Assessing conservation status of resident and migrant birds on Hispaniola with constant-effort mist-netting

John D Lloyd, Chris C Rimmer, Kent P McFarland

We analyzed temporal trends in mist-net capture rates of resident (n = 8) and overwintering Nearctic-Neotropical migrant (n = 3) bird species at two sites in montane broadleaf forest of the Sierra de Bahoruco, Dominican Republic, with the goal of providing quantitative information on population trends that could inform conservation assessments. We conducted sampling at least once annually during the winter months of January – March from 1997 – 2010. We found evidence of steep declines in capture rates for three resident species, including one species endemic to Hispaniola. Capture rate of Rufous-throated Solitaire (*Myadestes genibarbis*) declined by 3.9% per year (95% CL = 0%, 7.3%), Green-tailed Ground-Tanager (*Microlegia palustris*) by 6.8% (95% CL = 3.9%, 8.8%), and Greater Antillean Bullfinch (*Loxigilla violacea*) by 4.9% (95% CL = 0.9%, 9.2%). Two rare and threatened endemics, Hispaniolan Highland-Tanager (*Xenolegia montana*) and Western Chat-Tanager (*Calyptophilus tertius*), showed statistically significant declines, but we have low confidence in these findings because trends were driven by exceptionally high capture rates in 1997 and varied between sites. Analyses that excluded data from 1997 revealed no trend in capture rate over the course of the study. We found no evidence of temporal trends in capture rates for any other residents or Nearctic-Neotropical migrants. We do not know the causes of the observed declines, nor can we conclude that these declines are not a purely local phenomenon. However, our findings, along with other recent reports of declines in these same species, suggest that a closer examination of their conservation status is warranted. Given the difficulty in obtaining spatially extensive, long-term estimates of population change for Hispaniolan birds, we suggest focusing on other metrics of vulnerability that are more easily quantified yet remain poorly described, such as extent of occurrence.

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2 **effort mist-netting**

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9

10 **Abstract**

11 We analyzed temporal trends in mist-net capture rates of resident (n = 8) and overwintering

12 Nearctic-Neotropical migrant (n = 3) bird species at two sites in montane broadleaf forest of the

13 Sierra de Bahoruco, Dominican Republic, with the goal of providing quantitative information on

14 population trends that could inform conservation assessments. We conducted sampling at least

15 once annually during the winter months of January – March from 1997 – 2010. We found

16 evidence of steep declines in capture rates for three resident species, including one species

17 endemic to Hispaniola. Capture rate of Rufous-throated Solitaire (*Myadestes genibarbis*)

18 declined by 3.9% per year (95% CL = 0%, 7.3%), Green-tailed Ground-Tanager (*Microlegia*

19 *palustris*) by 6.8% (95% CL = 3.9%, 8.8%), and Greater Antillean Bullfinch (*Loxigilla violacea*)

20 by 4.9% (95% CL = 0.9%, 9.2%). Two rare and threatened endemics, Hispaniolan Highland-

21 Tanager (*Xenolegia montana*) and Western Chat-Tanager (*Calyptophilus tertius*), showed

22 statistically significant declines, but we have low confidence in these findings because trends

23 were driven by exceptionally high capture rates in 1997 and varied between sites. Analyses that

24 excluded data from 1997 revealed no trend in capture rate over the course of the study. We found

25 no evidence of temporal trends in capture rates for any other residents or Nearctic-Neotropical

26 migrants. We do not know the causes of the observed declines, nor can we conclude that these

27 declines are not a purely local phenomenon. However, our findings, along with other recent

28 reports of declines in these same species, suggest that a closer examination of their conservation

29 status is warranted. Given the difficulty in obtaining spatially extensive, long-term estimates of

30 population change for Hispaniolan birds, we suggest focusing on other metrics of vulnerability

31 that are more easily quantified yet remain poorly described, such as extent of occurrence.

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INTRODUCTION

Hispaniola supports a notably diverse avifauna, including at least 31 endemic species (Latta et

al., 2006), several of which appear to be the only extant members of ancient, family-level clades

(Barker et al., 2013; Barker et al., 2015). Many of these taxa are of substantial conservation

concern given extensive habitat loss caused by historical and ongoing deforestation in both Haiti

and the Dominican Republic (Stattersfield et al., 1998; Latta, 2005). Few, if any, of the endemic

birds of Hispaniola have been well studied, however, and assessments of their conservation

status are often qualitative, subjective, and based largely on expert opinion (Latta and Fernandez,

2002). Decisions about investments in conservation are often guided by population status

(Possingham et al., 2002; Rodrigues et al., 2006), and thus well-informed status assessments are

critically important for the effective allocation of limited funding for conservation.

Here, we seek to improve current understanding of the conservation status of the unique

and threatened assemblage of birds in montane cloud forest in Sierra de Bahoruco, Dominican

Republic. These forests are a hotspot of endemism on the island (Latta, 2005), support several

globally threatened resident bird species, and constitute a principal wintering area for the

globally Vulnerable (BirdLife International, 2012) Bicknell's Thrush (*Catharus bicknelli*), a

Nearctic-Neotropical migrant. Montane cloud forests also face substantial and ongoing threats

from deforestation for agricultural production and expansion of human settlements, even in

ostensibly protected areas such as Sierra de Bahoruco National Park (BirdLife International,

2015). We used data collecting during 13 years of constant effort mist-netting at two different

sites to estimate temporal trends in capture rate, which we use as an index of change in

population size and as a means to draw inference about conservation status. In other tropical

56   systems, long-term, constant-effort mist netting has proven a useful tool for identifying

57   population declines in bird assemblages that are otherwise difficult to monitor (e.g., Faaborg et

58   al., 2013)

59

60   **METHODS**

61   From 1995-2010 we operated a standardized array of 30-35 mist nets (12-m x 2.6-m, 36-mm

62   mesh) at two remote sites in Sierra de Bahoruco, southwestern Dominican Republic. The sites,

63   Pueblo Viejo (hereafter "PUVI"; 18.2090°N, -71.5080°W) and Palo de Agua (hereafter "PALO";

64   18.2047°N, -71.5321°W), consist of montane cloud forest at 1,775-1,850 m elevation in Sierra de

65   Bahoruco National Park and are separated by 2.6 km of contiguous forest. Both sites are

66   characterized by a dense understory composed largely of thick woody tangles, complete

67   broadleaf canopy cover with trees reaching heights between 15 and 20 m, and an abundance of

68   lianas and epiphytes (Veloz, 2007).   Human disturbance at both sites was limited primarily to

69   trimming vegetation to prevent overgrowth of our net lanes.

70   We visited PUVI at least once annually between November and May, except in 1999

71   when we did not visit either site. We made two visits in 1997 (March, November), 1998 (March,

72   November), 2002 (February, May), 2003 (February, May), and 2010 (March, November). Our

73   one visit to PUVI in 1996 occurred in early December. To minimize the potentially confounding

74   effects of seasonal variation in abundance and bird behavior that may affect capture rate, this

75   analysis does not include data collected during the May, November, and December visits.

76   Resident birds have commenced breeding by May, and so availability for capture may be

77   different during this period. Transient hatch-year birds, which likely have a very different

78   probability of capture, also begin appearing in large numbers in May. Migrant birds have

79   departed for their breeding grounds by May, but are still arriving at our sites during November

80   and, to a lesser extent, early December.

81   We also visited PALO at least once annually during the same period that we visited

82   PUVI, except for 1996 and 1999-2001. As with PUVI, we excluded data collected during the

83   three November visits (1997, 1998, and 2010). We did not visit PALO in May. At both sites in

84   1995, we only banded Nearctic-Neotropical migrants, and so we excluded data from that year

85   from this analysis. The final, censored data set for this analysis thus includes captures made

86   from 1997 – 2010 (with no data collected in 1999) on dates ranging from 24 January to 21

87   March. We believe that this date range reflects a period of relative stability at our sites, after

88   migrant species have arrived, settled, and established winter territories but before resident

89   species have commenced breeding. As such, we also believe that capture rates during this period

90   are comparable among years because availability for and probability of capture should be

91   relatively constant among years. Furthermore, limiting analysis to this time period reduces the

92   possibility of confounding seasonal effects with yearly trends in capture rate because in most

93   years we did not attempt early-season (November – December) or late-season (May) captures.

94   At each site, we established permanent net locations along three parallel foot trails 100-

95   150 m apart. The area bounded by the foot trails was ~25 ha at PUVI and ~15 ha at PALO. We

96   regularly used 30 net locations at PUVI and 35 at PALO, although at both sites we also

97   occasionally recorded captures at locations where nets were deployed opportunistically for

98   related research projects (e.g., Townsend et al., 2010). Nets were typically operated for 3 days at

99   each site, beginning in late afternoon of day 1, from dawn to dusk on days 2 and 3, and until

100   mid-morning on day 4. Nets were checked hourly and closed under adverse weather conditions.

101   We recorded daily opening and closing times of each net. Both sites were netted in succession

102   each year, with set-up at the second site occurring on the day that nets were removed from the

103   first site.

104   We placed U.S. Fish and Wildlife Service aluminum leg bands on all Nearctic-

105   Neotropical migrant species and custom-made, uniquely numbered leg bands (Gey Band and Tag

106   Company) on all Hispaniolan resident species. We did not have leg bands that fit Hispaniolan

107   Emeralds (*Chlorostilbon swainsonii*), so we temporarily marked individuals of this species by

108   clipping the distal 2-3 mm of up to three rectrices in a unique pattern. Although we believe that

109   doing so allowed us to identify within-session recaptures, we had no way to verify the validity of

110   this approach (e.g., that marks were unambiguously recognized by all observers) and so we did

111   not analyze capture rates for this species. We aged and sexed all North American species using

112   standard criteria according to Pyle (1997) and all resident species using criteria available in field

113   guides (Latta et al., 2006) or based on our own accumulated field knowledge. However, we

114   could only reliably age and sex a handful of species, so we pooled capture rates for all ages and

115   all sexes in our analyses.

116   We analyzed trends in capture rate for 6 endemic species that we believed were

117   adequately sampled by our methods (English common names follow Latta et al. [2006]):

118   Narrow-billed Tody (*Todus angustirostris*), Green-tailed Ground Tanager (*Microlegia palustris*),

119   Hispaniolan Highland-Tanager (*Xenoligea montana*), Black-crowned Palm-Tanager

120   (*Phaenicophilus palmarum*), Western Chat-Tanager (*Calyptophilus tertius*), and Hispaniolan

121   Spindalis (*Spindalis dominicensis*). Quantitative data on population trends are lacking for all of

122   these species, but two are suspected of being at risk of extinction: Hispaniolan Highland-

123   Tanager is recognized as Vulnerable on the IUCN Red List (BirdLife International 2012) and

124   Endangered by Latta et al. (2006), and Western Chat-Tanager is considered Critically

125 Endangered by Latta et al. (2006). BirdLife International does not recognize the taxonomic

126 separation of Eastern Chat-Tanager (*C. frugivorous*) and Western Chat-Tanager, and instead

127 considers the entire species group Vulnerable (BirdLife International 2012). We excluded

128 Hispaniolan Emerald from analysis because we could not mark them permanently, and we

129 excluded Hispaniolan Pewee because it spends much of its time in the canopy and was probably

130 poorly sampled by our mist nets. We decided that sample sizes of all of the remaining endemics

131 were too small (< 23 individuals across all years) to support reliable statistical analysis.

132   We also analyzed standardized capture rates for the two most common non-endemic

133 residents (Rufous-throated Solitaire [*Myadestes genibarbis*] and Greater Antillean Bullfinch

134 [*Loxigilla violacea*]) and the three most frequently encountered North American migrants at our

135 sites: Bicknell's Thrush, Ovenbird (*Seiurus aurocapilla*), and Black-throated Blue Warbler

136 (*Setophaga caerulescens*). Although our capture efforts occurred outside the local avian breeding

137 season (Latta et al., 2006), all of the resident species that we captured breed regularly at both

138 sites (Rimmer et al., 2008; CC Rimmer, unpublished data).

139   We assumed that the number of captures of each species could be approximated by the

140 Poisson distribution and used a generalized linear model to examine temporal and spatial trends

141 in capture rate among species. The response variable was the number of unique individuals (new

142 bandings and returns from previous sessions, but not repeat captures from the same session) of

143 each species captured during each unique capture session (hence two data points for PUVI in

144 1998, when we visited in both February and March). We accounted for variation in capture

145 effort by using the number of net hours per capture session (log-transformed) as an offset in the

146 model. We calculated net hours by multiplying the number of 12-m mist nets (or their

147 equivalent; e.g., a 6-m net open was equivalent to a 0.5 12-m net, whereas an 18-m net was

148   equivalent to 1.5 12-m nets) in use during each session by the length of time each was open. For

149   the purposes of standardization with other constant-effort mist-netting studies, we report capture

150   rate per 1,000 net hours (i.e., expected captures for every 1,000 hours that 12-m net was open).

151   The predictor variables included year, site, and the interaction between site and year. We

152   considered three models for each species: year only, site + year, and site*year. We chose among

153   these competing models with a likelihood-ratio test. We estimated temporal trends using the

154   estimated coefficient for the year effect in the best model, and established an a priori significance

155   level of $\alpha = 0.05$.

156   Once we had identified the best model in this way, we examined whether we could

157   further improve model fit by adding to the best model a parameter reflecting the average

158   multivariate El Niño-Southern Oscillation (ENSO) index (MEI) during June to December prior

159   to each banding session. We used lagged values from the previous June to December because

160   they provided a measure of the relative strength of the ENSO event and thus the potential

161   influence on rainfall during the wet and dry seasons preceding our banding sessions. As they

162   build in strength, warm ENSO events are associated with anomalously dry conditions during the

163   late wet season (September – October) and most of the subsequent dry season (November –

164   March), and with anomalously wet conditions during the early wet season (April – July) of the

165   following year as the event subsides (Chen and Taylor, 2002). We downloaded bimonthly MEI

166   values from http://www.esrl.noaa.gov/psd/enso/mei/table.html for the June to December prior to

167   each banding session, and averaged these values to produce a single average value for those six

168   months, which we then added as a covariate to the best-fitting model. We determined whether

169   addition of the MEI covariate improved model fit via a likelihood-ratio test.

170   We examined residual plots and QQ-plots for any deviations from model assumptions,

171   and we used the ratio of the residual deviance to the deviance degrees of freedom as a measure of

172   overdispersion. We found no evidence of deviation from model assumptions. We found little

173   evidence of overdispersion (residual deviance < 2 times the residual degrees of freedom), so we

174   made no adjustment to the models (although we note that quasi-Poisson and negative binomial

175   models produce results that do not differ qualitatively from the Poisson). We report pseudo-R^2

176   as an approximate measure of the explanatory power of the best model in each analysis,

177   calculated as: $1 - \frac{residual\ deviance}{null\ deviance}$.

178   All analyses were conducted using R (R Core Team, 2015). All data used in this analysis

179   are available in Lloyd (2015). Banding activities were conducted by permission of the USGS

180   Bird Banding Lab, under a permit issued to CCR, and research activities in the Dominican

181   Republic were approved by the Subsecretaria de Áreas Protegidas y Biodiversidad.

182

183   **RESULTS**

184   In 14 banding sessions conducted over 13 years (Table 1), yielding >22,000 net hours, we

185   captured a total of 31 species (Table 2). The endemic Green-tailed Ground-Tanager was the

186   most commonly encountered species; sample sizes for species included in this analysis ranged

187   from 69 - 245 individuals (Table 2).

188   Capture rates declined over the course of our study for Rufous-throated Solitaire (β_{year} = -

189   0.04, 95% CL = -0.076, -0.001; P = 0.04; Fig. 1), Green-tailed Ground Tanager (β_{year} = -0.07;

190   95% CL = -0.092, -0.040; P < 0.001; Fig. 2), and Greater Antillean Bullfinch (β_{year} = -0.05; 95%

191   CL = -0.097, -0.009, P = 0.02; Fig. 3). These estimated coefficients equate to expected annual

192   declines in the number of captures of 3.9% (95% CL = 0%, 7.3%) for Rufous-throated Solitaire,

193　6.8% (95% CL = 3.9%, 8.8%) for Green-tailed Ground-Tanager, and 4.9% (95% CL = 0.9%,

194　9.2%) for Greater Antillean Bullfinch. Capture rate also varied by site for these species; for

195　Rufous-throated Solitaire, expected counts were higher at PALO (Fig. 1), whereas for Green-

196　tailed Ground-Tanager and Greater Antillean Bullfinch counts were greater at PUVI (Figs. 2, 3).

197　We found no evidence of a site-by-year interaction in capture rates for any of these species, nor

198　did we find that adding the MEI as a covariate improved model fit. The reduction in residual

199　deviance gained by adding MEI as a covariate to the best model was consistently small (0.012 –

200　1.0) and always non-significant (all likelihood-ratio test P-values >0.28). The pseudo-R^2 of the

201　best model for Green-tailed Ground Tanager was relatively high (75.8%), whereas the percent of

202　variation explained by the best model was moderate for Greater Antillean Bullfinch (46.7%) and

203　low for Rufous-throated Solitaire (30.0%).

204　　　　Two other endemics, Hispaniolan Highland-Tanager and Western Chat-Tanager, showed

205　mixed evidence of declines in capture rate. The preferred model for Hispaniolan Highland-

206　Tanager included significant effects of year (β_{year} = -0.23; 95% CL = -0.35, -0.12; P = <0.001),

207　site (β_{site} = -418.8; 95% CL = -687.1, 168.3; P = 0.001), and their interaction ($\beta_{site*year}$ = 0.21; 95%

208　CL = 0.084, 0.344; P = 0.001) (Fig. 4). Adding MEI as a covariate did not improve model fit

209　(deviance reduction = 1.4, P = 0.31). The best model explained 37.7% of observed variation in

210　capture rate. Declines in expected number of captures were predicted for both sites, but the rate

211　of decline was greater at PALO than at PUVI (Fig. 2). At PALO, the expected annual decline

212　was 20.5% (95% CL = 11.7%, 28.8%), while at PUVI it was 2.1% (95% CI = 0.4% - 3.9%).

213　However, these relationships were driven by the exceptionally high capture rate at PALO in

214　1997. When we excluded data from 1997, none of the regression coefficients, including year

215　(β_{year} = -0.05; 95% CL = -0.12, 0.02; P = 0.12), were significantly different from zero.

216   The situation for Western Chat-Tanager was more complicated, as the preferred model

217   also included significant effects for year (β_{year} = -0.10; 95% CL = -0.173, -0.026; P = 0.008), site

218   (β_{site} = -245.1; 95% CL = -440.2, -52.8; P = 0.013), and their interaction ($\beta_{site*year}$ = 0.122; 95%

219   CL = 0.026, 0.220; P = 0.001), but the interaction was such that expected capture rates declined

220   at PALO while remaining steady or gaining slightly at PUVI (Fig. 5). Expected captures at

221   PALO declined by 9.5% (95% CL = 2.5%, 15.9%) each year, whereas at PUVI expected

222   captures rose by 2.3% per year (95% CL = 0%, 4.8%). Furthermore, the apparent decline in

223   captures at PALO was driven entirely by the especially high capture rate in 1997; when we

224   excluded that point and re-ran the analysis, none of the regression coefficients differed

225   significantly from zero. Adding MEI as a covariate did not significantly improve model fit

226   (deviance reduction = 3.5, P = 0.06; β_{MEI} = -0.20; 95% CL = -0.426, 0.009; P = 0.07). The

227   pseudo-R^2 for the best model (site*year) was 30.0%.

228   We found no evidence of any temporal trend in capture rate for Narrow-billed Tody (β_{year}

229   = 0.020; 95% CL = -0.017, 0.058; P = 0.30), Black-crowned Palm-Tanager (β_{year} = -0.028; 95%

230   CL = -0.079, 0.025; P = 0.30), or Hispaniolan Spindalis (β_{year} = 0.009; 95% CL = -0.040, 0.061;

231   P = 0.72), nor did we find evidence for temporal trends in any of the migrant species (Bicknell's

232   Thrush: β_{year} = -0.003; 95% CL = -0.042, 0.037; P = 0.87; Ovenbird: β_{year} = 0.004; 95% CL = -

233   0.029, 0.039; P = 0.78; Black-throated Blue Warbler: β_{year} = 0.013; 95% CL = -0.038, 0.065; P =

234   0.633). In no case was model fit improved by the inclusion of MEI as a covariate (range of

235   deviance reduction: 0.27 – 3.1; all likelihood-ratio test P-values > 0.09).

236

237   **DISCUSSION**

238   Captures rates of three resident species – including one Hispaniolan endemic – declined

239   substantially and consistently over the course of this study. Captures of Green-tailed Ground-

240   Tanager declined by ~63% from 1997-2010, Rufous-throated Solitaire by ~43%, and Greater

241   Antillean Bullfinch by ~51%. All of these species are currently considered Least Concern by the

242   IUCN (BirdLife International, 2012). However, if the trends that we observed are characteristic

243   of changes occurring range-wide, and if capture rate provides a valid index of population size,

244   then all of these species would meet the criteria for uplisting to Vulnerable (≥30% decline over

245   10 years; IUCN, 2012).

246   We have less confidence in estimated trends for two other Hispaniolan endemics,

247   Hispaniolan Highland-Tanager and Western Chat-Tanager. Trends in capture rate varied

248   between sites and were highly influenced by large numbers of individuals captured in 1997, the

249   first year considered in this analysis. We do not understand why capture rates were so high in

250   1997, but we are hesitant to conclude that these species declined solely on the basis of results

251   obtained in that year. An equally plausible conclusion is that populations of these two species at

252   our study sites were not in long-term decline, and that data from 1997 reflected an unusual and

253   temporary, if unexplained, increase in the local population available for capture in our nets.

254   Unfortunately, we did not collect information on residents species during our initial visit in 1995,

255   and the only data from 1996 were collected at one site (PUVI) at a different time of year (early

256   December) and so provide little insight into the apparently high capture rates observed in 1997.

257   Capture rates of the remaining endemics (Narrow-billed Tody, Black-crowned Palm-

258   Tanager, and Hispaniolan Spindalis) were stable during the course of our study. The three

259   migrant species that we examined also showed no trend in capture rate, largely in keeping with

260   concurrent trends estimated on their breeding grounds. Ovenbird and Black-throated Blue

261 Warbler surveys on the breeding grounds indicated a stable to modestly increasing population

262 over the period of our study (Sauer et al., 2014); range-wide estimates of population trend are not

263 available for Bicknell's Thrush, although local declines have been noted (Lambert et al., 2008).

264   We can only speculate about the causes of observed declines. We saw no clear

265 suggestion that declines were related to climate or weather. Reduced food availability mediated

266 by reduced precipitation during warm-phase ENSO events can limit survival of migrant and

267 resident birds in the Neotropics (Sillett, Holmes & Sherry, 2000; Wolfe et al., 2015), but we

268 found no evidence of a relationship between MEI and capture rate for any species. The intact

269 montane forest that characterized our study sites may be resistant to ENSO-driven variability in

270 precipitation (e.g., Wolfe et al., 2015), but the lack of any clear signal of ENSO may also reflect

271 variability in the effect of ENSO on local and regional precipitation patterns (e.g., Jury,

272 Malmgren & Winter, 2007). Hurricanes can have profound effects on bird populations in the

273 Caribbean (Waide 1981; Wiley and Wunderle 1993), but we saw no obvious relationship

274 between the passage of tropical cyclones through our study sites and changes in capture rate. For

275 example, Hurricane Georges, the most powerful cyclone to affect our study area during the

276 course of this research, caused widespread damage across the Dominican Republic and passed

277 almost directly over Sierra de Bahoruco in September 1998. We did not visit either study site in

278 1999, but capture rates for most species were relatively high in 2000, even among those species

279 that showed long-term declines in capture rate. If hurricane-related changes in habitat conditions

280 were responsible for the declines that we observed, then we would have expected a sharp drop in

281 capture rate after 1998.

282   We doubt that changes in the efficiency of our capture efforts can explain the declines, as

283 we controlled for variation in effort and held constant factors that might influence capture rate

284   independently of abundance, such as net location, mesh size, vegetation structure immediately

285   around the nets, and seasonal timing of capture efforts. We also suspect little role for local

286   changes in vegetation structure or composition. Anthropogenic effects on vegetation structure at

287   the sites were minimal and restricted largely to our maintenance of net lanes. Surrounding

288   forests were also largely free from direct, human-caused disturbance. Natural disturbances were

289   limited to a few small, tree-fall canopy gaps, and we consider it unlikely that patchy successional

290   changes caused the consistent declines in capture rate that we observed.

291   Both study sites support large populations of introduced rats (*Rattus rattus* and *R.*

292   *norvegicus*), which are probably important predators of adult birds and nests (Townsend et al.,

293   2009), but why they would affect only certain species is unclear. Also unclear is why rats, which

294   have been established on most islands of the Caribbean for several hundred years (Harper and

295   Bunbury, 2015), including presumably Hispaniola, would precipitate recent declines. However,

296   we do not know the colonization history by rats of our study sites, and so we cannot rule out the

297   possibility that they are recent arrivals and that they may play a role in driving population

298   declines among birds at our study sites. Finally, habitat loss outside of the study area caused by

299   extensive, ongoing deforestation (BirdLife International, 2015) could drive local declines by

300   reducing the regional population and thus reducing both recruitment into local populations and

301   the number of transient individuals exposed to our sampling efforts.

302   Acknowledging that we cannot identify the cause of the observed declines, how might

303   these findings inform assessments of conservation status? Capture rate in mist nets is often a

304   valid index of abundance, and trends in capture rate are usually – but not always – correlated

305   with population trends estimated using other methods (Dunn and Ralph, 2002). Long-term mist-

306   netting efforts like ours have proven useful in identifying We do not have any independent data

307   to verify the relationship between abundance and capture rate at our study sites, but we assume

308   that trends in capture rate reflected trends in the number of individuals available for capture.

309   However, whether the trends described here were purely a local phenomenon is uncertain. We

310   sampled a small number (n = 2) of purposefully selected sites (undisturbed by human activity)

311   within the montane cloud forest of Sierra de Bahoruco, and so we cannot use these data to draw

312   inference more broadly about the range-wide status of any species. Nonetheless, when combined

313   with other sources of information on population trends, our findings are useful in highlighting

314   which species warrant closer scrutiny. For example, all of the declining species in this study

315   have shown similar trends in other recent evaluations of conservation status (Latta et al., 2006;

316   BirdLife International, 2012), which suggests, but does not demonstrate, that the patterns we

317   described may not be limited solely to our study areas.

318   Even with these findings, which represent the only long-term, quantitative information

319   available on population trends for these species, substantial uncertainty remains regarding range-

320   wide patterns of vulnerability. Intensive studies like ours can provide useful information about

321   ecology and local demographics, but are limited in the scope of inference that they allow

322   regarding overall changes in population parameters. Given the challenges of funding and

323   executing geographically extensive biodiversity monitoring studies, especially in the tropics, it is

324   unlikely that any additional quantitative information can be collected in the short-term that

325   would help resolve this uncertainty. As such, a useful next step might be to focus on other

326   criteria important in assessing vulnerability (IUCN 2012), such as extent of occurrence, that

327   could be quantified using currently available data (e.g., species distribution modeling using data

328   from sources like eBird) but which are not well described at present.

329

Acknowledgments

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1

Trends in capture rate of Rufous-throated Solitaire (*Myadestes genibarbis*)

Observed capture rate (dots) of Rufous-throated Solitaire and changes in expected capture rate (solid line; shaded area = 95% confidence interval) per 1000 net-hours at two sites (PALO = Palo de Agua, PUVI = Pueblo Viejo) in montane broadleaf forest of Sierra de Bahoruco, Dominican Republic.
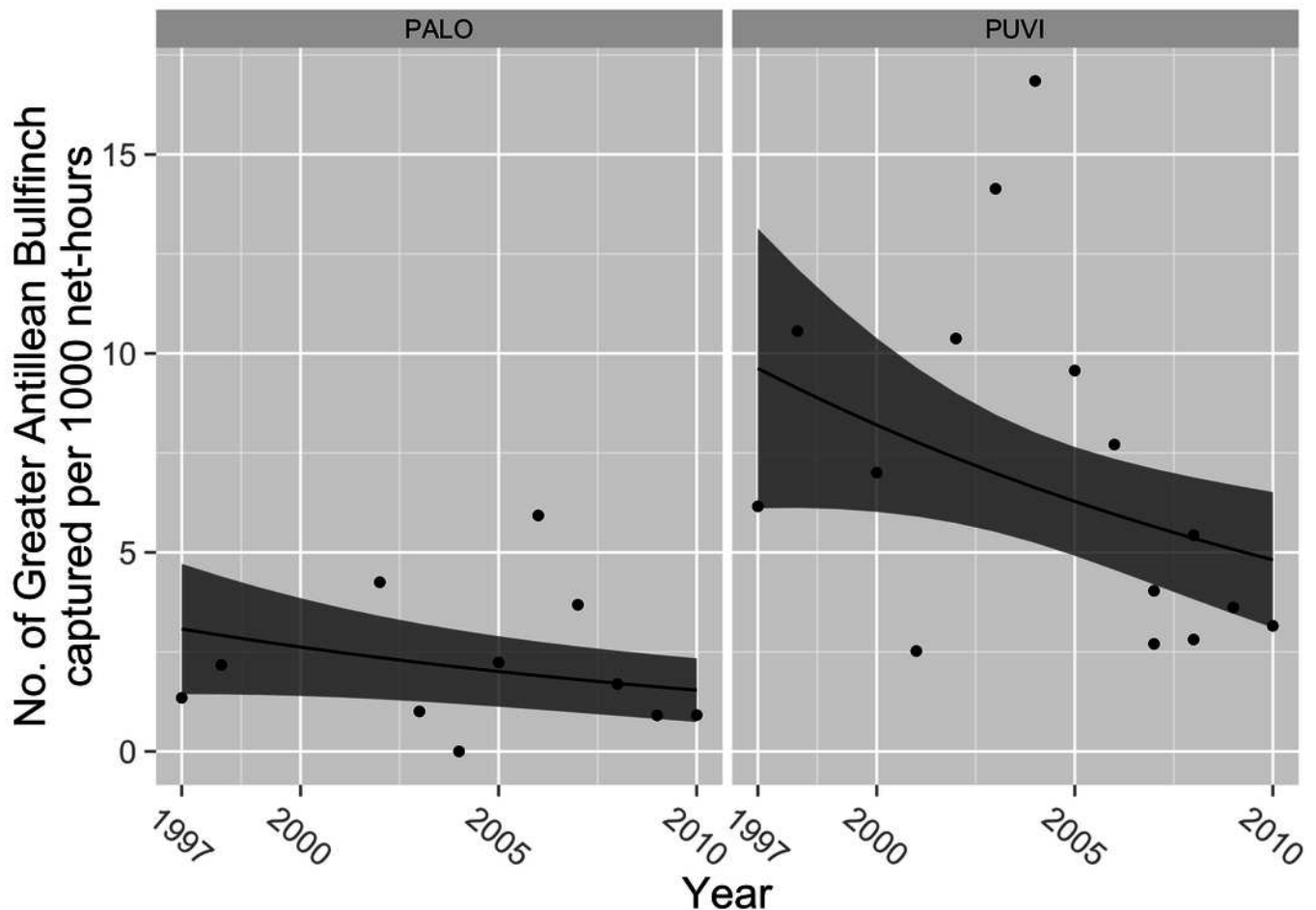
2
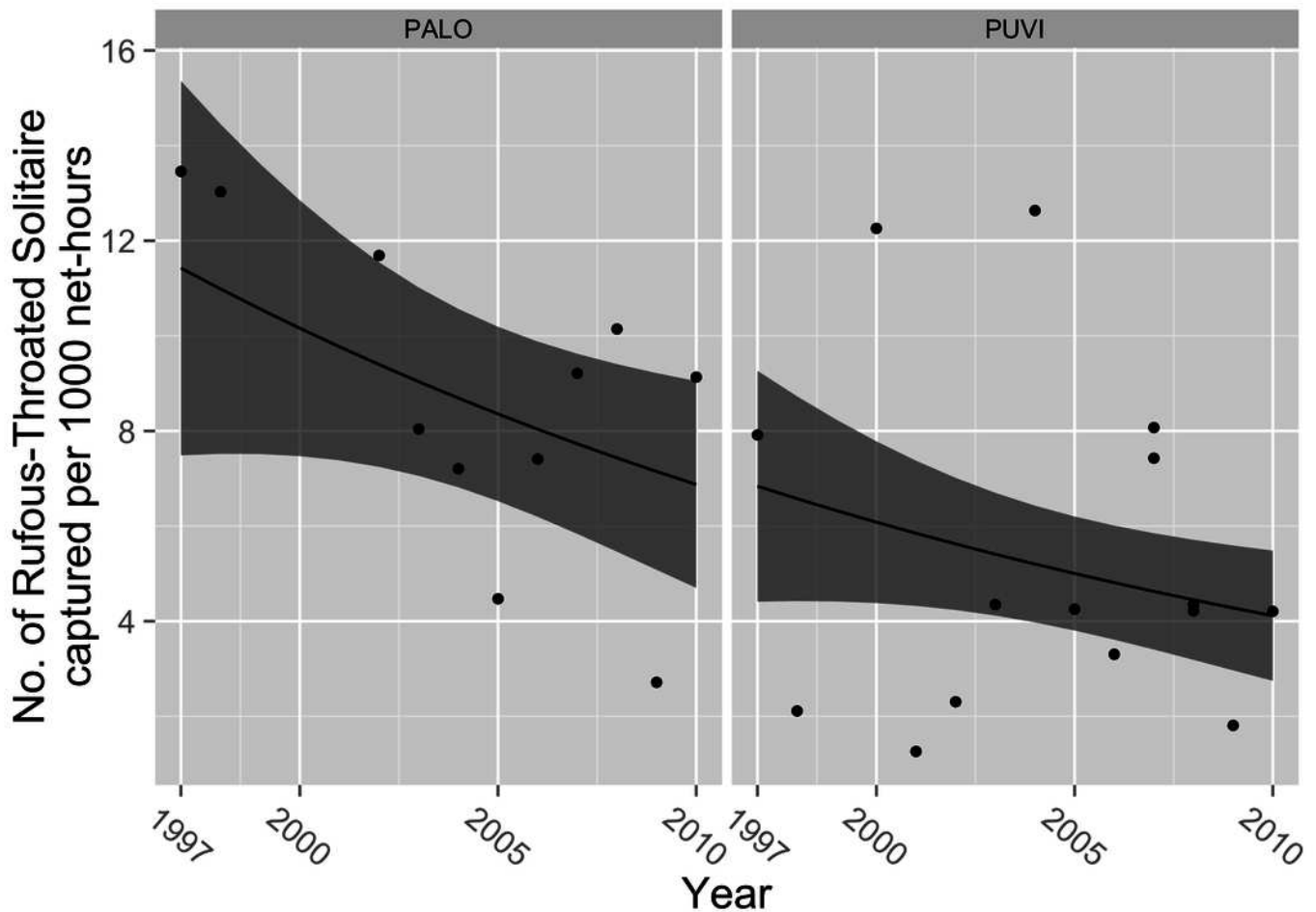
Trends in capture rate of Green-tailed Ground-Tanager (*Microlegia palustris*)

Observed capture rate (dots) of Green-tailed Ground-Tanager and changes in expected capture rate (solid line; shaded area = 95% confidence interval) per 1000 net-hours at two sites (PALO = Palo de Agua, PUVI = Pueblo Viejo) in montane broadleaf forest of Sierra de Bahoruco, Dominican Republic.

3
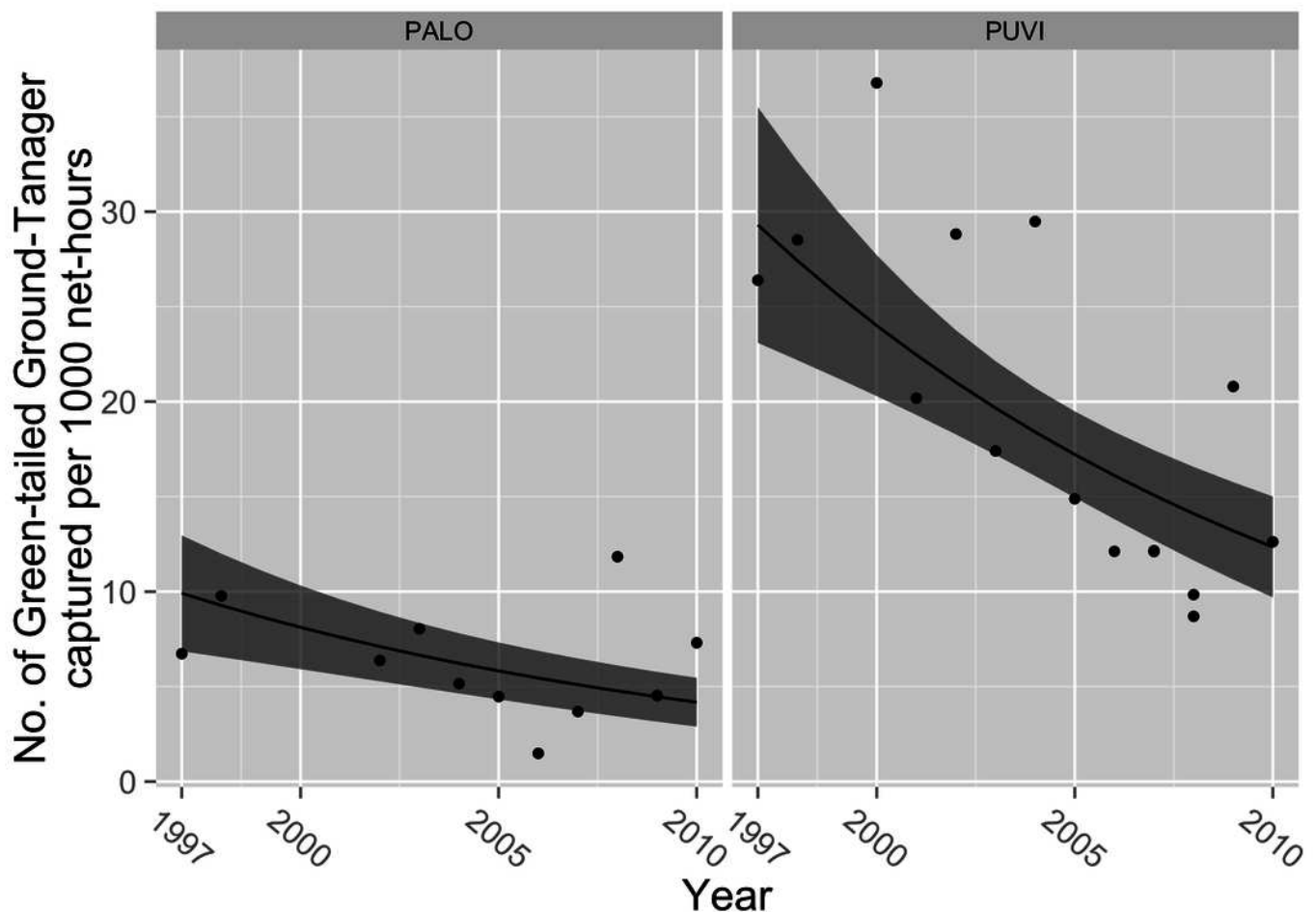
Trends in capture rate of Greater Antillean Bullfinch (*Loxigilla violacea*)

Observed capture rate (dots) of Greater Antillean Bullfinch and changes in expected capture rate (solid line; shaded area = 95% confidence interval) per 1000 net-hours at two sites (PALO = Palo de Agua, PUVI = Pueblo Viejo) in montane broadleaf forest of Sierra de Bahoruco, Dominican Republic.

4

Trends in capture rate of Hispaniolan Highland-Tanager (*Xenolegia montana*)

Observed capture rate (dots) of Hispaniolan Highland-Tanager and changes in expected capture rate (solid line; shaded area = 95% confidence interval) per 1000 net-hours at two sites (PALO = Palo de Agua, PUVI = Pueblo Viejo) in montane broadleaf forest of Sierra de Bahoruco, Dominican Republic.

5
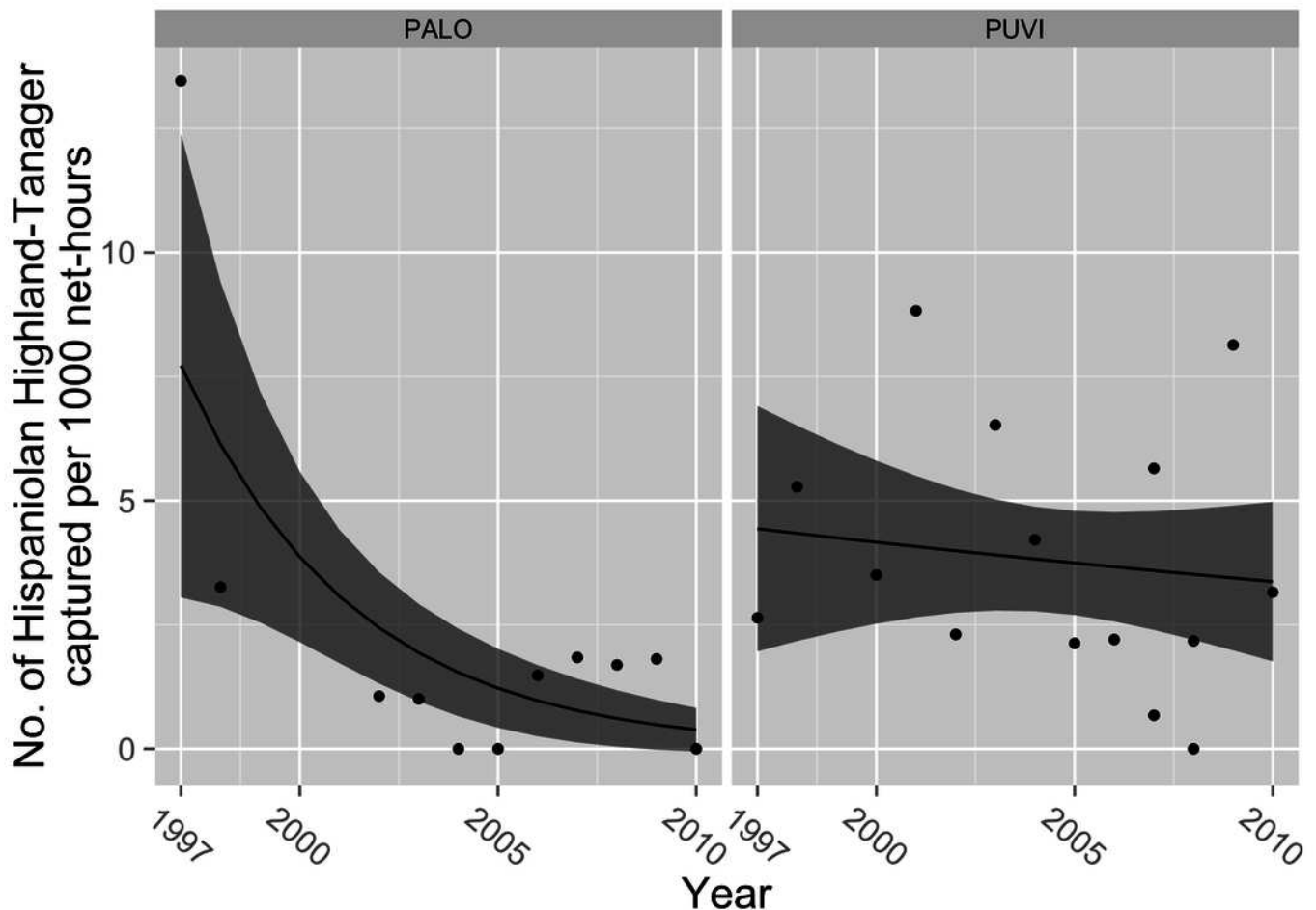
Trends in capture rate of Western Chat-Tanager (*Calyptophilus tertius*)

Observed capture rate (dots) of Western Chat-Tanager and changes in expected capture rate (solid line; shaded area = 95% confidence interval) per 1000 net-hours at two sites (PALO = Palo de Agua, PUVI = Pueblo Viejo) in montane broadleaf forest of Sierra de Bahoruco, Dominican Republic.
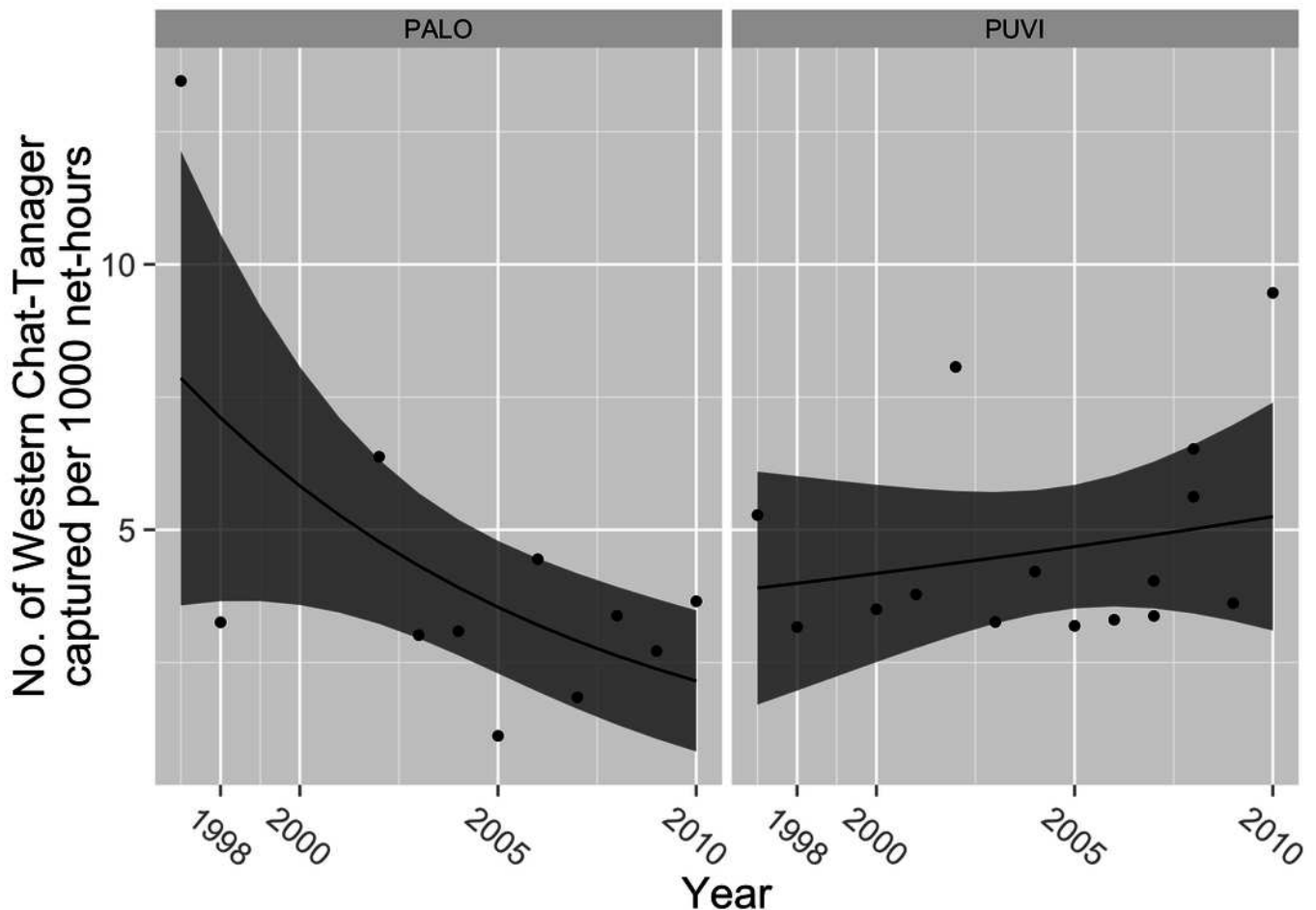
Table 1(on next page)

Summary of capture effort.

Dates of banding sessions and total net hours at two sites in the Sierra de Bahoruco, Dominican Republic from 1997-2010.

Date	Site[a]	Net hours[b]
1997 February 28 – March 08	PUVI	1136.8
	PALO	743.1
1998 March 04 – March 11	PUVI	947.0
	PALO	921.0
2000 January 24 – January 27	PUVI	571.0
	PALO	0
2001 January 30 – February 05	PUVI	793.0
	PALO	0
2002 February 10 – February 17	PUVI	867.5
	PALO	941.0
2003 January 30 – February 11	PUVI	919.5
	PALO	995.0
2004 February 19 – February 28	PUVI	474.9
	PALO	971.5
2005 February 04 – February 10	PUVI	940.5
	PALO	895.8
2006 January 26 – January 31	PUVI	907.9
	PALO	674.8
2007 January 31 – February 07	PUVI	1481.0
	PALO	1085.8
2008 February 07 – February 12	PUVI	920.0
	PALO	591.5
2008 March 13 – March 16	PUVI	711.5
	PALO	0
2009 February 13 – February 20	PUVI	1106.0
	PALO	1105.0
2010 March 14 – March 21	PUVI	951.0
	PALO	1095.0

1 [a] PUVI = Pueblo Viejo, PALO = Palo de Aqua

2 [b] Net hours = total number of 12-m-equivalent nets * number of hours open

3

Table 2(on next page)

Summary of number of individuals captured.

Number of individuals captured during annual banding sessions conducted at two sites in the Sierra de Bahoruco, Dominican Republic from 1997-2010.

1

Species	Total individuals captured
Sharp-shinned Hawk (*Accipiter striatus*)	12
White-fronted Quail-Dove (*Geotrygon leucometopia*)*	7
Hispaniolan Parakeet (*Aratinga chloroptera*)*	1
Hispaniolan Emerald (*Chlorostilbon swainsonii*)*,[1]	47
Narrow-billed Tody (*Todus angustirostris*)*	140
Hispaniolan Woodpecker (*Melanerpes striatus*)*	22
Hispaniolan Trogon (*Priotelus roseigaster*)*	10
Hispaniolan Pewee (*Contopus hispaniolensis*)*	43
Greater Antillean Elaenia (*Elaenia fallax*)	29
Rufous-throated Solitaire (*Myadestes genibarbis*)	126
Bicknell's Thrush (*Catharus bicknelli*)	149
La Selle Thrush (*Turdus swalesi*)*	22
Red-legged Thrush (*Turdus plumbeus*)	31
Gray Catbird (*Dumetella carolinensis*)	1
Ovenbird (*Seiurus aurocapilla*)	162
Worm-eating Warbler (*Helmitheros vermivorum*)	4
Black-and-white Warbler (*Mniotilta varia*)	28
Swainson's Warbler (*Limnothlypis swainsonii*)	7
Kentucky Warbler (*Geothlypis formosa*)	1
Common Yellowthroat (*Geothlypis trichas*)	2
American Redstart (*Setophaga ruticilla*)	3
Black-throated Blue Warbler (*Setophaga caerulescens*)	83
Pine Warbler (*Setophaga pinus*)	1
Hispaniolan Highland-Tanager (*Xenoligea montana*)*	69
Green-tailed Ground-Tanager (*Microligea palustris*)*	245
Banaquit (*Coereba flaveola*)	4
Black-crowned Palm-Tanager (*Phaenicophilus palmarum*)*	77
Western Chat-Tanager (*Calyptophilus tertius*)*	72
Hispaniolan Spindalis (*Spindalis dominicensis*)*	85
Black-faced Grassquit (*Tiaris bicolor*)	28
Greater Antillean Bullfinch (*Loxigilla violacea*)	86

2 *Hispaniolan endemic.

3 [1] The total number of unique individuals captured is unknown because we could not permanently

4 mark individuals with leg bands.

5