LOSS OF TROPICAL FOREST AND AVIAN EXTINCTION AND DECLINE: PREDICTIONS AND RESEARCH NEEDS FOR CENTRAL AFRICA

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ABSTRACT

Human populations have increased dramatically over the past several decades, leading to pressure to develop or farm land and a concomitant acceleration of the rate of tropical deforestation. How native organisms respond to habitat loss and fragmentation has been the subject of intense study in temperate ecosystems and at various tropical sites in Latin America and southeastern Asia, but there has been little study of this phenomenon in Africa, a continent harboring a rich tropical avifauna. A survey of studies from Asia and the Neotropics yields relatively consistent patterns of which types of birds are most vulnerable to local extinction in the face of shrinking habitat. Large-bodied species (regardless of guild), flocking species, frugivores and insectivores, and species with low dispersal abilities and high habitat specialization are especially prone to sharp population declines and extirpation. On the basis of these generalizations from other regions, for which we present an example from the northern Neotropics (Palenque, Mexico), we generated predictions for which tropical birds of Central Africa would be most vulnerable and most resilient to forest fragmentation, including which are likely to colonize. Whether the predictions hold is a matter for further studies, ones that complement those in southeastern Asia and Latin America. Documenting the patterns in the Albertine Rift, for example, requires a two-pronged approach: (a) historical data needs to be collated in a manner comparable with (b) standardized and rigorous surveys to
map current distributions and population sizes of birds. As a first step, compiling a year-to-year sighting record of species can provide a statistically sound means of determining probability of extinction or measuring population reduction. In the interim, as a first approximation, urgent conservation planning can take established patterns of forest bird vulnerability into account to begin the process of mitigating against further declines and extinctions. We highlight these needs with a focus on developing conservation plans for Rwanda.

**Keywords:** deforestation, local extinction, Rwanda, vulnerability

The process of species loss from human-created habitat patches, such as forest fragments, has been of interest because understanding that process may allow predictions of the size of reserves necessary to maintain viable populations. — W. Douglas Robinson (2001)

**INTRODUCTION**

Tropical forests are being felled worldwide at a distressing rate, estimated recently at ~6,000,000 ha / year (Achard et al. 2002), and little of what remains is undisturbed (Willis et al. 2004; Lewis 2006). Tropical deforestation generally has occurred as a result of agricultural and pastoral expansion (Gaston et al. 2003; Carr 2004). Forest is being traded for field. Beyond the irrevocable loss of the world’s natural heritage, tropical deforestation may also exacerbate global climate change (Fearnside & Laurance 2004), which could lead indirectly to extinctions of endemics (Malcolm et al. 2006). But one does not have to invoke indirect threats to realize that tropical deforestation could itself lead directly to extinctions of a host of species dependent on forest habitats (Laurance & Bierregaard 1997; Pimm & Raven 2000). As Turner (1996) noted, “Enough studies have been conducted for us to conclude safely that the fragmentation of tropical rainforest is a major threat to local biodiversity.” And in most cases, direct loss of habitat rather than fragmentation per se is the focus of such studies (see Fahrig 2003).

Although the extent to which current deforestation rates will continue unabated or slow considerably is perhaps debatable (Wright & Muller-Landau 2006a,b; Brook et al. 2006; Laurance 2006), there is little doubt that direct loss of forest poses a problem for organisms that depend on it (Laurance & Bierregaard 1997). Furthermore, the extent to which forest is capable of regenerating fully will determine a large portion of its future biological value (Gardner et al. 2007). Natural regeneration of tropical forest in abandoned agricultural fields is likely to be slow (Chapman & Chapman 1999; Myster
2004), meaning dependent species with short generation times will suffer the effects of deforestation long after a cleared plot ceases to be worked. Only active restoration, including planting nurse trees and spreading fertilizer, may speed the process (Holl & Kappelle 1999; Aide 2000; Myster 2004). In the meantime, an understanding of the long-term effects of deforestation is needed if we hope to preserve natural habitats and their species into the distant future.

**DEFORESTATION AND FOREST BIRDS**

It is axiomatic that reductions in forested habitats will lead to reductions in population densities and species richness of forest birds (e.g., Brook et al. 2003). Richness falls as habitat area shrinks, as predicted by species–area relationships (Pimm et al. 1995; Pimm and Raven 2000), although in some cases the decline is shallower (e.g., Brooks et al. 2002) or steeper (e.g., Brooks et al. 1999) than the predicted 25–50% species loss with 90% habitat loss (Sodhi et al. 2004). Rates at which avian species richness declines have been determined through experimentation (e.g., Ferraz et al. 2003, 2007). The general pattern is one of an initial spike in richness and abundance as species and individuals pack into a small area, followed by a sharp decline until reaching a new species richness at a total lower than the original, pre-disturbance level (Bierregaard et al. 1992). Still, species loss can occur rapidly (Lees & Peres 2006).

It may seem an odd question, but why is species richness lower? After all, in principle a smaller area could retain the same number of species; it would have only fewer individuals of each species than a larger area. Lower species richness could occur through stochastic processes, wherein there is no predictability of which species will persist and which will disappear. Against this null model stands a host of empirical data showing clearly that local extinction is non-random (Purvis et al. 2000; Bennett et al. 2005; Gray et al. 2007): particular species or groups are more vulnerable, others are more resilient. Detecting patterns can be challenging, especially when comparing among avifaunas, and more data are needed from geographically diverse sites. For example, from 1970–2000, over half (54.3%) of the research on biotic effects of deforestation was undertaken in the Neotropics, compared with only 16.5% in the Afrotropics (Rudel et al. 2000), and a disproportionate amount of the Neotropical research was a single long-term study, the Biological Dynamics of Forest Fragments Project at Manaus, Brazil (Turner 1996). Similarly, Sodhi et al. (2004) noted that 75% of studies reporting avian extirpations were conducted in the Neotropics.
Nonetheless, particular correlates of vulnerability have emerged often enough to have reached a kind of consensus. Species with a large body size and (typically correlated) low annual fecundity are particularly prone to extinction threat (Bennett et al. 2005). So too are species with low mobility, an indication of limited dispersal ability, and low population density (Turner 1996; Reed 1999). Regarding the latter, however, it is possible that high interannual variation in population size is a better predictor of extinction risk than is rarity per se (Karr 1982). Another key correlate of vulnerability is the degree of habitat specialization (Sodhi et al. 2004). A species restricted to a particular habitat that requires a particular resource or otherwise has specific social needs—such as leks or mixed-species flocks—is at higher risk than a species with a wider habitat range or more plastic behavior (Reed 1999).

This suite of indicators for extinction proneness has lead to various hypotheses about processes underlying the patterns. Unlike the patterns, however, processes remain murky, with more research needed to understand what makes a particular species vulnerable or resilient (Stratford & Robinson 2005). Each of the “over a dozen” (Şekercioğlu et al. 2002) hypotheses put forth in explanation of vulnerability falls into one of five broad groups (Ford et al. 2001; Sodhi et al. 2004; Bennett et al. 2005), biogeographical, morphological, behavioral, evolutionary, and phylogenetic, although some hypotheses span several groups.

Biogeography plays a role in determining vulnerability, in that a species at the edge of its geographic range is likely to be at its physiological limits (e.g., Root 1988) and occur at a lower density (e.g., Brown et al. 1995). Thus, locales at the historical edges of the distribution of tropical forests are likely to harbor populations that are less resilient to deforestation or fragmentation. (We provide an example of such a forest below.) Rarity is also associated with vulnerability, although not always in a logically expected way: rare species are not necessarily the most extinction prone (Karr 1982). Moreover, rarity itself correlates strongly and positively with body size and specialization and strongly and negatively with the extent of geographic range (Sodhi et al. 2004).

As noted above, morphology associates with vulnerability in terms of body size. Larger species are less resilient (Bennett et al. 2005; Gray et al. 2007), a pattern generally attributed to the inverse correlation between body size and population size (Pimm et al. 1988). In addition to overall size, a species specialized to feed on certain foods or in a certain manner is likely to have evolved a specialized morphology (or behavior), making it less able to tolerate alterations to its habitat or resource base. Indeed, most birds with specialized modes of feeding decline in response to habitat disturbance
(Gray et al. 2007). A key example of specialization is the obligate army ant followers in the Neotropics that disappear quickly from fragments (Bierregaard et al. 1992). Beyond foraging modes, other behaviors may increase vulnerability, such as flocking. In general, species that lek or join mixed-species flocks are less resilient, with the latter more likely to disappear from small forest patches (Sodhi et al. 2004). The degree of specialization may even manifest itself in a narrower physiological tolerance of tropical forest birds or in their inability to tolerate change in the light environment (Stratford & Robinson 2005). In essence, even minor changes in microclimate (e.g., ambient temperature or light penetration) may be enough to drive local extinctions.

Other behavioral or evolutionary (life history) traits linked to vulnerability are dispersal, fecundity, survival, and generation time. Poor dispersers are less resilient, and tropical birds—at least in the Americas—are thought to have poor dispersal abilities relative to their temperate counterparts (Şekercioğlu et al. 2002; Sodhi et al. 2004; Stratford & Robinson 2005). Despite their poor dispersal abilities, however, tropical birds tend to have larger home ranges and, consequently, lower population densities than temperate birds (Stratford & Robinson 2005). In addition, species with low annual fecundity, low survivorship, and short generation times are more prone to extinction (Bennett et al. 2005). Compared to phylogenetically similar north temperate birds, Neotropical birds have low annual fecundity (Stutchbury & Morton 2001) and clutch size (Patten 2008), even if survival rates tend to be higher (Stutchbury & Morton 2001).

Phylogenetic inertia and constraint, typically studied as taxonomic effects (Sodhi et al. 2004), make some groups more vulnerable than others (Bennett et al. 2005). At the level of avian family, examples of phylogenetic constraints abound, ranging from large body size (e.g., Phasianidae) to low annual fecundity (e.g., Diomedeidae) to presumed poor dispersal ability because of weak flight (e.g., Tinamidae). Nevertheless, such constraints specific to tropical birds require more study.

Bear in mind that most hypotheses deal with direct effects and are not mutually exclusive. Most extinctions are probably the result of multiple factors. Beyond direct effects, higher-order and synergistic effects may account for vulnerability (Turner 1996). A key example is loss of obligate ant followers from a fragment in the wake of loss of army ants (e.g., Bierregaard et al. 1992); i.e., loss of birds is a second order effect dependent on loss of ants. Competition with invasive species can lead to declines (Sodhi et al. 2004). “Mesopredator release”—the proliferation of mid-sized mammalian predators after extirpation of large predators that kept their populations in check
(see Crooks & Soulé 1999)—can lead to trophic cascades that result in increased predation on bird nests. Complex synergies can also develop, as when fruiting trees rely on avian frugivores for dispersal and these frugivores rely on the trees for food. Declines in either player leads to a vicious cycle wherein mutual extinction is assured.

DEFORESTATION IN THE AFROTROPICS

... studies on tropical deforestation ... are unevenly distributed geographically. Central America and Southeast Asia receive disproportionate amounts of attention, whereas Central Africa receives little attention. — Thomas K. Rudel and colleagues (2000)

The steadily growing volume of conservation research seen in other tropical regions has yet to reach Africa: by his rough tally, Pimm (2007) reported that a mere 6% of 3300 recent papers in the journal Conservation Biology were about Africa. We noted a similar paucity of African research at the 2007 conference of the Association of Tropical Biology and Conservation, where only 3 of over 700 presentations dealt with the “Dark Continent.” This situation is unfortunate given the rapid loss of forest in the Afrotropics (Barnes 1990; Achar et al. 2002; Lung & Schaab 2006; Laporte et al. 2007). Africa as a whole is losing more than 4,000,000 ha / yr, the vast majority of it in a belt on either side of the equator (FAO 2006). Within this belt, Central and East Africa, including the Congo Basin, Albertine Rift, and Eastern Arc Mountains, are losing vast amounts of forest annually (Table 3.1). What forest remains receives little protection. As an example, from the Atlantic Ocean to the eastern edge of the Democratic Republic of Congo, only 12% of forest is protected but some 30% is under logging concession (Laporte et al. 2007).

Table 3.1: Changes in the extent of forest cover in central and eastern Africa from 1990 to 2005

<table>
<thead>
<tr>
<th>Country</th>
<th>1990</th>
<th>2000</th>
<th>% change</th>
<th>2005</th>
<th>% change</th>
</tr>
</thead>
<tbody>
<tr>
<td>DRC</td>
<td>140 531</td>
<td>135 207</td>
<td>-0.4</td>
<td>133 610</td>
<td>-0.2</td>
</tr>
<tr>
<td>Uganda</td>
<td>4 924</td>
<td>4 059</td>
<td>-1.9</td>
<td>3 627</td>
<td>-2.2</td>
</tr>
<tr>
<td>Rwanda</td>
<td>318</td>
<td>344</td>
<td>0.8</td>
<td>480</td>
<td>6.9</td>
</tr>
<tr>
<td>Burundi</td>
<td>289</td>
<td>198</td>
<td>-3.7</td>
<td>152</td>
<td>-5.2</td>
</tr>
<tr>
<td>Kenya</td>
<td>3 708</td>
<td>3 582</td>
<td>-0.3</td>
<td>3 522</td>
<td>-0.3</td>
</tr>
<tr>
<td>Tanzania</td>
<td>41 441</td>
<td>37 318</td>
<td>-1.0</td>
<td>35 257</td>
<td>-1.1</td>
</tr>
<tr>
<td>Africa</td>
<td>699 361</td>
<td>655 613</td>
<td>-0.64</td>
<td>635 412</td>
<td>-0.62</td>
</tr>
</tbody>
</table>

Source: (data from FAO 2005). Tabled values for forest cover are in 1000s of hectares. DRC = Democratic Republic of Congo.
Moreover, nominal protection has meant little in recent history. Rwanda, for instance, has lost 65% of its gazetted parklands within the past 40 years (Rutagarama & Martin 2006). In particular, four forested reserves took extensive hits by military actions or stroke of governmental pen. Parc National des Volcans, Parc National de Nyungwe (formerly a Forest Reserve), and Gishwati and Mukura Forest Reserves were shrunk by the Rwandan Civil War of the mid-1990s (Kanyamibwa 1998; Plumptre et al. 2001). Rwanda’s western neighbor, the Democratic Republic of Congo (DRC), was also affected by the resettling of Rwandan refugees near Virunga National Park, which resulted in large areas of forest cleared for fuelwood (Plumptre et al. 2001). Parc des Volcans likewise experienced direct impacts when harboring conflicting forces of the Rwanda Patriotic Front, interahamwe, and Rwandan military, as well as Congolese guerillas from the civil conflict raging in the DRC. Fuel-wood deforestation and other impacts affected wildlife in Nyungwe, home to 26 birds endemic to the Albertine Rift (Plumptre et al. 2007). A heavy toll was exacted as a direct result of the conflict and its aftermath from the flood of poachers and refugees (from Rwanda’s and Burundi’s civil wars) that poured into the reserve, taking free reign once reserve staff were forced to vacant. By the late 1990s, smallholder agricultural plots reportedly had fragmented the southeastern part of Nyungwe (Kanyamibwa 1998). Gishwati and Mukura Forest Reserves experienced high rates of deforestation from wartime shelling and the need to provide land for displaced populations (Kanyamibwa 1998; Plumptre et al. 2001).

Beyond conservation issues, Rwanda and many other Central African countries have more to deal with than just physical aftermaths of wars. In many cases, ethnic tensions, amplified by colonial intervention and unremitting power struggles, gave rise to socioeconomic disparities that catalyzed war and that persist today (Uvin 1999). Rampant poverty, experienced by the majority of Rwandans, perpetuates the need for farmland and fuel-wood. Furthermore, population growth rates—such as >3.0% in Rwanda (Rutagarama & Martin 2006)—continue to pressure natural resources. Because of these myriad issues, as with Asia and the Neotropics, deforestation rates are unlikely to ease in the foreseeable future. Indeed, the FAO’s (2006) optimistic remark that there are “signs that the net loss in Africa is decreasing” may be the result of having less and less intact forest (currently) accessible to teeming human populations. Besides, if rates are leveling off after spikes in the 1980s and 1990s, who is to say that other spikes will not occur one or two decades hence?
DETERMINING EFFECTS OF DEFORESTATION ON BIRDS

...data for Africa came from only three studies... [that] produced highly variable results and made generalizations on the basis of these data inappropriate. — Michael A. Gray and colleagues (2007)

The paucity of conservation research in Africa extends, of course, to the continent’s unique avifauna. Studies on the effects of deforestation on birds take one of two approaches, dubbed the “historical method” and “fragmentation method” by Sodhi et al. (2004). Of these approaches, most recent studies have used the fragmentation method, likely because it is more tractable: it has no requirement for a set of long-term data. In essence, this method compares avifauna in remnant forest patches of varying sizes to those in a large reserve to determine which species are resilient and which are vulnerable. This method has been used with some success in eastern Africa (e.g., Newmark 1991; Lens et al. 2002; Githuru et al. 2007), including between disturbed and undisturbed sites (e.g., Dale & Slembe 2005). Much more “fragmentation method” work is needed in Uganda, Rwanda, Burundi, and the Democratic Republic of Congo.

The historical approach compares data from past surveys to those from present surveys (e.g., Sodhi et al. 2005; Sigel et al. 2006). This method may therefore seem to require extensive reference data, such as from a thorough inventory or census for decades past. Yet this apparent limitation need not be one, as our recent work in a rainforest of southern Mexico attests. Our sporadic visits to the Mayan ruins of Palenque National Park, Chiapas, Mexico, over nearly 15 years led us to ponder the extent to which land use changes since the 1970s had affected the bird community.

Palenque sits at the northern tip of the Selva Lacandona, a reach of lowland rainforest that extended formerly in an unbroken band to well into the northern district (the Petén) of neighboring Guatemala but exists now in scattered patches of varying sizes, with Palenque occupying a forested island (Estrada et al. 2002). Despite a lack of formal surveys, we were able to take advantage of both historical data (i.e., museum specimens, chiefly from 1900, 1939, 1946, and 1949) and extensive recent data from visiting birders. After compiling a pre-1970 baseline list of species, we compiled annual lists for species presence/absence. We analyzed these binary data using techniques designed for inferring extinction probability from a sighting record (Solow 2005; McNemery et al. 2006) and using binomial tests to infer declines or increases (Patten et al. ms.). Without recourse to survey data we nevertheless were able to deduce patterns
of species loss and gain (Figure 3.1) and to determine that forest species have suffered while open-country species have benefitted (Figure 3.2).

![Diagram showing species changes](image)

**Figure 3.1: Dynamics of the avifauna at Palenque, Chiapas, Mexico, 1900-2007, expressed as a percent of species whose status changed.** A species was considered extirpated or to have colonized on the basis of a Weibull statistic across a presence/absence sighting record (Solow 2005). A species was considered to have increased or decreased on the basis of a binomial test comparing encounter rate from 1970–1989 to encounter rate from 1990–2007. Species designated as “unchanged” lacked a significant trend. Changes were attributed to the extensive loss of rainforest in the region (see Figure 3.2).

We were further able to determine that gamebirds, mid-sized frugivores, and small insectivores have suffered disproportionately (Patten *et al.* ms.). Thus, foraging guild plays a key role in determining vulnerability. Similar patterns have been reported elsewhere in the Neotropics (e.g., Robinson 2001, Sigel *et al.* 2006), and in a meta-analysis of avian responses to tropical forest disturbance, Gray *et al.* (2007) found that, across studies in Asia and the Neotropics, granivores tended to increase but frugivores and insectivores decreased. But even granivores may suffer when comparing species richness in native vegetation and in smallholding agricultural areas (Sinclair *et al.* 2007).
Taking a broad view, across studies worldwide, the real issue is one of predictability. If well-supported hypotheses developed from numerous studies in the Neotropics or Southeast Asia (Table 3.2) have real value, then they can generate predictions of which species in the Afrotropics—where comparatively little work has been conducted—are likely to be vulnerable and which are likely to be resilient to local extinction. This exercise is particularly important in the Albertine Rift, home to 41 species of endemic bird and 12 additional species considered globally threatened (Plumptre et al. 2007). We offer such predictions herein as a first step in this direction, which has the dual goal of testing general hypotheses of species vulnerability and providing sorely needed baseline data for Central Africa. Indeed, given enough information on basic natural history, vulnerability can be predicted for a given species on the basis of diet, foraging habitat, and nesting habitat across various habitats and management regimes in a manner rigorous enough to infer relative extinction risk (e.g., Butler et al. 2007).
Table 3.2: Traits associated with avian vulnerability to extinction.

<table>
<thead>
<tr>
<th>Trait</th>
<th>Low</th>
<th>High</th>
<th>Predicted Relationship</th>
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<tbody>
<tr>
<td><strong>Vulnerability</strong></td>
<td></td>
<td></td>
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<tr>
<td><strong>dispersal ability</strong></td>
<td>low</td>
<td>high</td>
<td>negative exponential</td>
</tr>
<tr>
<td><strong>diet</strong></td>
<td></td>
<td></td>
<td>linear</td>
</tr>
<tr>
<td>[yes] granivore?</td>
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<td></td>
<td></td>
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<tr>
<td>[no] insectivore?</td>
<td></td>
<td></td>
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<tr>
<td>[no] frugivore?</td>
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<tr>
<td><strong>microhabitat</strong></td>
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<td>negative exponential</td>
</tr>
<tr>
<td>[low] light sensitivity</td>
<td></td>
<td>high</td>
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<tr>
<td>[high] temperature range</td>
<td></td>
<td>low</td>
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<tr>
<td>[low] “edge sensitivity”</td>
<td></td>
<td>high</td>
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<tr>
<td><strong>nesting requirements</strong></td>
<td></td>
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<td>linear</td>
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<tr>
<td>[cavity/closed] nest type</td>
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<td>[cup/open]</td>
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<td><strong>body size</strong></td>
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<td>log-linear</td>
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<td>small</td>
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<tr>
<td>high</td>
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<td>linear</td>
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<tr>
<td><strong>annual fecundity</strong></td>
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<td>small</td>
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<td>negative exponential</td>
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<td><strong>home range size</strong></td>
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<tr>
<td>flexible</td>
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<tr>
<td><strong>flocking behavior</strong></td>
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<tr>
<td>[no mixed-species flocking?]</td>
<td></td>
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<td></td>
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<tr>
<td>[no lekking?]</td>
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Note: In the context of this study, we are concerned with extinction from forest patches arising from broad-scale deforestation. Traits are summarized from a variety of sources, most notably Sodhi et al. (2004), Bennett et al. (2005), Stratford & Robinson (2005), Gray et al. (2007), and Lindell et al. (2007). The “predicted relationship” is based on our own supposition and the kind of data (e.g., an edge effect ought to manifest itself relative to the square of the distance).

Several families of birds are endemic to Africa, one of which—Musophagidae (turacos)—is essentially confined to forest. Other “major families of forest birds“ (per
Primack & Corlett 2005) in tropical Africa include the Cuculidae (cuckoos), Alcedinidae (kingfishers), Bucerotidae (hornbills), Pycnonotidae (bulbuls and greenbuls), Laniidae (shrikes), Sylviidae (Old World warblers sensu lato), Muscicapidae (Old World flycatchers), Nectariniidae (sunbirds), and Ploceidae (weavers). To this list we would add the Turdidae (thrushes). Compared to northern North America, Europe, and Australia, little is known of the natural history of these birds, yet enough can be gleaned from basic references (e.g., Zimmerman et al. 1999; Stevenson & Fanshawe 2002) to make educated guesses. Our three categories below—highly vulnerable, moderately vulnerable, and resilient—roughly correspond to, respectively, Lees & Peres’s (2006) “sensitivity” 1 + 2 (birds restricted to forest), 3 (forest birds tolerate of second growth), and 4 (non-forest, scrub, and open country species).

Using our framework (Table 3.2), we predict that the following Central African families, genera, or species will be highly vulnerable to local extinction as a result of forest loss: Francolinus (francolins), Treron pigeons, Aplopelia larvata (Lemon Dove), Musophagidae, Phoeniculidae (wood-hoopoes), Bycanistes hornbills, Eurylaimidae (broadbills), Pittidae (pittas), Campephagidae (cuckoo-shrikes), Phyllastrephus greenbuls, Sheppardia (akalats), Alethe, Zoothera ground-thrushes, Bathmocercus rufus (Black-faced Rufous Warbler), Graueria vittata (Grauer’s Warbler), Platysteira (wattle-eyes), Batis diops (Rwenzori Batis), Illadopsis and Kakamega (illadopsis), Neocosyphus (ant-thrushes), Spermophaga ruficailla (Red-headed Bluebill), Nigrita (negrofinches), and Cryptospiza (crimsonwings).

We predict that the following will be moderately vulnerable: Turtur wood-doves, Chrysococcyx cuckoos, Trogonidae (trogons), Andropadus greenbuls, Cosypha robin-chats (except C. hueglini), Sylvietta (crombecs), Phylloscopus woodland-warblers, Apalis (except A. flavida), Camaroptera, Eminia lepida (Grey-capped Warbler), Trochocercus (crested-flycatchers), Zosteropidae (white-eyes), Oriolidae (Old World orioles), and Cinnynicinclus starlings.

Lastly, we predict that the following families, genera, or species will be resilient to local extinction as a result of forest loss or may colonize in the wake of such loss: Columba pigeons, Streptopelia doves, Centropus (coucals), Coliidae (mousebirds), Tockus hornbills, Lybiidae (African barbets), Pycnonotus barbatus (Common Bulbul), Cosypha hueglini (White-browed Robin-Chat), Cercotrichas (scrub-robins), Turdus thrushes, Cisticola, Prinia, Apalis flavida (Yellow-breasted Apalis), Sylvia warblers, Muscicapidae flycatchers, Turdoides babblers, Malacoontidae (bush-shrikes), Prionopidae (helmet-
shrikes), Nectariniidae (sunbirds), Dicuridae (drongos), Lamprotornis starlings, Ploceus (weavers), Estrilda (waxbills), Lonchura (mannikins), and Serinus (canaries).

These lists are hardly comprehensive, and a few species could be argued to one category or another. No matter. Our point is that existing research can be used to generate testable hypotheses about which species are likely to have remained in or disappeared from remnant patches of tropical forest. Moreover, on-the-ground studies of species presence/absence in forest patches can work jointly with biogeographic analyses of avifaunal patterns (e.g., FjeldsD 2007) to set conservation priorities in Central Africa. Given the increased amount of attention devoted to dynamics of persistence and extinction in forest patches, the opportunity to test ecologically, demographically, or phylogenetically derived hypotheses on an independent data set—the birds of the Afrotropics—in a potential boon to this field of research.

**RESEARCH NEEDS FOR CENTRAL AFRICAN BIRDS, WITH A FOCUS ON RWANDA**

Relative to birds in other parts of the world, particularly in the north temperate zone, little is known about the birds of Central Africa. In many cases even basic status and distribution is known only broadly—there are few specifics and many open questions. This deficiency is highlighted by the only field guide that covers Rwanda and Burundi (Stevenson & Fanshawe 2002), which displays numerous maps showing ranges, for example, that extend to a perfect fit of the southern border of Uganda but do not cross into Rwanda. Still, we must go beyond compiling species lists of birds to determine their actual distribution, relative abundance, seasonal occurrence, and extent of elevation movements within Rwanda and elsewhere in Central Africa. As with status and distribution, we lack information on habitat requirements and demography for these same species. We have little understanding of their ecologies, whether microclimate, dispersal, diet, phenology, or nesting requirements.

Therefore, to begin plugging gaps, routine surveys need to be conducted countrywide for a minimum of two years to determine status and distribution. These data should be added to a comprehensive database of all avian research conducted in Rwanda, a database that should include museum specimens, research expedition notes, field notes from bird watchers, and published literature (including “grey literature” published by the Rwandan government). Requests can be made to tour companies and their field trip leaders to supply species lists for inclusion in the database.
Once data have been collected, a presence/absence analysis (as discussed above) can be run to determine historical and current status, which will aid the Rwandan government in conservation decision making. Rwanda is fortunate to have a contingent of young scientists, scholars, professionals, non-profit agencies, and governmental officials dedicated to conserving the country’s biodiversity and natural heritage. This dedication was wholeheartedly expressed and confirmed at the first Rwanda International Research Conference on Biodiversity and Sustainable Management of Natural Resources in July 2007, which was sponsored by the Rwandan government, and at which President Paul Kagame delivered an optimistic speech about biodiversity preservation. The continued hard work of these Rwandan citizens and foreign advocates, along with fulfillment of the promises made by the president and his ministers, Rwanda will not only have a catalog of its biodiversity, including birds, but it will also have the management tools necessary to comprehend its value and conserve it for future generations.

Beyond basic research needs, the future should hold reforesting Rwanda’s devastated landscape with native trees, including the establishment of wildlife corridors, a notion long supported theoretically but only recently receiving solid empirical support (Levey et al. 2005). Dealing with the burgeoning population is a key to success of this or any other conservation initiative. More people means more demand of dwindling natural resources, and high population density puts additional pressure on small reserves (Harcourt et al. 2001). Worse, protected areas are tending to become more isolated with increased clearing of habitat (DeFries et al. 2005). The problem of increased isolation is especially vexing if intervening matrix is unsuitable for dispersal. Assuming the critical issue of human population control can be tackled quickly and effectively, there is hope for the future of the rich biodiversity of Rwanda and other countries in Central Africa.

ACKNOWLEDGMENTS

We thank Katie Fawcett, Research Director of Karisoke Research Center, Dian Fossey Gorilla Fund International, for funding our visit to Rwanda and for her continued advice; FidPle Ruzigandekwe, Executive Director, Rwanda Wildlife Agency, Rwanda Office of Tourism and National Parks (ORTPN) for supporting development of an ornithological research program at Karisoke Research Center (and eventually for Rwanda as a whole); and the promising young scholar Claudien Nsabagasani for sharing his research as well as his time to bring us closer to the birds of Rwanda.
REFERENCES


CHAPTER 3


