p. 330).

Although strict monitoring has protected some habitat (BLI 2009a, p. 3), the actual dry forest is not protected. In 1999, land rights to sections of the forest were sold for agricultural conversion, and government intervention has been necessary to prevent further sales of land for conversion to agriculture (BLI 2009a, p. 3). In 1999, Murales Forest and adjacent areas contained approximately 1,221 ac (494 ha) of habitat, and reportedly supported 140 Peruvian plantcutters (BLI 2000, p. 402). In 2004, the population was estimated to be 20 to 40 individuals (Walther 2004, p. 73). Therefore, the presence of the Peruvian plantcutter within this protected area has not mitigated the threats to the species from ongoing habitat loss and associated population decline (Factor A).

Incidences of illegal activity that occur throughout the species' range also impact the Peruvian plantcutter. Ongoing firewood cutting and charcoal production degrades the small amount of remaining dry forest habitat throughout the species' range (BLI 2009d, pp. 1-2; BLI 2009g, p. 2; Ridgely and Tudor 1994, p. 734; Rodríguez *et al.* 2007, p. 269; Snow 2004, p. 69; Williams 2005, p. 1). In Talara Province (Piura Region), a recent increase in the illegal extraction of crude oil has generated further demand for *P. pallida* firewood, which is used as fuel to heat-distill the oil. According to Flanagan *et al.* (in litt. 2009, p. 8), enforcement to combat this illegal activity is difficult. Therefore, existing laws are ineffective at mitigating the ongoing threat of habitat destruction (Factor A).

Summary of Factor D

Peru has enacted various laws and regulatory mechanisms to protect and manage wildlife and their habitats. The Peruvian plantcutter is "endangered" under Peruvian law and occurs within two protected areas in Peru. As discussed under Factor A, the Peruvian plantcutter prefers *P. pallida* dry forest. This habitat has been drastically reduced and remaining habitat is comprised of small remnant patches of dry forest that are separated by great distances. Habitat throughout the species' range has been and continues to be destroyed and altered as a result of human activities, primarily conversion to agriculture, and continual degradation by timber and firewood harvest and charcoal production, and grazing by goats. These activities are ongoing, including within protected areas and despite the species' "endangered" status, indicating that the laws governing wildlife and habitat protection in Peru are either inadequate

or inadequately enforced to protect the species or to mitigate ongoing habitat loss (Factor A) and population declines (Factor E). Therefore, we find that the existing regulatory mechanisms are inadequate to mitigate the current threats to the continued existence of the Peruvian plantcutter throughout its range.

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

An additional factor that affects the continued existence of the Peruvian plantcutter is the species' small population size. BirdLife International has placed the Peruvian plantcutter in the population category of between 500 and 1,000 individuals (BLI 2009g, p. 1). The species' population size is not characterized as "small" in published literature and there is insufficient information on similar species (i.e., the other South American plantcutters) to understand whether the Peruvian plantcutter's population size is small relative to other plantcutters. However, there are several indications that this number of individuals represents a small population.

First, the Peruvian plantcutter's population size—which is defined by BirdLife International as the total number of mature individuals—is not the same as the effective population size—the number of individuals that actually contribute to the next generation (Shaffer 1981, pp. 132-133; Soulé 1980, pp. 160-162). Not all individuals in a population will contribute to reproduction each year. Therefore, the estimated population size for the Peruvian plantcutter is an overestimate of the species' effective population size. Moreover, the subpopulation structure and the extent of interbreeding among the occurrences of the Peruvian plantcutter are unknown (see Population Estimates). If further research indicates that species does not breed as a single population, its effective population size would be further reduced.

Second, the extant Peruvian plantcutter population occurs primarily in 2 disjunct subpopulations—Talara and Pómac Forest Historical Sanctuary (BLI 2009g, pp. 1-2; Walther 2004, p. 73)—and in several smaller sites (Flanagan and More 2003, pp. 5-9; Flanagan *et al.* in litt. 2009, pp. 2-7; Walther 2004, p. 73; Williams 2005, p. 1). Talara and Pómac Forest Historical Sanctuary are approximately 160 mi (257 km) apart (FCC (Federal Communications Commission)-Audio Division 2009). The Peruvian plantcutter is dependent upon

undisturbed *Prosopis pallida* dry forest with good floristic diversity (Collar *et al.* 1992, p. 805; Engblom 1998, p. 1; Flanagan and More 2003, p. 4). Its habitat is heavily degraded and localities are small, severely fragmented, and widely separated (BLI 2009a, pp. 2-3; Bridgewater *et al.* 2003, p. 132; Flanagan *et al.* in litt. 2009, pp. 1-9; Ridgely and Tudor 1994, p. 18) (see Factor A). It is possible that the distance between patches of suitable habitat is too great to support interbreeding between localities, so that the extant occurrences of this species would function as genetically isolated subpopulations.

For these reasons, we consider the Peruvian plantcutter's current estimated population to be small and, as such, this species is subject to the risks associated with small population sizes. Small population size renders a species vulnerable to any of several risks, including inbreeding depression, loss of genetic variation, and accumulation of new mutations. Inbreeding can have individual or population-level consequences either by increasing the phenotypic expression (the outward appearance or observable structure, function, or behavior of a living organism) of recessive, deleterious alleles or by reducing the overall fitness of individuals in the population (Charlesworth and Charlesworth 1987, p. 231; Shaffer 1981, p. 131). Small, isolated wildlife populations are also more susceptible to environmental fluctuations and demographic shifts (Pimm *et al.* 1988, pp. 757, 773-775; Shaffer 1981, p. 131), such as reduced reproductive success of individuals and chance disequilibrium of sex ratios. Species tend to have a higher risk of extinction if they occupy a small geographic range and occur at low density (Purvis *et al.* 2000, p. 1949).

The Peruvian plantcutter has experienced a population decline of between 1 and 9 percent in the past 10 years due to habitat loss and this decline is expected to continue in close association with continued habitat loss and degradation (see Factor A) (BLI 2009a, pp. 1-3; BLI 2009g, pp. 1-3; Ridgely and Tudor 1994, p. 18; Snow 2004, p. 69). Extinction risk is heightened in small, declining populations by an increased vulnerability to the loss of genetic variation due to inbreeding depression and genetic drift. This, in turn, compromises a species' ability to adapt genetically to changing environments (Frankham 1996, p. 1507) and reduces fitness, thus increasing extinction risk (Reed and Frankham 2003, pp. 233-234). Once a population is reduced below a

certain number of individuals, it tends to rapidly decline towards extinction (Franklin 1980, pp. 147-148; Gilpin and Soulé 1986, p. 25; Holsinger 2000, pp. 64-65; Soulé 1987, p. 181).

Complications arising from the species' small population size are exacerbated by its fragmented distribution. Because remaining habitat patches are small, heavily degraded, and widely separated, the Peruvian plantcutter's current distribution is highly restricted and severely fragmented (BLI 2009a, pp. 2-3; Bridgewater *et al.* 2003, p. 132; Flanagan *et al.* in litt. 2009, pp. 1-9; Ridgely and Tudor 1994, p. 18). Habitat fragmentation can cause genetic isolation and heighten the risks to the species associated with short-term genetic viability. A species' small population size, combined with a restricted and fragmented distribution, exacerbates a species' vulnerability to adverse natural events (e.g., genetic, demographic, or environmental) and manmade activities (e.g., land clearing, timber and firewood cutting, and grazing by goats) (Holsinger 2000, pp. 64-65; Primack 1998, pp. 279-308; Young and Clarke 2000, pp. 361-366).

Summary of Factor E

The Peruvian plantcutter has a small population size that renders it vulnerable to genetic risks that negatively impact the species' viability. The species occurs primarily in two disjunct subpopulations, and remaining habitat is highly fragmented and continues to be altered by human activities (Factor A). Its small population size, combined with its restricted and fragmented range, increases the Peruvian plantcutter's vulnerability to extinction, through demographic or environmental fluctuations. Based on its small population size and fragmented distribution, we have determined that the Peruvian plantcutter is particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., deforestation, habitat alteration, and infrastructure development) that destroy individuals and their habitat. The genetic and demographic risks associated with small population sizes are exacerbated by ongoing human activities that continue to curtail the species' habitat throughout its range. The species' population has declined and is predicted to continue declining commensurate with ongoing habitat loss (Factor A) and we consider that the risks associated with small population size will continue to impact this species and may accelerate, if

habitat destruction continues unabated. Therefore, we find that the species' small population size, in concert with its fragmented distribution and its heightened vulnerability to adverse natural events and manmade activities, are threats to the continued existence of the Peruvian plantcutter throughout its range.

Status Determination for the Peruvian Plantcutter

The Peruvian plantcutter, a small, herbivorous bird, is endemic to semiarid lowland dry forests of coastal northwest Peru. The species' primary habitat is *Prosopis pallida* dry forest between 33 and 1,804 ft (10 and 550 m) above sea level. The species is dependent on year-round availability of high-quality food and is known primarily in two disjunct subpopulations, with several smaller, widely separated sites in the Regions of Piura, Lambayeque, Cajamarca, La Libertad, and Ancash (from north to south). The actual range of the Peruvian plantcutter is smaller than the reported range of 1,892 mi² (4,900 km²), given the severely fragmented distribution of the species. The species' population size is estimated to be 500-1,000 individuals.

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the Peruvian plantcutter and have concluded that there are three primary factors that threaten the continued existence of the Peruvian plantcutter: (1) Habitat destruction, fragmentation, and degradation; (2) limited size and isolation of remaining populations; and (3) inadequate regulatory mechanisms.

Human activities that degrade, alter, and destroy habitat are ongoing throughout the Peruvian plantcutter's range. Widespread land conversion to agriculture has removed the vast majority of *P. pallida* dry forest habitat throughout the range of the Peruvian plantcutter. Peruvian plantcutter habitat continues to be altered by human activities, such as timber and firewood cutting, burning, and grazing, which result in the continued degradation, conversion, and destruction of habitat and reduce the quantity, quality, distribution, and regeneration of remaining dry forest habitat. Current research indicates that narrow endemics, such as the Peruvian plantcutter, are especially susceptible to climate fluctuations, because of the synergistic effect these fluctuations have on declining populations that are also experiencing range reductions due to human activities. Remaining habitat is highly fragmented and disjunct, separated by large areas of converted

land (e.g., agriculture, urban areas) and sparse vegetation (e.g., Sechura Desert), and excessive rains induce further habitat destruction and prevent regeneration, as people take advantage of better grazing and growing climate conditions.

The Peruvian plantcutter's population is small, rendering the species particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., deforestation and firewood extraction) that destroy individuals and their habitat. Ongoing human activities that cause habitat loss throughout the species' range exacerbate the genetic and demographic risks associated with small population sizes (Factor E). The population has declined 1–9 percent in the past 10 years (see Population Estimates), in association with continued habitat loss (Factor A). Habitat loss was a factor in this species' historical decline (see Historical Range and Distribution)—the Peruvian plantcutter has been extirpated from 11 of its 14 historical sites (see Current Range and Distribution)—and the species is considered to be declining today in association with the continued reduction in habitat (Factors A and E). Moreover, current research indicates that narrow endemics, such as the Peruvian plantcutter, are especially susceptible to climate fluctuations because of the synergistic effect these fluctuations have on declining populations that are also experiencing range reductions due to human activities (Factor A).

Despite the species' endangered status in Peru and its occurrence within two protected areas (Factor D), habitat loss and degradation continue throughout the Peruvian plantcutter's habitat (Factor A). Therefore, regulatory mechanisms are either inadequate or ineffective at mitigating the existing threats to the Peruvian plantcutter and its habitat (Factor D).

Section 3 of the Act defines an "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range" and a "threatened species" as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Based on the immediate and ongoing significant threats to the Peruvian plantcutter throughout its entire range, as described above, we determine that the Peruvian plantcutter is in danger of extinction throughout all of its range. Therefore, on the basis of the best available scientific and commercial information, we are proposing to list the

Peruvian plantcutter as an endangered species throughout all of its range.

V. Royal cinclodes (*Cinclodes aricomae*)

Species Description

The royal cinclodes, also known as "churrete real" and "remolinera real," is a large-billed ovenbird in the Furnariidae family that is native to high-altitude woodlands of the Bolivian and Peruvian Andes (BLI 2009i, pp. 1-2; del Hoyo *et al.* 2003, p. 253; InfoNatura 2007, p. 1; Supreme Decree No. 034-2004-AG 2004, p. 27685; Valqui 2000, p. 104). The adult is nearly 8 in (20 cm) in length, with dark chocolate-brown plumage on the upperparts, with a darker crown and a buff-colored area above the eyes. The throat is buff-colored, and the remaining underparts are gray-brown to buff-white. The wings are dark with prominent edging that forms a distinctive wing-bar in flight. The large, dark bill is slightly curved at the tip (BLI 2009i, p. 1).

Taxonomy

When the species was first taxonomically described, the royal cinclodes was placed in the genus *Upucerthia* (Carraker 1932, pp. 1-2) and was then transferred to *Geositta* as a subspecies (*Geositta excelsior aricomae*) (Vaurie 1980, p. 14). Later, it was transferred to the genus *Cinclodes*, where it was considered a race or subspecies of the stout-billed cinclodes (*Cinclodes excelsior*) until recently (BLI 2009i, p. 1; Fjelds  and Krabbe 1990, pp. 337-338; Vaurie 1980, p. 15). The royal cinclodes is now considered a distinct species (*C. aricomae*) based on differences in its habitat, morphology, and genetic distance (Chesser 2004, p. 763; del Hoyo *et al.* 2003, p. 253). Therefore, we accept the species as *Cinclodes aricomae*, which also follows ITIS (2009, p. 1).

Habitat and Life History

The royal cinclodes is restricted to moist and mossy habitat amidst the steep, rocky slopes of semihumid *Polylepis* or *Polylepis - Gynoxys* woodlands, where the species is found at elevations between 11,483 and 12,092 ft (3,500 and 4,600 m) (BLI 2000, p. 345; BLI 2009i, p. 2; Collar *et al.* 1992, p. 588; del Hoyo *et al.* 2003, p. 253). The characteristics of *Polylepis* habitat were described above as part of the Habitat and Life History of the ash-breasted tityrant (Aucca and Ramsay 2005, p. 1; Chutas *et al.* 2008, p. 3; De la Via 2004, p. 10; IPNI 2009, p. 1; Kessler 1998, p. 1; Kessler and Schmidt-Lebuhn 2006, pp. 1-2; Purcell *et al.* 2004, p. 455). The

royal cinclodes prefers dense woodlands (BLI 2000, p. 345; BLI 2009i, p. 2; Collar *et al.* 1992, p. 588; del Hoyo *et al.* 2003, p. 253), with more closed canopies that provide habitat for more lush moss growth (Engblom *et al.* 2002, p. 57). The moss-laden vegetation and shaded understory harbor a rich diversity of insects, making good feeding grounds for insectivorous birds (De la Via 2004, p. 10) such as the royal cinclodes (del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57). In Bolivia, the royal cinclodes has been observed only in *P. pepeii* forests, but it is found amongst a greater variety of *Polylepis* species in Peru (Chutas *et al.* 2008, p. 16; I. G mez, National Museum of National History-Ornithology Section-Bolivian Fauna Collection, La Paz, Bolivia, e-mail to Division of Scientific Authority, in litt. December 4, 2007, p. 1).

Information on the ecology and breeding behavior of royal cinclodes is limited. The species' feeding territory ranges from 7 to 10 ac (3 to 4 ha) (del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57). Breeding pairs may occupy smaller, 2.5-ac (2-ha) territories (Chutas 2007, p. 7). The royal cinclodes is described as "nervous" and is easily disturbed by humans (Engblom *et al.* 2002, p. 57). The breeding season probably begins in December, but territorialism among pairs can be seen in austral winter (June-August) (BLI 2000, p. 345; del Hoyo *et al.* 2003, p. 253). *Cinclodes* species construct burrows or use natural cavities, crevices, or rodent burrows for nesting (Fjelds  and Krabbe 1990, p. 337; Vaurie 1980, pp. 30, 34). The royal cinclodes' clutch size may be similar to that of the closely-related stout-billed cinclodes (*C. excelsior*), which, according to Graves and Arango (1988, p. 252), is two eggs per clutch.

Its diet consists primarily of invertebrates, sometimes seeds, and occasionally small vertebrates (small frogs) (del Hoyo *et al.* 2003, p. 253). The royal cinclodes forages, solitary or in pairs, by probing through moss and debris on the forest floor (BLI 2000, p. 345; Collar *et al.* 1992, p. 589; del Hoyo *et al.* 2003, p. 253; Fjelds  2002b, p. 9). This ground-feeding strategy generally requires a relatively large territory and may also facilitate interbreeding amongst groups located on adjoining mountain peaks when the species likely descends the mountains during periods of snow cover (Engblom *et al.* 2002, p. 57).

Historical Range and Distribution

The royal cinclodes may once have been locally common and distributed

across most of central to southern Peru and into the Bolivian highlands, in once-contiguous expanses of *Polylepis* forests above 9,843 ft (3,000 m) (BLI 2000, p. 345; BLI 2009i, p. 1; Fjeldså 2002a, pp. 111-112, 115; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97-101). *Polylepis* woodlands are now restricted to elevations of 11,483 to 16,404 ft (3,500 to 5,000 m) (Fjeldså 1992, p. 10). As discussed above for the Historical Range and Distribution of the ash-breasted tit-tyrant, researchers consider human activity to be the primary cause for historical habitat decline and resultant decrease in species richness (Fjeldså and Kessler 1996, Kessler 1995a, b, and Lægaard 1992, as cited in Fjeldså 2002a, p. 112; Fjeldså 2002a, p. 116; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97-101; Kessler and Herzog 1998, pp. 50-51). The royal cinclodes may have been extirpated from its type locality (Aricoma Pass, Puno), and possibly throughout the entire Puno Region, where *Polylepis* forest no longer occurs exists (Collar *et al.* 1992, p. 589; Engblom *et al.* 2002, p. 57) (see Population Estimates). It is estimated that between 2-3 percent and 10 percent of the original forest cover still remains in Peru and Bolivia, respectively (BLI 2009i, p. 1; Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113) (see Factor A). Of this amount, less than 1 percent of the remaining woodlands occur in humid areas, where *Polylepis* denser stands occur (Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113) and which are preferred by the royal cinclodes (del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57). The royal cinclodes was initially discovered in Bolivia in 1876, but was not observed there again until recently (BLI 2009i, p. 2; Hirshfeld 2007, p. 198) (see Current Range and Distribution).

Current Range and Distribution

The royal cinclodes is restricted to moist and mossy habitat amidst the steep rocky slopes of semihumid *Polylepis* or *Polylepis - Gynoxys* woodlands, where the species is found at elevations between 11,483 and 12,092 ft (3,500 and 4,600 m) (BLI 2000, p. 345; BLI 2009i, p. 2; Collar *et al.* 1992, p. 588; del Hoyo *et al.* 2003, p. 253). The current range of the species is approximately 1,042 mi² (2,700 km²) (BLI 2009i, p. 1), which is an overestimate of the actual range (as discussed under the Current Range and Distribution of the ash-breasted tit-tyrant) (BLI 2000, pp. 22, 27), given the fragmented nature of the species' remaining habitat (BLI 2009i, p. 1; Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113).

The royal cinclodes was only rediscovered in Bolivia within the last decade, after more than 100 years of not being observed there (BLI 2009i, p. 2; Hirshfeld 2007, p. 198). Within the last 15 years, royal cinclodes has been observed in Peru's Runtacocha highlands (Apurímac), Pariahuanca Valley (Junín), and Cordillera Vilcanota (Cusco), and in Bolivia's Cordillera Apolobamba and the Cordillera Real (including Ilampu Valley, Sanja Pampa, and Cordillera de La Paz), all in the Bolivian Department of La Paz (BLI 2007, pp. 1-2; BLI 2009i, pp. 1-2; del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57; Hirshfeld 2007, p. 198; InfoNatura 2007, p. 1; Valqui 2000, p. 104).

Population Estimates

Population information is presented first on the range-country level and then in terms of a global population estimate. The range country estimates begin with Peru, where the majority of the population resides.

Local population estimate, Peru: In 1990, the global population was estimated to be 100-150 individuals (Fjeldså and Krabbe 1990, p. 338). This number represents the estimated Peruvian population because the royal cinclodes was only thought to be extant in Peru at the time of this estimate (BLI 2009i, p. 2; Hirshfeld 2007, p. 198). Chutas (2007, p. 8) reported an estimated 189 birds located within four disjunct *Polylepis* forest patches in Peru, with a combined area of 1,554 ac (629 ha). This estimate included 116 birds and 30 birds in Cordilleras Vilcanota and Vilcabamba, respectively (Cusco); 2 birds in Cordillera de Carabaya (Puno); and 41 birds in "Cordillera del Apurímac" (Runtacocha highlands in Apurímac) (Chutas 2007, pp. 4, 8). Subpopulations at four locations in the Cordillera Vilcanota contain as few as 1-4 individuals (BLI 2008, p. 2).

In the Puno Region of Peru, it is unclear whether a viable population of royal cinclodes remains. The royal cinclodes was first observed in Puno in 1930 (Fjeldså and Krabbe 1990, p. 338) and has continued to be reported there in general terms (BLI 2007, pp. 1-2; BLI 2009i, pp. 1-2; Collar *et al.* 1992, p. 588; del Hoyo 2003, p. 253). However, based on habitat availability, InfoNatura (2007, p. 1) predicted that the royal cinclodes does not occur in Puno because the habitat no longer exists there. Indeed, only two royal cinclodes individuals have been reported in the Puno Region (Cordillera de Carabaya) in recent decades (Chutas 2007, pp. 4, 8). There are no other recent observations of the royal cinclodes in Puno (BLI 2009i, p.

2; del Hoyo 2003, p. 283; Engblom *et al.* 2002, p. 57). The species is believed to be extirpated from its type locality (Collar *et al.* 1992, p. 589; Engblom *et al.* 2002, p. 57).

Local population estimate, Bolivia: The species' current range is more widespread in Bolivia than previously understood. The royal cinclodes had not been observed in Bolivia for more than one century, when it was rediscovered there in 1997 (BLI 2009i, p. 2; Hirshfeld 2007, p. 198). Recent surveys in La Paz Department found at least 13 localities (8 in Cordillera Apolobamba and 5 in Cordillera La Paz) (BLI 2009i, p. 1).

Although BirdLife International reports an estimated population size of 50-70 royal cinclodes in Bolivia (Gómez, in litt. 2003, 2008, as cited in BLI 2009i, p. 2), recent surveys indicate that the estimate may be smaller. As discussed above for the local population estimate of the ash-breasted tit-tyrant in Bolivia, Gómez (in litt. 2007, p. 1) conducted intensive studies in Bolivia. From this research, the presence of 1-8 royal cinclodes in each of 30 forest patches was inferred or observed. Thus, they estimated that the royal cinclodes population in Bolivia totals approximately 30 birds. Researchers add that, because the royal cinclodes does not always respond to tape-playbacks, these numbers may underestimate the actual population size (Gómez in litt. 2007, p. 1).

Global population estimate: In 1990, the global population of the royal cinclodes was estimated to be 100-150 individuals (Fjeldså and Krabbe 1990, p. 338). Since at least 2000, BirdLife International has placed this species in the population category of between 50 and 249 individuals (BLI 2000, p. 345). In 2002, Engblom *et al.* (2002, p. 57) estimated a total population size of up to 250 pairs of birds. This amount far exceeds any previous estimates and has not been confirmed by BirdLife International (BLI 2009i, p. 1). In 2003, the global population was once again reported to include only "a few hundred individuals" (del Hoyo *et al.* 2003, p. 253). Based on recent observations in both countries, there are approximately 189 birds in Peru and 50-70 in Bolivia, totaling 239-259 individuals. Recognizing that the royal cinclodes does not always respond to tape-playbacks, this could be an underestimate of the population size (Gómez in litt. 2007, p. 1). While the species continues to be categorized by BirdLife International as having an estimated population between 50-249 individuals (BLI 2007, p. 1; BLI 2009i, p. 1; Fjeldså 2002b, p. 9; Hirshfeld 2007, p. 198), it is possible that the recent

observations in Bolivia will lead to a revision of the species' population estimate (BLI 2009i, p. 1).

It should be noted that the total population size, which includes immature individuals, is not an accurate reflection of the species' effective population size (the number of breeding individuals that contribute to the next generation) (Shaffer 1981, pp. 132-133; Soulé 1980, pp. 160-162). The IUCN estimated that the entire royal cinclodes population contains fewer than 250 mature individuals and no more than 50 mature individuals in any subpopulation (BLI 2008, p. 1; IUCN 2001, pp. 8-12). However, population estimates are incomplete for several of the known localities, and the subpopulation structure and the extent of interbreeding amongst the various localities are unknown. The species' territory ranges from 7 to 10 ac (3 to 4 ha), and its habitat is fragmented, dispersed and sparse (del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57). However, there is no information to indicate the distance that this species is capable of or likely to travel between localities. Engblom *et al.* (2002, p. 57) noted that gene flow between localities likely occurs when the species descends the mountains to forage in the valleys during periods of snow cover at the higher altitudes such that interbreeding may occur at least among localities with shared valleys. This suggests that the species does not breed as a single population. However, there is insufficient information to determine the extent to which this species functions as genetically isolated subpopulations.

The species has experienced a population decline of between 30 and 49 percent in the past 10 years, and this rate of decline is predicted to continue (BLI 2009i, pp. 1, 5). The population is considered to be declining in close association with continued habitat loss and degradation (BLI 2009i, p. 6) (see Factors A and E).

Conservation Status

The royal cinclodes is considered "critically endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276854). The IUCN considers the royal cinclodes to be "Critically Endangered" due to its extremely small population, which consists of "tiny" subpopulations that are severely fragmented and dependent upon a rapidly declining habitat (BLI 2007, p. 1; BLI 2009i, p. 1). The royal cinclodes occurs within the Peruvian protected area of Santuario Histórico Machu Picchu, in Cusco (BLI 2009h, p. 1; BLI 2009i, p. 6; Chutas *et al.* 2008, p.

16). In La Paz Department, Bolivia, the species is found in Parque Nacional y área Natural de Manejo Integrado Madidi, Parque Nacional y área Natural de Manejo Integrado Cotapata, and the co-located protected areas of Reserva Nacional de Fauna de Apolobamba, área Natural de Manejo Integrado de Apolobamba, and Reserva de la Biosfera de Apolobamba (BLI 2009a, p. 1; BLI 2009b, p. 1; BLI 2009c, p. 1; BLI 2009d, p. 1; BLI 2009i, p. 6; Chutas *et al.* 2008, p. 16).

Summary of Factors Affecting the Royal Cinclodes

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

The royal cinclodes is restricted to high-elevation (11,483-12,092 ft (3,500-4,600 m)), moist, moss-laden areas of semihumid *Polylepis* or *Polylepis* - *Gynoxys* woodlands (BLI 2000, p. 345; BLI 2009i, p. 2; Collar *et al.* 1992, p. 588; del Hoyo *et al.* 2003, p. 253). As described more fully for the ash-breasted tit-tyrant (Factor A), *Polylepis* habitat is characterized as a threatened woodland ecosystem on national, regional, and global levels (BLI 2009a, p. 2; Purcell *et al.* 2004, p. 457; Renison *et al.* 2005, as cited in Lloyd 2009, p. 10), with several *Polylepis* species within the royal cinclodes' range considered to be "Vulnerable," according to the IUCN (WCMC 1998a, p. 1; WCMC 1998b, p. 1). *Polylepis* woodlands are dispersed and sparse, with an estimated remaining area of 386 mi² (1,000 km²) in Peru and 1,931 mi² (5,000 km²) in Bolivia (Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113). Within the remaining *Polylepis* woodlands, the royal cinclodes' range is approximately 1,042 mi² (2,700 km²) (BLI 2009i, p. 1). Less than 1 percent of the remaining woodlands occur in humid areas, where denser stands occur (Fjeldså and Kessler 1996, as cited in Fjeldså 2002a, p. 113). The optimal habitat for the royal cinclodes is dense woodlands, with a closed canopy that supports the best foraging habitat (shady, moss-laden vegetation) for this and other insectivorous birds (see Habitat and Life History) (De la Via 2004, p. 10; del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57).

Habitat loss, conversion, and degradation occur throughout the royal cinclodes' range and have been attributed to human activities (a full description of which is provided above as part of the Factor A analysis for the ash-breasted tit-tyrant). These activities include:

(1) Clearcutting and uncontrolled burning for agriculture and pastureland for domesticated animals, all of which contributes to loss of understory moss cover, soil erosion, and degradation, which prevent woodland regeneration (BLI 2009a, p. 2; BLI 2009b, p. 2; BLI 2009c, p. 2; BLI 2009d, p. 2; BLI 2009e, p. 3; BLI 2009f, p. 1; BLI 2009g, p. 1; BLI 2009h, p. 4; BLI 2009i, pp. 2, 6; Engblom *et al.* 2002, p. 56; Fjeldså 2002a, pp. 112, 120; Fjeldså 2002b, p. 8; Jameson and Ramsay 2007, p. 42; Purcell *et al.* 2004, p. 458; WCMC 1998a, p. 1; WCMC 1998b, p. 1);

(2) Extractive activities, such as wood and timber, for local and commercial-scale uses, including firewood and construction (Aucca and Ramsay 2005, p. 287; BLI 2009a, p. 2; BLI 2009b, p. 2; BLI 2009c, p. 2; BLI 2009d, p. 2; BLI 2009e, p. 3; BLI 2009g, p. 1; BLI 2009i, p. 2; Engblom 2000, p. 1; p. 2; Engblom *et al.* 2002, p. 56; Purcell *et al.* 2004, p. 458; WCMC 1998a, p. 1);

(3) Human encroachment, including tourism and industrialization projects, which puts greater demand on natural resources, spurs additional habitat destruction as arable land becomes scarce, and increases infrastructure development that further facilitates encroachment (BLI 2009b, p. 2; BLI 2009d, p. 2; Hensen 2002, p. 199; Purcell and Brelford, 2004, pp. 156-157; Purcell *et al.* 2004, pp. 458-459); and

(4) Unpredictable climate fluctuations that exacerbate the effects of habitat fragmentation (Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027).

These habitat-altering activities are ongoing throughout the royal cinclodes' range, including the Apurimac (BLI 2009g, p. 1) and Cusco Regions (BLI 2009e, p. 1; BLI 2009f, p. 1; BLI 2009h, p. 1) in Peru and also in the Bolivian Department of La Paz (BLI 2009a, p. 1; BLI 2009b, p. 1; BLI 2009c, pp. 1; BLI 2009d, p. 1; Hensen 2002, p. 199; Purcell and Brelford 2004, p. 157; Purcell *et al.* 2004, pp. 458-459). A combination of urbanization, road building, and industrialization projects (such as construction of hydroelectric power stations) in the Bolivian Department of La Paz have resulted in a nearly 40 percent loss of the forest cover between 1991 and 2003 alone; at this rate it is predicted that the remaining *Polylepis* forest in La Paz will be extirpated within the next 30 years (Purcell and Brelford 2004, pp. 156-157).

Community-based *Polylepis* conservation programs have been under way in Peru and Bolivia since 2004 (Gómez in litt. 2003, 2008, as cited in BLI 2009i, p. 2; MacLennan 2009, p. 2),

and have focused on known sites for the royal cinclodes (BLI 2009i, p. 2), including Cordillera Vilcanota and Vilcabamba, and highlands of the Apurímac Region (Aucca and Ramsay 2005, p. 287; ECOAN n.d., p. 1; Lloyd 2009, p. 10). These programs confront the main causes of *Polylepis* woodland loss—burning, grazing, and wood-cutting (Aucca and Ramsay 2005, pp. 187-288; BLI 2009i, p. 2; ECOAN n.d., p. 1; Engblom *et al.* 2002, p. 56; Gómez in litt. 2003, 2008, as cited in BLI 2009i, p. 2; Lloyd 2009, p. 10; MacLennan 2009, p. 2)—and are more fully described above as part of the Factor A analysis for the ash-breasted tit-tyrant (Aucca and Ramsay 2005, p. 287; Engblom *et al.* 2002, p. 56; MacLennan 2009, p. 2). While the *Polylepis* conservation programs foster local, sustainable use of resources (Aucca and Ramsay 2005, p. 287; ECOAN n.d., p. 1; Engblom *et al.* 2002, p. 56), commercial-scale activities, such as clearcutting, logging, tourism, and infrastructure development, that are ongoing throughout this species' range, alter otherwise sustainable resource use practices (Aucca and Ramsay 2005, p. 287; Engblom 2000, p. 2; Engblom *et al.* 2002, p. 56; MacLennan 2009, p. 2; Purcell and Brelford 2004, pp. 156-157; Purcell *et al.* 2004, pp. 458-459; WCMC 1998a, p. 1). Tourism and human encroachment are particularly problematic for the royal cinclodes, which is described as a "nervous" species that is easily disturbed by humans (Engblom *et al.* 2002, p. 57).

In the Cordillera de Vilcanota (Cusco, Peru), where a large portion of the known royal cinclodes population occurs (116 birds were observed there, out of 189 total birds observed in 4 study sites in Peru) (Chutas 2007, pp. 4, 8), *Polylepis* woodland habitat is highly fragmented and degraded. As described more fully for the ash-breasted tit-tyrant (Factor A), recent research indicated:

(1) That four forest patches in the Cordillera de Vilcanota disappeared completely in the last half a century, that the size of remaining *Polylepis* remnants is small (with a mean patch size of 7.4 ac (3 ha)) (Jameson and Ramsay 2007, p. 42), and that many patches are less than 2.5 ac (1 ha) (Lloyd and Marsden in press, as cited in Lloyd 2008, p. 532);

(2) Ten percent of the remaining forest patches showed a decline in forest density (Jameson and Ramsay 2007, p. 42); and

(3) There were no indications of forest regeneration within the study area.

Thus, forest patches in Cordillera Vilcanota are at or below the minimum area required for the royal cinclodes to

obtain food, given that the ground-feeding strategy used by the royal cinclodes generally requires a relatively large territory, from 7 to 10 ac (3 to 4 ha) (del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57). Because the moist, moss-covered woodlands that provide optimal foraging habitat for insectivorous birds (De la Via 2004, p. 10), and which this ground-feeding species prefers (del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57), require a closed canopy, degradation of the royal cinclodes habitat has serious consequences for this species. Reduction of forest density (or, decreased canopy cover) increases desiccation of the moist and mossy ground cover, which, in turn, reduces foraging microhabitats for the species (Engblom *et al.* 2002, p. 57).

Lack of *Polylepis* forest regeneration during nearly 50 years underscores the ramifications of continued burning and clearing to maintain pastures and farmland, which are prevalent activities throughout the royal cinclodes' range (BLI 2009a, p. 2; BLI 2009b, p. 2; BLI 2009c, p. 2; BLI 2009d, p. 2; BLI 2009e, p. 3; BLI 2009f, p. 1; BLI 2009g, p. 1; BLI 2009h, p. 4; BLI 2009i, p. 2; Engblom *et al.* 2002, p. 56; Fjeldsá 2002a, pp. 112, 120; Fjeldsá 2002b, p. 8; Purcell *et al.* 2004, p. 458; WCMC 1998a, p. 1; WCMC 1998b, p. 1). These habitat-altering activities are considered to be key factors preventing regeneration of *Polylepis* woodlands (Fjeldsá 2002a, p. 112, 120; Fjeldsá 2002b, p. 8) and are factors in the historical decline of *Polylepis*-dependent bird species, including the royal cinclodes (BLI 2009i, p. 6; Fjeldsá and Kessler 1996, Kessler 1995a, b, and Lægaard 1992, as cited in Fjeldsá 2002a, p. 112; Fjeldsá 2002a, p. 116; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97-101; Kessler and Herzog 1998, pp. 50-51).

The royal cinclodes may once have been locally common and distributed across most of central to southern Peru and into the Bolivian highlands, in once contiguous expanses of *Polylepis* forests (BLI 2000, p. 345; BLI 2009i, p. 1; Fjeldsá 2002a, pp. 111-112, 115). The royal cinclodes' population size is considered to be declining in close association with continued habitat loss and degradation (BLI 2007, p. 1; BLI 2008, p. 1; BLI 2009i, p. 6). The species may have been extirpated from its type locality (Aricoma Pass, Puno), where *Polylepis* forest no longer occurs, and a search for the species in 1987 resulted in no observations of the royal cinclodes (Collar *et al.* 1992, p. 589; Engblom 2002, p. 57). Based on habitat availability, the royal cinclodes is not predicted to occur in Puno because the

habitat no longer exists there (InfoNatura 2007, p. 1), and only two birds have been observed there in recent years (Chutas 2007, pp. 4, 8). Therefore, further habitat loss will continue to impact the species' already small population size (see Factor E).

Royal cinclodes are also impacted by unpredictable climate fluctuations, which are more fully described under the Factor A analysis of the ash-breasted tit-tyrant and are summarized here. Peru is subject to unpredictable climate fluctuations that exacerbate the effects of habitat fragmentation, such as those that are related to the El Niño Southern Oscillation (ENSO). Changes in weather patterns, such as ENSO cycles (El Niño and La Niña events), tend to increase precipitation in normally dry areas, and decrease precipitation in normally wet areas (Holmgren *et al.* 2001, p. 89; TAO Project n.d., p. 1), exacerbating the effects of habitat fragmentation on the decline of a species (England 2000, p. 86; Holmgren *et al.* 2001, p. 89; Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027; Parmesan and Mathews 2005, p. 334; Plumart 2007, pp. 1-2; Timmermann 1999, p. 694), especially for narrow endemics (Jetz *et al.* 2007, p. 1213) such as the royal cinclodes (see also Factor E). ENSO cycles strongly influence the growth of *Polylepis* species (Christie *et al.* 2008, p. 1) by altering the *Polylepis* species' age structure and mortality, especially where woodlands have undergone disturbance, such as fire and grazing (Villalba and Veblen 1997, pp. 121-123; Villalba and Veblen 1998, pp. 2624, 2637).

ENSO cycles may have already accelerated the fire cycle (Block and Richter 2007, p. 1; Power *et al.* 2007, pp. 897-898), which is a key factor preventing *Polylepis* regeneration (Fjeldsá 2002a, p. 112, 120; Fjeldsá 2002b, p. 8) because *Polylepis* species recover poorly following a fire (Cierjacks *et al.* 2007, p. 176). ENSO cycles are ongoing, having occurred several times within the last decade (NWS 2009, p. 2), and evidence suggests that ENSO cycles have already increased in periodicity and severity (Richter 2005, pp. 24-25; Timmermann 1999, p. 694), which will exacerbate the negative impacts of habitat destruction on a species. It is predicted that, by 2050, approximately 3 to 15 percent of the royal cinclodes' current remaining range is likely to be unsuitable for this species due to climate change and, by 2100, it is predicted that about 8 to 18 percent of the species' range is likely to be lost as a direct result of global climate change (Jetz *et al.* 2007, Supplementary Table 2, p. 89).

Human activities that alter the species' habitat are also ongoing within protected areas, including Santuario Histórico Machu Picchu (in Peru) (BLI 2009h, p. 4), and Parque Nacional y área Natural de Manejo Integrado Madidi, Parque Nacional y área Natural de Manejo Integrado Cotapata, and the co-located protected areas of Reserva Nacional de Fauna de Apolobamba, área Natural de Manejo Integrado de Apolobamba, and Reserva de la Biosfera de Apolobamba (in Bolivia) (BLI 2009a, p. 2; BLI 2009b, p. 2; BLI 2009c, p. 2; BLI 2009d, p. 5). Ongoing habitat destruction and alteration within protected areas, including clearing and human encroachment, is further discussed under Factor D.

Summary of Factor A

Polylepis habitat throughout the royal cinclodes' range has been and continues to be altered and destroyed as a result of human activities, including clearcutting and burning for agriculture, grazing lands, and industrialization; extractive activities, including firewood and timber extraction; and human encroachment and concomitant increased pressure on natural resources. An estimated 1 percent of the once-extensive dense *Polylepis* woodlands remains, and other remaining *Polylepis* woodlands are fragmented and degraded. The royal cinclodes occupies an area of approximately 1,042 mi² (2,700 km²), and is particularly vulnerable to reduction in forest cover, because the moist habitats that serve as their feeding grounds quickly dry out as the forest canopy diminishes. Researchers estimate that the royal cinclodes territories are 7-10 ac (3-4 ha). In Cordillera Vilcanota (Cusco, Peru), where a large concentration of the royal cinclodes individuals were observed in 2007, the average size of forest fragments just meets the lower threshold of the species' ecological requirements.

While the species' range is more widespread in Bolivia than previously understood, ongoing and accelerated habitat destruction of the remaining *Polylepis* forest fragments in Peru and Bolivia continues to reduce the quantity, quality, distribution, and regeneration of remaining patches. In the Administrative Region of Puno, Peru, habitat loss may have led to extirpation of the species from its type locality and the species may no longer be viable in that Region due to habitat loss. Current research indicates that climate fluctuations exacerbate the effects of habitat loss on species, especially for narrow endemics such as the royal cinclodes that are already undergoing range reduction due to

human activities. Historical decline in habitat availability is attributed to the same human activities that are causing habitat loss today, and climate models predict that this species' habitat will continue to decline. In addition, the royal cinclodes is "nervous" around humans, such that human encroachment is a particular problem. Human activities that degrade, alter, and destroy habitat are ongoing throughout the species' range, including within protected areas (see also Factor D). Experts consider the species' population decline to be commensurate with the declining habitat (Factor E). Therefore, we find that destruction and modification of habitat are threats to the continued existence of the royal cinclodes throughout its range.

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

We are not aware of any information currently available that indicates that overutilization of the royal cinclodes for commercial, recreational, scientific, or educational purposes has occurred or is occurring at this time. As a result, we are not considering overutilization to be a threat to the continued existence of the royal cinclodes.

C. Disease or Predation

We are not aware of any scientific or commercial information that indicate disease or predation poses a threat to this species. As a result, we are not considering disease or predation to be a threat to the continued existence of the royal cinclodes.

D. Inadequacy of Existing Regulatory Mechanisms

Existing regulatory mechanisms within Peru and Bolivia that have the potential to confer protection to the royal cinclodes or its habitat are analyzed on a country-by-country basis, beginning with Peru.

Peru: The royal cinclodes is considered "critically endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276855). This Decree prohibits hunting, take, transport, and trade of protected species, except as permitted by regulation. As hunting, take, transport, and trade do not currently threaten the royal cinclodes, this regulation does not mitigate any current threats to this species.

Peru has several categories of national habitat protection, which were described above as part of the Factor D analysis for the ash-breasted tit-tyrant (BLI 2008, p. 1; IUCN 1994, p. 2; Rodríguez and Young 2000, p. 330).

Protected areas have been established through regulation at one site occupied by the royal cinclodes in Peru: Santuario Histórico Machu Picchu (Cusco, Peru) (BLI 2009h, p. 4). Within the sanctuary, resources are supposed to be managed for conservation (Rodríguez and Young 2000, p. 330). However, habitat destruction and alteration, including burning to maintain pastures for grazing, are ongoing within Santuario Histórico Machu Picchu, preventing the regeneration of the woodlands (BLI 2009h, p. 4; Engblom *et al.* 2002, p. 58). Therefore, the occurrence of the royal cinclodes within protected areas in Peru does not protect the species, nor does it mitigate the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

Bolivia: The 1975 Law on Wildlife, National Parks, Hunting, and Fishing (Decree Law No. 12,301 1975, pp. 1-34; eLAW 2003, p. 2), was described above as part of the Factor D analysis for the ash-breasted tit-tyrant. This law designates national protection for all wildlife. However, there is no information as to the actual protections this confers to the species itself or its habitat, and ongoing habitat destruction throughout the species' range indicates that this law does not protect the species nor does it mitigate the threat to the species from ongoing habitat loss (Factor A) and concomitant population decline within Bolivia (Factor E).

Environmental Law No. 1333 (eLAW 2003, p. 1; Law No. 1,333 1992, pp. 1-26), was signed in 1992 to protect and conserve the environment. However, we are not aware that any specific legislation needed to implement these laws has been passed (eLAW 2003, p. 1). Therefore, this law does not protect the species, nor does it mitigate the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

Various levels of habitat protection in Bolivia were described above as part of the Factor D analysis for the ash-breasted tit-tyrant (eLAW 2003, p. 3; Supreme Decree No. 24,781 1997, p. 3). The royal cinclodes occurs within several protected areas in the Department of La Paz, Bolivia: Parque Nacional y área Natural de Manejo Integrado Madidi, Parque Nacional y área Natural de Manejo Integrado Cotapata, and the co-located protected areas of Reserva Nacional de Fauna de Apolobamba, área Natural de Manejo Integrado de Apolobamba, and Reserva de la Biosfera de Apolobamba (Auzá and Hennessey 2005, p. 81; BLI 2009a, p. 1; BLI 2009b, p. 1; BLI 2009c, p. 1; BLI 2009d, p. 1). Within Parque

Nacional y área Natural de Manejo Integrado Madidi, habitat destruction is caused by timber harvest used for construction, wood collection for firewood, and burning that often goes out of control to maintain pastures (BLI 2009a, p. 2; WCMC 1998a, p. 1). In addition, one of the most transited highways in the country is located a short distance from the Parque Nacional y área Natural de Manejo Integrado Cotapata; firewood collection and grazing also occur within the protected area (BLI 2009b, p. 2; BLI 2009c, p. 2). Within the Apolobamba protected areas, uncontrolled clearing, extensive agriculture, grazing, and tourism are ongoing (Auza and Hennessey 2005, p. 81; BLI 2009d, p. 5). Therefore, the occurrence of the royal cinclodes within protected areas in Bolivia does not protect the species, nor does it mitigate the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

Summary of Factor D

Peru and Bolivia have enacted various laws and regulatory mechanisms to protect and manage wildlife and their habitats. The royal cinclodes is "critically endangered" under Peruvian law and occurs within several protected areas in Peru and Bolivia. As discussed under Factor A, the royal cinclodes requires dense woodlands, which has been reduced by an estimated 99 percent in Peru and Bolivia. The remaining habitat for the royal cinclodes is fragmented and degraded. Habitat throughout the species' range has been and continues to be altered as a result of human activities, including clearcutting and burning for agriculture, grazing lands, and industrialization; extractive activities, including, firewood, timber, and mineral extraction; and human encroachment and concomitant increased pressure on natural resources. These activities are ongoing within protected areas and despite the species' critically endangered status in Peru, indicating that the laws governing wildlife and habitat protection in both countries are either inadequate or inadequately enforced to protect the species or to mitigate ongoing habitat loss (Factor A) and population declines (Factor E). Therefore, we find that the existing regulatory mechanisms are inadequate to mitigate the current threats to the continued existence of the royal cinclodes throughout its range.

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

An additional factor that affects the continued existence of the royal cinclodes is the species' small population size. Based on recent observations in Peru and Bolivia, the total population is between 239 and 259 individuals (Chutas 2007, pp. 4, 8; Gómez in litt. 2007, p. 1) (see Population Estimates), and BirdLife International characterizes the species as having an "extremely small population" size (BLI 2000, p. 345; BLI 2009i, p. 1). Although there is insufficient information to fully understand gene flow within this species (see Population Estimates), Engblom *et al.* (2002, p. 57) noted that the royal cinclodes may descend the mountains to forage in the valleys during periods of snow cover at the higher altitudes. Thus, interbreeding may occur at least among localities with shared valleys, but there is insufficient information to determine that the species breeds as a single population. Moreover, the total population size, which includes immature individuals, is not an accurate reflection of the species' effective population size (the number of breeding individuals that contribute to the next generation) (Shaffer 1981, pp. 132-133; Soulé 1980, pp. 160-162). Therefore, 239–259 is an overestimate of the species' effective population size.

Small population size renders species vulnerable to genetic risks that can have individual or population-level genetic consequences, such as inbreeding depression, loss of genetic variation, and accumulation of new mutations. These genetic problems may affect the species' viability by increasing its susceptibility to demographic shifts or environmental fluctuations, as described above in the Factor E analysis for the ash-breasted tit-tyrant (Charlesworth and Charlesworth 1987, p. 238; Pimm *et al.* 1988, pp. 757, 773-775; Shaffer 1981, p. 131).

Small population size leads to a higher risk of extinction and, once a population is reduced below a certain number of individuals, it tends to rapidly decline towards extinction (Frankham 1996, p. 1507; Franklin 1980, pp. 147-148; Gilpin and Soulé 1986, p. 25; Holsinger 2000, pp. 64-65; Purvis *et al.* 2000, p. 1949; Reed and Frankham 2003, pp. 233-234; Soulé 1987, p. 181). If further research indicates that interbreeding does not between subpopulations, this would heighten the risks to the species

associated with short-term genetic viability.

Complications arising from the species' small population size are exacerbated by the species' fragmented distribution. The royal cinclodes is currently restricted to high-elevation, moist, moss-laden patches of semihumid woodlands in Peru and Bolivia (BLI 2009i, p. 6) (Factor A). Fjeldså and Kessler (1996, as cited in Fjeldså 2002a, p. 113). Remaining *Polylepis* woodlands are highly fragmented and degraded, and it is estimated that only 1 percent of the dense woodlands preferred by the species remain (del Hoyo *et al.* 2003, p. 253; Engblom *et al.* 2002, p. 57) (see Habitat and Life History and Historical Distribution). Therefore, the species' current range is restricted and severely fragmented (BLI 2000, p. 345; BLI 2009i, pp. 1-2; Collar *et al.* 1992, p. 588; del Hoyo *et al.* 2003, p. 253). Habitat fragmentation can cause genetic isolation and heighten the risks to the species associated with short-term genetic viability. The royal cinclodes has undergone a population decline between 30 and 49 percent in the past 10 years, in close association with the continued loss and degradation of the *Polylepis* forest (BLI 2009i, p. 6) (Factor A). The species' small population size, combined with its restricted and severely fragmented range, increases the species' vulnerability to adverse natural events and manmade activities that destroy individuals and their habitat (Holsinger 2000, pp. 64-65; Primack 1998, pp. 279-308; Young and Clarke 2000, pp. 361-366).

Summary of Factor E

The royal cinclodes has a small population size that renders it vulnerable to genetic risks that negatively impact the species' long-term viability, and possibly its short-term viability. The species has a restricted range and occurs in highly fragmented habitat that continues to undergo degradation and curtailment due to human activities (Factor A). The restricted and fragmented range, as well as the small population size, increases the species' vulnerability to extinction through demographic or environmental fluctuations. Based on its small population size and fragmented distribution, we have determined that the royal cinclodes is particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., deforestation, habitat alteration, and infrastructure development) that destroy individuals and their habitat. The genetic and

demographic risks associated with small population sizes are exacerbated by ongoing human activities that continue to curtail the species' habitat throughout its range. The species' population has declined and is predicted to continue declining commensurate with ongoing habitat loss (Factor A). Therefore, we find that the species' small population size, in concert with its fragmented distribution and its heightened vulnerability to adverse natural events and manmade activities, are threats to the continued existence of the royal cinclodes throughout its range.

Status Determination for the Royal Cinclodes

The royal cinclodes, a large-billed ovenbird, is native to the high-altitude, semihumid *Polylepis* or *Polylepis - Gynoxys* woodlands of the Bolivian and Peruvian Andes, where it occupies a narrow range of distribution at elevations between 11,483 and 12,092 ft (3,500 and 4,600 m). Preferring dense woodlands with more closed canopies, the royal cinclodes is a ground-feeding insectivore that probes the mossy forest undergrowth for food. The species has a highly restricted and severely fragmented range (approximately 1,042 mi² (2,700 km²)), and is found only in the Peruvian Administrative Regions of Apurímac, Cusco, Júpín, and Puno, and in the Bolivian Department of La Paz. The known population of the royal cinclodes is estimated to be 239–259 individuals.

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the royal cinclodes and have concluded that there are three primary factors impacting the continued existence of the royal cinclodes: (1) Habitat destruction, fragmentation, and degradation; (2) limited size and isolation of remaining populations; and (3) inadequate regulatory mechanisms.

Human activities that degrade, alter, and destroy habitat are ongoing throughout the royal cinclodes' range. Widespread deforestation and the conversion of forests for grazing, agriculture, and human settlement have led to the fragmentation of habitat throughout the range of the royal cinclodes (Factor A). Researchers estimate that only 1 percent of the dense *Polylepis* woodlands preferred by the species remain extant. Limited by the availability of suitable habitat, the species occurs today only in some of these fragmented and disjunct locations. Royal cinclodes habitat is particularly vulnerable to the drying effects

accompanied by diminished forest cover.

Royal cinclodes habitat continues to be altered by human activities, which result in the continued degradation, conversion, and destruction of habitat and reduction of the quantity, quality, distribution, and regeneration of remaining patches. Habitat loss was a factor in this species' historical decline, resulting in extirpation of the species from its type locality and, possibly, the loss of a viable population in the entire Region of Puno, Peru (see Historical Range and Distribution). The species' population is considered to be declining in association with the reduction in habitat (Factors A and E).

The royal cinclodes population is small, rendering the species particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., deforestation and habitat alteration) that destroy individuals and their habitat. Human activities that continue to curtail the species' habitat throughout its range exacerbate the genetic and demographic risks associated with small population sizes (Factor E). The population has declined 30–49 percent in the past 10 years (see Population Estimates), and is predicted to continue declining commensurate with ongoing habitat loss (Factor A). Current research indicates that narrow endemics, such as the royal cinclodes, are especially susceptible to climate fluctuations because of the synergistic effect these fluctuations have on declining populations that are also experiencing range reductions due to human activities (Factor A).

Despite the species' "critically endangered" status in Peru and its occurrence within several protected areas in Peru and Bolivia (Factor D), the species' habitat continues to be destroyed and degraded as a result of human activities (Factor A), which corresponds with population declines (Factor E). Therefore, regulatory mechanisms are either inadequate or ineffective at mitigating the existing threats to the royal cinclodes and its habitat (Factor D).

Section 3 of the Act defines an "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range" and a "threatened species" as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Based on the immediate and ongoing threats to the royal cinclodes throughout its entire range, as described above, we determine that the royal cinclodes is in danger of

extinction throughout all of its range. Therefore, on the basis of the best available scientific and commercial information, we are proposing to list the royal cinclodes as an endangered species throughout all of its range.

VI. White-browed tit-spinetail (*Leptasthenura xenothorax*)

Species Description

The white-browed tit-spinetail, or "*tijeral cejiblanco*," is a small dark ovenbird in the Furnaridaii family that is native to high-altitude woodlands of the Peruvian Andes (BLI 2000, p. 347; Chapman 1921, pp. 8–9; del Hoyo *et al.* 2003, pp. 266–267; Fjeldsá and Krabbe 1990, p. 348; Parker and O'Neill 1980, p. 169). The sexes are similar, with individuals approximately 7 in (18 cm) in length. The species is characterized by its bright rufous crown and prominent white supercilium (eyebrow) (del Hoyo *et al.* 2003, p. 267; Lloyd 2009, p. 2), which gives the species its name. The species is highly vocal, "often singing while acrobatically foraging from the outermost branches of *Polylepis* trees" (Lloyd 2009, p. 2).

Taxonomy

The white-browed tit-spinetail was first described by Chapman in 1921 (del Hoyo *et al.* 2003, p. 267). The species has been synonymized with the nominate subspecies of the rusty-crowned tit-spinetail (*Leptasthenura pileata pileata*) by Vaurie (1980, p. 66), but examination of additional specimens in combination with field observations strongly suggests that *L. xenothorax* is a valid species (Collar *et al.* 1992, p. 596; Fjeldsá and Krabbe 1990, p. 348; Parker and O'Neill 1980, p. 169). Therefore, we accept the species as *Leptasthenura xenothorax*, which also follows the Integrated Taxonomic Information System (ITIS 2009, p. 1).

Habitat and Life History

The white-browed tit-spinetail is restricted to high-elevation, semihumid *Polylepis* and *Polylepis-Gynoxys* woodlands, where the species is found between 12,139 and 14,928 ft (3,700 and 4,550 m) above sea level (BLI 2000, p. 347; Collar *et al.* 1992, p. 595; del Hoyo *et al.* 2003, p. 267; Fjeldsá and Krabbe 1990, p. 348; Lloyd 2009, pp. 5–6). The characteristics of *Polylepis* habitat were described above as part of the Habitat and Life History of the ash-breasted tit-tyrant (Aucca and Ramsay 2005, p. 1; Chutas *et al.* 2008, p. 3; De la Via 2004, p. 10; IPNI 2009, p. 1; Kessler 1998, p. 1; Purcell *et al.* 2004, p. 455). The white-browed tit-spinetail prefers areas of primary forest that have a high density of tall, large trees and dense *Polylepis*

stands and vegetation cover (Lloyd 2008a, as cited in Lloyd 2009, p. 6).

Dense stands of *Polylepis* woodlands are characterized by moss-laden vegetation and a shaded understory, and provide for a rich diversity of insects, making these areas good feeding grounds for insectivorous birds (De la Via 2004, p. 10), such as the white-browed tit-spinetail (BLI 2009d, p. 2). According to Engblom *et al.* (2002, pp. 57-58), the species has been recorded in patches of woodland as small as 0.6 ac (0.25 ha) in Cordillera Vilcabamba. Based on these observations, Engblom *et al.* (2002, p. 58) suggest that the species is able to persist in very small forest fragments, especially if a number of these patches are in close proximity. The lower elevation of this species' range changes to a mixed *Polylepis-Escallonia* (no common name) woodland, and the white-browed tit-spinetail has been observed there on occasion, such as during a snowstorm (Collar *et al.* 1992, p. 595; del Hoyo *et al.* 2003, p. 267; Fjelds  and Krabbe 1990, p. 348).

There is limited information the ecology and breeding behavior of the white-browed tit-spinetail. Lloyd (2006, as cited in Lloyd 2009, p. 8) reports that the species breeds in October in Cordillera Vilcanota in southern Peru. In the same area, one adult was seen attending a nesting hole in a *Polylepis* tree in November 1997 (del Hoyo *et al.* 2003, p. 267; C. Bushell in litt. (1999), as cited in BLI 2009d, p. 2). Only one nest of the white-browed tit-spinetail has ever been described. According to Lloyd (2006, as cited in Lloyd 2009, p. 8), the nest was located within a natural cavity of a *Polylepis racemosa* tree's main trunk, approximately 7 ft (2 m) above the ground. To construct their nest, the white-browed tit-spinetail pair uses moss, lichen, and bark fibers they stripped from *Polylepis* tree trunks, large branches and large boulders while foraging. The nest was cup-shaped and contained two pale-colored eggs (Lloyd 2006, as cited in Lloyd 2009, p. 8).

The white-browed tit-spinetail is insectivorous, with a diet consisting primarily of arthropods (del Hoyo *et al.* 2003, p. 267; Lloyd 2009, p. 7). The species forages in pairs or small family groups of three to five, and often in mixed-species flocks, gleaning insects from bark crevices, moss, and lichens on twigs, branches, and trunks (BLI 2009d, pp. 2-3; Engblom *et al.* 2002, pp. 57-58; Parker and O'Neill 1980, p. 169). The white-browed tit-spinetail is highly arboreal, typically foraging acrobatically from the outer branches of *Polylepis* trees while hanging upside-down (del

Hoyo *et al.* 2003, p. 267; Lloyd 2008b, as cited in Lloyd 2009, p. 7).

Historical Range and Distribution

In our 2008 Annual Notice of Findings on Resubmitted Petitions for Foreign Species (73 FR 44062; July 29, 2008), we stated that historically, the white-browed tit-spinetail may have occupied the *Polylepis* forests of the high-Andes of Peru and Bolivia. We included both countries in the historical range of the species because the species' primary habitat, the *Polylepis* forest, was historically large and contiguous throughout the high-Andes of both Peru and Bolivia (Fjelds  2002a, p. 115). However, based on further research, we have determined that historically, the species was only known from two Regions in south-central Peru, Cusco and Apur mac (Collar *et al.* 1992, p. 594; del Hoyo *et al.* 2003, p. 267), and not in Bolivia.

The white-browed tit-spinetail may once have been distributed throughout south-central Peru, in previously contiguous *Polylepis* forests above 9,843 ft (3,000 m) (BLI 2000, p. 347; BLI 2009d, pp. 1-2; Fjelds  2002a, pp. 111-112, 115; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97-101). However, *Polylepis* woodlands are now restricted to elevations of 11,483 to 16,404 ft (3,500 to 5,000 m) (Fjelds  1992, p. 10). As discussed above for the Historical Range and Distribution of the ash-breasted tit-tyrant, researchers consider human activity to be the primary cause for historical habitat decline and resultant decrease in species richness (Fjelds  and Kessler 1996, Kessler 1995a, b, and L egaard 1992, as cited in Fjelds  2002a, p. 112; Fjelds  2002a, p. 116; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97-101; Kessler and Herzog 1998, pp. 50-51). It is estimated that only 2-3 percent of the original forest cover still remains in Peru (Fjelds  2002a, pp. 111, 113). Less than 1 percent of the remaining woodlands occur in humid areas, where denser stands are found (Fjelds  and Kessler 1996, as cited in Fjelds  2002a, p. 113), and which are preferred by the white-browed tit-spinetail (BLI 2009d, p. 2; Lloyd 2008a, as cited in Lloyd 2009, p. 6).

Current Range and Distribution

Today, the white-browed tit-spinetail is restricted to high-elevation, semihumid patches of *Polylepis* and *Polylepis-Gynoxys* woodlands in the Andes mountains of south-central Peru, where the species occurs between 12,139-14,928 ft (3,700-4,550 m) (BLI 2000, p. 347; Collar *et al.* 1992, p. 595; del Hoyo *et al.* 2003, p. 267; Fjelds  and

Krabbe 1990, p. 348; InfoNatura 2007, p. 1; Lloyd 2009, pp. 1, 5-6). The species has a highly restricted and severely fragmented range, and is currently known from only a small number of sites: The Runtacocha highlands (in Apur mac Region) and the Nevado Sacsarayoc massif, Cordillera Vilcabamba (Chapman 1921, p. 8), and Cordillera Vilcanota (in the Cusco Region) (BLI 2000, p. 347; BLI 2009d, p. 2; Lloyd 2009, p. 5). The estimated range of the species is approximately 965 mi² (2,500 km²) (BLI 2000, p. 347; BLI 2009d, pp. 1, 5).

Population Estimates

Population information is presented first on the local level and then in terms of a global population estimate.

Local population estimates: Between 1987 and 1989, populations of 35-70 individuals were estimated to occur at 3 sites in Cusco; since then, declines in the populations at some of these sites have been observed (Fjelds  and Kessler 1996, as cited in BLI 2000, p. 347). At Abra M laga (Cusco Region), it is estimated that there are approximately 30-50 birds (del Hoyo *et al.* 2003, p. 267; Engblom *et al.* 2002, p. 58). In the Runtacocha highlands (Apur mac Region), the population density of the white-browed tit-spinetail is very low (Fjelds  and Kessler 1996, as cited in BLI 2000, p. 347). Chutas (2007, p. 8) reported an estimated 305 birds located within 3 disjunct *Polylepis* forest patches in Peru. This included 205 birds and 36 birds in Cordilleras Vilcanota and Vilcabamba, respectively (Cusco), and 64 birds in "Cordillera del Apur mac" (Runtacocha highlands of Apur mac) (Chutas 2007, p. 8).

Density estimates derived from surveys conducted at 3 sites in Cordillera Vilcanota range from 25.3 (\pm 15.1) individuals per km², to 9.6 (\pm 21.7) individuals per km², and the species may occur at even higher densities in other areas of *Polylepis* forests (Lloyd 2008c, as cited in Lloyd 2009, p. 9). According to Lloyd (2008c, as cited in Lloyd 2009, p. 9), this quantitative data from Cordillera Vilcanota shows that the white-browed tit-spinetail is "one of the most abundant *Polylepis* specialists in southern Peru."

Global population estimate: BirdLife International categorizes the white-browed tit-spinetail as having a population size between 500 and 1,500 individuals (BLI 2009d, pp. 1, 5). The category was determined from the population estimates reported by Engblom *et al.* 2002 (p. 58), who estimated "the known population to be around 500 individuals with a possible total population of 1,500 individuals."

In 2002, Fjelds  (2002b, p. 9) estimated a total population size of between 250 and 1,000 pairs of birds, which coincides with the BirdLife International category of 500–1,500 individuals.

The species has experienced a population decline of between 10 and 19 percent in the past 10 years, and this rate of decline is predicted to continue (BLI 2009d, p. 5). The population is considered to be declining in close association with continued habitat loss and degradation (see Factors A and E) (BLI 2009d, p. 6).

Conservation Status

The white-browed tit-spinetail is considered “endangered” by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276854). The IUCN considers the white-browed tit-spinetail to be “Endangered” due to its very small and severely fragmented range and population, which continue to decline with ongoing habitat loss and a lack of habitat regeneration (BLI 2009d, p. 1). The white-browed tit-spinetail occurs within the Peruvian protected area of Santuario Hist rico Machu Picchu in Cusco (BLI 2009c, pp. 1, 3; BLI 2009d, p. 6; del Hoyo *et al.* 2003, p. 267).

Summary of Factors Affecting the White-browed Tit-Spinetail

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

The white-browed tit-spinetail is restricted to high-elevation, semihumid *Polylepis* and *Polylepis-Gynoxys* woodlands, where it is found between 12,139 and 14,928 ft (3,700 and 4,550 m) above sea level (BLI 2000, p. 347; Collar *et al.* 1992, p. 595; del Hoyo *et al.* 2003, p. 267; Fjelds  and Krabbe 1990, p. 348; Lloyd 2009, pp. 1, 5-6). High-Andean *Polylepis* habitat is characterized as a threatened woodland ecosystem on national, regional, and global levels (Purcell *et al.* 2004, p. 457; Renison *et al.* 2005, as cited in Lloyd 2009, p. 10), with several *Polylepis* species within the white-browed tit-spinetail’s range considered to be “Vulnerable,” according to the IUCN (WCMC 1998a, p. 1; WCMC 1998b, p. 1). As described more fully for the ash-breasted tit-tyrant (Factor A), *Polylepis* woodlands have been much reduced from historical estimates, with an estimated remaining area of 386 mi² (1,000 km²) in Peru (Fjelds  and Kessler 1996, as cited in Fjelds  2002a, p. 113). The majority of these remaining forests are much dispersed, and less than 1

percent is located in the humid parts of the highlands, where denser stands occur (Fjelds  and Kessler 1996, as cited in Fjelds  2002a, p. 113). The white-browed tit-spinetail prefers areas of dense *Polylepis* primary forest with understory vegetation that provides optimal foraging habitat (BLI 2009d, p. 2; De la Via 2004, p. 10; Lloyd 2008a, as cited in Lloyd 2009, p. 6) (see Habitat and Life History).

In the Cordillera de Vilcanota (Cusco, Peru), where a large portion of the known white-browed tit-spinetail population occurs (205 birds were observed there, out of 305 total birds observed in 3 study sites in Peru) (Chutas 2007, p. 8), *Polylepis* woodland habitat is highly fragmented and degraded. As described more fully for the ash-breasted tit-tyrant (Factor A), recent research indicated that:

(1) Four forest patches in the Cordillera de Vilcanota disappeared completely in the last half a century, and the remaining *Polylepis* remnants are small (with a mean patch size of 7.4 ac (3 ha) (Jameson and Ramsay 2007, p. 42) and commonly separated from the larger patches by distances of 98–4,921 ft (30–1,500 m) (Lloyd and Marsden in press, as cited in Lloyd 2008, p. 532);

(2) Ten percent of the remaining forest patches showed a decline in forest density (Jameson and Ramsay 2007, p. 42); and

(3) There were no indications of forest regeneration within the study area.

These findings have consequences for the white-browed tit-spinetail given the species’ ecological requirements. As *Polylepis* woodlands decline in number, the distances between patches increase. According to Engblom *et al.* (2002, pp. 57-58), the species has been recorded in patches of woodland as small as 0.6 ac (0.25 ha) in Cordillera Vilcabamba, but the species’ persistence in small patches appears to be dependent on the patches being in close proximity to each other. Habitat degradation impacts the white-browed tit-spinetail, given its preference for dense *Polylepis* woodlands, where optimal foraging habitat is found (BLI 2009d, p. 2; De la Via 2004, p. 10; Lloyd 2008a, as cited in Lloyd 2009, p. 6). The lack of *Polylepis* forest regeneration in the area over nearly 50 years underscores the ramifications of continued burning and clearing to maintain pastures and farmland that are prevalent throughout the white-browed tit-spinetail’s range (BLI 2009a, p. 1; Engblom *et al.* 2002, p. 56; Fjelds  2002a, pp. 112, 120; Fjelds  2002b, p. 8; Purcell *et al.* 2004, p. 458; WCMC 1998a, p. 1; WCMC 1998b, p. 1).

Habitat loss, conversion, and degradation throughout the white-

browed tit-spinetail’s range are attributed to human activities (a full description of which is provided above as part of the Factor A analysis for the ash-breasted tit-tyrant). Ongoing activities include:

(1) Clearcutting and uncontrolled burning for agriculture and pastureland for domesticated animals, all of which contributes to soil erosion, and habitat degradation, which prevent forest regeneration and restrict *Polylepis* woodlands to localized and highly fragmented landscapes (BLI 2009a, p. 2; BLI 2009b, p. 1; BLI 2009c, p. 3; BLI 2009d, p. 3; Engblom *et al.* 2002, p. 56; Fjelds  2002a, pp. 112, 120; Fjelds  2002b, p. 8; Jameson and Ramsay 2007, p. 42; Purcell *et al.* 2004, p. 458; Renison *et al.* 2006, as cited in Lloyd 2009, p. 11; WCMC 1998a, p. 1; WCMC 1998b, p. 1);

(2) Extractive activities, such as harvest for timber, firewood, and charcoal, for use on local- and commercial-scales as fuel, construction, fencing and tool-making (Aucca and Ramsay 2005, p. 287; BLI 2009a, p. 2; BLI 2009b, p. 1; BLI 2009d, p. 3; Engblom 2000, pp. 1-2; Engblom *et al.* 2002, p. 56; Fjelds  and Kessler 1996, as cited in BLI 2009d, p. 3; Purcell *et al.* 2004, pp. 458-459; WCMC 1998a, p. 1); and

(3) Unpredictable climate fluctuations that exacerbate the effects of habitat fragmentation (Jetz *et al.* 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027). These habitat-altering activities are ongoing throughout the range of the white-browed tit-spinetail, including in Apur mac (BLI 2009b, p. 1) and Cusco (BLI 2009a, pp. 1-2; BLI 2009c, pp. 1-3) of south-central Peru and within the one protected area in which the species occurs, Santuario Hist rico Machu Picchu (BLI 2009c, p. 3).

Polylepis conservation programs have been under way in Peru since 2004, including in Cordilleras Vilcanota and Vilcabamba and highlands of the Apur mac Region, where white-browed tit-spinetail also occurs (Aucca and Ramsey 2005, p. 287; Chutas 2007, p. 8; ECOAN n.d., p. 1; Lloyd 2009, p. 10). These community-based programs, which are more fully described above as part of the Factor A analysis for the ash-breasted tit-tyrant, confront the primary causes of *Polylepis* deforestation: Burning, grazing, and wood-cutting. One such program, called the “Vilcanota Project,” is under way at three locations in the Cordillera de Vilcanota (Abra M laga, Hulloc, and Cancha-Cancha) (Aucca and Ramsay 2005, p. 287; ECOAN n.d., p. 1; Lloyd 2009, p. 10). Since local populations rely on *Polylepis* wood for firewood and

charcoal production (Aucca and Ramsay 2005, p. 287; Engblom *et al.* 2002, p. 56), the Vilcanota Project works to deliver non-*Polylepis* firewood to families for cooking, as well as supply them with fuel-efficient cooking stoves (ECOAN n.d., p. 1). A short-term aim of these projects is to restore balance to local sustainable resource use (Aucca and Ramsay 2005, p. 288; ECOAN n.d., p. 1). However, at Abra Málaga (one of the Vilcanota Project's sites), *Polylepis* woodlands continue to be impacted by extraction for firewood and burning for agriculture and pastureland (BLI 2009a, pp. 1-2). In addition, commercial-scale activities, such as logging and fuel wood collection, which are ongoing throughout this species' range, alter otherwise sustainable resource use practices (Aucca and Ramsay 2005, p. 287; Engblom 2000, p. 2; Engblom *et al.* 2002, p. 56; MacLennan 2009, p. 2; Purcell *et al.* 2004, pp. 458-459; WCMC 1998a, p. 1).

Habitat destruction caused by burning and grazing, which have prevented regeneration of habitat, is a factor in the historical decline of *Polylepis*-dependent bird species (Fjeldsá 2002a, p. 116). The white-browed tit-spinetail's population size is considered to be declining in close association with the continued habitat loss and degradation of *Polylepis* woodlands (BLI 2009d, p. 6). The species may once have been distributed throughout south-central Peru, in once contiguous *Polylepis* forests (BLI 2000, p. 347; BLI 2009d, pp. 1-2; Fjeldsá 2002a, pp. 111-112, 115; Herzog *et al.* 2002, p. 94; Kessler 2002, pp. 97-101). Today, the species has a highly restricted and severely fragmented range, and is currently known from only a small number of sites in the Regions of Apurímac and Cusco in south-central Peru (BLI 2000, p. 347; BLI 2009d, pp. 1-2; Lloyd 2009, p. 5).

White-browed tit-spinetails are also impacted by unpredictable climate fluctuations, which are more fully described under the Factor A analysis of the ash-breasted tit-tyrant and are summarized here. Peru is subject to unpredictable climate fluctuations that exacerbate the effects of habitat fragmentation, such as those that are related to the El Niño Southern Oscillation (ENSO). Changes in weather patterns, such as ENSO cycles (El Niño and La Niña events), tend to increase precipitation in normally dry areas, and decrease precipitation in normally wet areas (Holmgren *et al.* 2001, p. 89; TAO Project n.d., p. 1), exacerbating the effects of habitat fragmentation on the decline of a species (England 2000, p. 86; Holmgren *et al.* 2001, p. 89; Jetz *et*

al. 2007, pp. 1211, 1213; Mora *et al.* 2007, p. 1027; Parmesan and Mathews 2005, p. 334; Plumart 2007, pp. 1-2; Timmermann 1999, p. 694), especially for narrow endemics (Jetz *et al.* 2007, p. 1213) such as the white-browed tit-spinetail (see also Factor E). ENSO cycles strongly influence the growth of *Polylepis* species (Christie *et al.* 2008, p. 1) by altering *Polylepis* species' age structure and mortality, especially where woodlands have undergone disturbance, such as fire and grazing (Villalba and Veblen 1997, pp. 121-123; Villalba and Veblen 1998, pp. 2624, 2637). These cycles may have already accelerated the fire cycle (Block and Richter 2007, p. 1; Power *et al.* 2007, pp. 897-898), which is a key factor preventing *Polylepis* regeneration (Fjeldsá 2002a, p. 112, 120; Fjeldsá 2002b, p. 8) because *Polylepis* species recover poorly following a fire (Cierjacks *et al.* 2007, p. 176). ENSO cycles are ongoing, having occurred several times within the last decade (NWS 2009, p. 2), and evidence suggests that ENSO cycles have already increased in periodicity and severity (Richter 2005, pp. 24-25; Timmermann 1999, p. 694), which will exacerbate the negative impacts of habitat destruction on a species. It is predicted that, by 2050, another 1 percent of the white-browed tit-spinetail's current remaining range is likely to be unsuitable for this species due to climate change; and, by 2100, it is predicted that about 43 percent of the species' range is likely to be lost as a direct result of global climate change (Jetz *et al.* 2007, Supplementary Table 2, p. 89).

Summary of Factor A

Polylepis habitat throughout the range of the white-browed tit-spinetail has been and continues to be altered and destroyed as a result of human activities, including clearcutting and burning for agriculture and grazing lands and extractive activities, including harvest for timber, firewood, and charcoal. It is estimated that only 1 percent of the dense *Polylepis* woodlands preferred by the species remain, and the remaining woodlands are highly fragmented and degraded. Observations suggest that the white-browed tit-spinetail is able to persist in very small forest fragments (e.g., areas as small as 0.6 ac (0.25 ha) in Cordillera Vilcabamba); however, this depends on whether or not a number of patches are in close proximity to one another. Since the remaining *Polylepis* woodlands are and continue to be severely fragmented, the distance between some of the small woodland patches may be too large for the species to be able to persist. Today,

the species is known from only a small number of sites at four locations: The Runtacocha highlands (in Apurímac Region), and the Nevado Sacsarayoc massif, Cordillera Vilcabamba, and Cordillera Vilcanota (in Cusco Region). Historical decline in habitat availability is attributed to the same human activities that are causing habitat loss today. Ongoing and accelerated habitat destruction of the remaining *Polylepis* woodlands in Peru continues to reduce the quantity, quality, distribution, and potential regeneration of *Polylepis* forests. Human activities that degrade, alter, and destroy habitat are ongoing throughout the species' range, including within protected areas (see also Factor D). Experts consider the species' population decline to be commensurate with the declining habitat (Factor E). Current research indicates that climate fluctuations exacerbate the effects of habitat loss to species, especially for narrow endemics such as the white-browed tit-spinetail that are already undergoing range reduction due to human activities. Climate models predict that this species' habitat will continue to decline. Therefore, we find that destruction and modification of habitat are threats to the continued existence of the white-browed tit-spinetail throughout its range.

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

We are not aware of any information currently available that indicates that overutilization of the species for commercial, recreational, scientific, or educational purposes has occurred or is occurring at this time. As a result, we are not considering overutilization to be a threat to the continued existence of the white-browed tit-spinetail.

C. Disease or Predation

We are not aware of any scientific or commercial information that indicates that disease or predation poses a threat to the species. As a result, we are not considering disease or predation to be a threat to the continued existence of the white-browed tit-spinetail.

D. Inadequacy of Existing Regulatory Mechanisms

The white-browed tit-spinetail is considered "endangered" by the Peruvian Government under Supreme Decree No. 034-2004-AG (2004, p. 276854). This Decree prohibits hunting, take, transport, and trade of protected species, except as permitted by regulation. As hunting, taking, or trade do not currently threaten the white-browed tit-spinetail, this regulation does

not mitigate any current threats to the species.

Peru has several categories of national habitat protection, which were described above as part of Factor D for the ash-breasted tit-tyrant (IUCN 1994, p. 2; Rodríguez and Young 2000, p. 330). Protected areas have been established through regulation at one site occupied by the white-browed tit-spinetail in Peru: the Santuario Histórico Machu Picchu (Cusco, Peru); (BLI 2009c, pp. 1, 3; BLI 2009d, p. 6). Resources within Santuario Histórico Machu Picchu are supposed to be managed for conservation (Rodríguez and Young 2000, p. 330). However, habitat destruction and alteration, including burning, cutting, and grazing are ongoing within the sanctuary, preventing regeneration of the woodlands (BLI 2009c, p. 3; Engblom *et al.* 2002, p. 58). Therefore, the occurrence of the white-browed tit-spinetail within protected areas in Peru does not protect the species, nor does it mitigate the threats to the species from ongoing habitat loss (Factor A) and concomitant population decline (Factor E).

Summary of Factor D

Peru has enacted various laws and regulatory mechanisms to protect and manage wildlife and their habitats. The white-browed tit-spinetail is “endangered” under Peruvian law and occurs within one protected area in Peru. As discussed under Factor A, the white-browed tit-spinetail prefers dense *Polylepis* woodlands, which have been reduced by an estimated 98 percent in Peru. The *Polylepis* habitat that does remain is highly fragmented and degraded. Habitat throughout the species’ range has been and continues to be altered as a result of human activities, including clearcutting and burning for agriculture and grazing lands; and extractive activities such as timber harvest, firewood collection, and charcoal production. These activities are ongoing within protected areas despite the species’ endangered status, indicating that the laws governing wildlife and habitat protection in Peru are either inadequate or inadequately enforced to protect the species or to mitigate ongoing habitat loss (Factor A) and population declines (Factor E). Therefore, we find that the existing regulatory mechanisms are inadequate to mitigate the current threats to the continued existence of the white-browed tit-spinetail throughout its range.

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

An additional factor that affects the continued existence of the white-browed tit-spinetail is the species’ small population size. As discussed above (see Population Estimates), BirdLife International has placed the white-browed tit-spinetail in the population category of between 500 and 1,500 individuals (BLI 2009d, pp. 1, 5), and characterizes the species as having a “very small population” size (BLI 2000, p. 347; BLI 2009d, p. 1).

Small population size renders species vulnerable to genetic risks that can have individual or population-level genetic consequences, such as inbreeding depression, loss of genetic variation, and accumulation of new mutations, and may affect the species’ viability by increasing its susceptibility to demographic shifts or environmental fluctuations, as explained in more detail above in the Factor E analysis for the ash-breasted tit-tyrant (Charlesworth and Charlesworth 1987, p. 238; Pimm *et al.* 1988, pp. 757, 773-775; Shaffer 1981, p. 131). Small population size leads to a higher risk of extinction and, once a population is reduced below a certain number of individuals, it tends to rapidly decline towards extinction (Frankham 1996, p. 1507; Franklin 1980, pp. 147-148; Gilpin and Soulé 1986, p. 25; Holsinger 2000, pp. 64-65; Purvis *et al.* 2000, p. 1949; Reed and Frankham 2003, pp. 233-234; Soulé 1987, p. 181).

Complications arising from the species’ small population size are exacerbated by the species’ fragmented distribution. The white-browed tit-spinetail is currently confined to high-elevation, semihumid patches of forest in the Andes of Peru, and its population has declined at a rate between 10 and 19 percent in the past 10 years, in close association with the continued loss and degradation of the *Polylepis* forest (BLI 2009d, pp. 5-6) (Factor A). Fjeldså and Kessler (1996, as cited in Fjeldså 2002a, p. 113) describe the remaining *Polylepis* woodlands as highly fragmented and degraded, and estimate that only 1 percent of the dense woodlands preferred by the species remain (BLI 2009d, p. 2; De la Via 2004, p. 10; Lloyd 2008a, as cited in Lloyd 2009, p. 6) (see Habitat and Life History). As a result, experts say that the species’ current range is highly restricted and severely fragmented (BLI 2000, p. 347; BLI 2009d, p. 1; Collar *et al.* 1992, p. 595; del Hoyo *et al.* 2003, p. 267; Fjeldså and Krabbe 1990, p. 348; InfoNatura 2007, p. 1; Lloyd 2009, p. 5). The species’ small

population size, combined with its highly restricted and severely fragmented range, increases the species’ vulnerability to adverse natural events and manmade activities that destroy individuals and their habitat (Holsinger 2000, pp. 64-65; Primack 1998, pp. 279-308; Young and Clarke 2000, pp. 361-366).

Summary of Factor E

The white-browed tit-spinetail has a small population size that renders it vulnerable to genetic risks that negatively impact the species’ viability. The species has a severely restricted range and occurs in highly fragmented habitat that continues to undergo degradation and curtailment due to human activities (Factor A). The restricted and fragmented range, as well as the small population size, increases the species’ vulnerability to extinction through demographic or environmental fluctuations. Based on its small population size and fragmented distribution, we have determined that the white-browed tit-spinetail is particularly vulnerable to the threat of adverse natural events (e.g., genetic, demographic, or environmental) and human activities (e.g., deforestation and habitat alteration) that destroy individuals and their habitat. The genetic and demographic risks associated with small population sizes are exacerbated by ongoing human activities that continue to curtail the species’ habitat throughout its range. The species’ population has declined and is predicted to continue declining commensurate with ongoing habitat loss (Factor A). Therefore, we find that the species’ small population size, in concert with its fragmented distribution and its heightened vulnerability to adverse natural events and manmade activities, are threats to the continued existence of the white-browed tit-spinetail throughout its range.

Status Determination for the White-Browed Tit-Spinetail

The white-browed tit-spinetail, a small dark ovenbird, is restricted to high-altitude woodlands of the Peruvian Andes. Preferring dense, semihumid *Polylepis* and *Polylepis-Gynoxys* woodlands, the ash-breasted tit-tyrant occupies a narrow range of distribution at elevations between 12,139 and 14,928 ft (3,700 and 4,550 m) above sea level. The species has a highly restricted and severely fragmented range (approximately 965 mi² (2,500 km²)), and is currently known from only a small number of sites in the Apurímac and Cusco Regions, in south-central Peru. The known population of the

white-browed tit-spinetail is estimated to be approximately 500 to 1,500 individuals.

We have carefully assessed the best available scientific and commercial information regarding the past, present, and potential future threats faced by the white-browed tit-spinetail. There are three primary factors impacting the continued existence of the white-browed tit-spinetail: (1) Habitat destruction, fragmentation, and degradation; (2) limited size and isolation of remaining populations; and (3) inadequate regulatory mechanisms.

Human activities that degrade, alter, and destroy habitat are ongoing throughout the white-browed tit-spinetail. Widespread deforestation and the conversion of forests for grazing and agriculture have led to the fragmentation of habitat throughout the range of the white-browed tit-spinetail (Factor A). Researchers estimate that only 1 percent of the dense *Polylepis* woodlands preferred by the species remain. Limited by the availability of suitable habitat, the species occurs today only in a few fragmented and disjunct locations.

White-browed tit-spinetail habitat continues to be altered by human activities, which result in the continued degradation, conversion, and destruction of habitat and reduction of the quantity, quality, distribution, and regeneration of remaining forest patches. Habitat loss was a factor in this species' historical decline (see Historical Range and Distribution), and the species is considered to be declining today in association with the continued reduction in habitat (Factors A and E). The species' severely restricted range, combined with its small population size, renders it particularly vulnerable to the threat of adverse natural (e.g., genetic, demographic, or environmental) and manmade (e.g., deforestation, habitat alteration, wildfire) events that destroy individuals and their habitat. Human activities that continue to curtail the species' habitat throughout its range exacerbate the genetic and demographic risks associated with small population sizes (Factor E). The species has experienced a population decline of between 10 and 19 percent in the past 10 years (see Population Estimates), and is predicted to continue declining commensurate with ongoing habitat loss and degradation. Current research indicates that narrow endemics, such as the white-browed tit-spinetail, are especially susceptible to climate fluctuations because of the synergistic effect these fluctuations have on declining populations that are also

experiencing range reductions due to human activities (Factor A).

Despite the species' endangered status in Peru and its occurrence within one protected area, human activities that degrade, alter, and destroy habitat are ongoing throughout the white-browed tit-spinetail's range, including within protected areas. Therefore, regulatory mechanisms are either inadequate or ineffective at curbing the threats to the white-browed tit-spinetail of habitat loss (Factor A) and corresponding population decline (Factor E).

Section 3 of the Act defines an "endangered species" as "any species which is in danger of extinction throughout all or a significant portion of its range" and a "threatened species" as "any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range." Based on the immediate and ongoing threats to the white-browed tit-spinetail throughout its entire range, as described above, we determine that the white-browed tit-spinetail is in danger of extinction throughout all of its range. Therefore, on the basis of the best available scientific and commercial information, we are proposing to list the white-browed tit-spinetail as an endangered species throughout all of its range.

Available Conservation Measures

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

Section 7(a) of the Act, as amended, and as implemented by regulations at 50 CFR part 402, requires Federal agencies to evaluate their actions within the United States or on the high seas with respect to any species that is proposed or listed as endangered or threatened, and with respect to its critical habitat, if any has been proposed or designated. However, given that the ash-breasted tit-tyrant, Junín grebe, Junín rail, Peruvian plantcutter, the royal cinclodes, and the white-browed tit-spinetail are not native to the United States, we are not proposing critical habitat for these species under section 4 of the Act.

Section 8(a) of the Act authorizes the provision of limited financial assistance for the development and management of programs that the Secretary of the Interior determines to be necessary or

useful for the conservation of endangered and threatened species in foreign countries. Sections 8(b) and 8(c) of the Act authorize the Secretary to encourage conservation programs for foreign endangered and threatened species and to provide assistance for such programs in the form of personnel and the training of personnel.

The Act and its implementing regulations set forth a series of general prohibitions and exceptions that apply to all endangered and threatened wildlife. As such, these prohibitions would be applicable to the ash-breasted tit-tyrant, Junín grebe, Junín rail, Peruvian plantcutter, the royal cinclodes, and the white-browed tit-spinetail. These prohibitions, under 50 CFR 17.21, make it illegal for any person subject to the jurisdiction of the United States to "take" (take includes to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct) any endangered wildlife species within the United States or upon the high seas; or to import or export; deliver, receive, carry, transport, or ship in interstate or foreign commerce in the course of commercial activity; or to sell or offer for sale in interstate or foreign commerce any endangered wildlife species. It is also illegal to possess, sell, deliver, carry, transport, or ship any such wildlife that has been taken in violation of the Act. Certain exceptions apply to agents of the Service and State conservation agencies.

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

Peer Review

In accordance with our joint policy with National Marine Fisheries Service, "Notice of Interagency Cooperative Policy for Peer Review in Endangered Species Act Activities," published in the **Federal Register** on July 1, 1994 (59 FR 34270), we will seek the expert opinions of at least three appropriate independent specialists regarding this proposed rule. The purpose of peer review is to ensure that our final determination is based on scientifically sound data, assumptions, and analyses. We will send copies of this proposed rule to the peer reviewers immediately

Species		Historic range	Vertebrate population where endangered or threatened	Status	When listed	Critical habitat	Special rules
Common name	Scientific name						
Rail, Junín	<i>Laterallus tuerosi</i>	Peru	Entire	E		NA	NA
*	*	*	*	*		*	*
Tit-spinetail, white-browed	<i>Leptasthenura xenothorax</i>	Peru	Entire	E		NA	NA
Tit-tyrant, ash-breasted	<i>Anairetes alpinus</i>	Bolivia, Peru	Entire	E		NA	NA
*	*	*	*	*		*	*

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Dated: December 16, 2009
Sam D. Hamilton,
 Director, U.S. Fish and Wildlife Service
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