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## Potential predators of an invasive frog (*Eleutherodactylus coqui*) in Hawaiian forests

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In Hawaii, where there are no native reptiles or amphibians, 27 species of reptiles and amphibians have established (Kraus 2003); however, few have been studied to determine their ecological impacts. For example, little is known about the impacts of the Puerto Rican frog, *Eleutherodactylus coqui* Thomas, that recently invaded (late 1980s) (Kraus *et al.* 1999), and has established on all four main Hawaiian Islands (Kraus & Campbell 2002). However, there are likely to be consequences because *E. coqui* can attain high densities (20 570 frogs ha<sup>-1</sup> on average in Puerto Rico) and consume large quantities of invertebrates (114 000 prey items ha<sup>-1</sup> per night on average in Puerto Rico) (Stewart & Woolbright 1996).

One potential consequence of the invasion is that *E. coqui* could indirectly reduce endemic prey by suddenly increasing population sizes of introduced predators (Courchamp *et al.* 2000). More specifically, Kraus *et al.* (1999) proposed that *E. coqui* may indirectly contribute to endemic bird declines by bolstering populations of non-native bird predators, such as rats (e.g. black (*Rattus rattus* L.) and Polynesian (*R. exulans* Peale)) and the small Indian mongoose (*Herpestes javanicus* E. Geoffroy Saint-Hilaire). In this study, we investigated whether *E. coqui* serves as a food source for introduced predators, *R. rattus*, *R. exulans* and *H. javanicus*, and the cane toad (*Bufo marinus* L.), in an area with high *E. coqui* densities.

Research was conducted in Lava Tree State Park (LTSP; N 19° 28.99' W 154° 54.20'; 181 m asl) located on the eastern side of the Island of Hawaii. The park is located on 200–750 y old a'a type lava rock, and has a mean annual temperature of 23 °C and mean annual precipitation of 2500 mm (Giambelluca *et al.* 1996). *Falcataria moluccana* (Miquel) Barneby & Grimes, *Cecropia obtusifolia* Bertol. and *Metrosideros polymorpha* Gaud. dominate

the forest overstorey, and *Clidemia hirta* (L.) D. Don, *Psidium cattleianum* Sabine and *Melastoma candidum* D. Don dominate the understorey. The *E. coqui* population at LTSP was established in 1996, and was determined to have a density of 55 000 individuals ha<sup>-1</sup> in August 2004 (Beard & Pitt 2005).

The sampling area for rats and mongooses consisted of a trapping web with an effective trapping area of c. 2 ha. The web was bordered by a trail on two sides and Highway 132 on another side. The web contained sixteen 80-m transects radiating from a central point. Eight traps were placed along each transect at 10-m intervals. Every other transect had eight Tomahawk live traps or eight snap traps for a total of 128 traps.

Removal trapping was conducted from 10–16 June 2004 (period 1), from 2–14 January 2005 (period 2) and from 27 January–20 February 2005 (period 3). Sampling periods were intended to capture seasonal differences (Sugihara 1997). Traps were left open during sampling periods and checked twice daily, starting at 08h00 and 19h00. Traps were baited with shredded coconut and/or fish emulsion. Specimens trapped in Tomahawk traps were taken back to the laboratory and euthanized. All carcasses were frozen for later examination.

We used regression-based depletion estimates to estimate the population size ( $\hat{N}$ ) of mammal species for each sampling period (Krebs 1999). Data were pooled across days into four sampling points for each sampling period. For some sampling periods for some species, it was not possible to fit a depletion model; hence these sampling periods were excluded from the mean. Standard deviations (SD) were calculated across sampling periods not around values of  $\hat{N}$ . Density was then estimated by dividing population size by area sampled.

In August 2004, we established a 100 × 100-m plot within the trapping web to collect *B. marinus*. For five nights, starting at 21h00, transects within the plot were

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searched for toads for a total of 100 min by three researchers with headlamps. Toads were taken back to the laboratory and euthanized. Carcasses were stored in 70% ethanol for later examination.

For each animal collected, the entire digestive tract (oesophagus to colon) was removed and contents were suspended in a 5% buffered formalin solution. Contents were then washed with distilled water, run through a 2-mm sieve, and dried for 24 h at 35 °C. For each individual, dried samples were dissected under a microscope and contents were separated into broad categories (plant, invertebrate (to scientific order where possible), avian, amphibian, reptilian or mammalian) and weighed.

We used undigested subadult and adult *E. coqui* and *B. marinus* skeletons to identify digested frogs, using bone size and unique bone structures, such as urostyles, ilia and skulls. No other frogs were observed in the study area. Avian items were identified by the presence of feathers and unique bones, such as sterna or mandibles; mammalian parts were identified by the presence of hair; and the reptilian and Lepidoptera larvae samples were identified by their skin.

During 33 d (2112 trap-days), a total of 46 rats and 22 *H. javanicus* was collected. During sampling period 1, 3 *R. exulans*, 14 *R. rattus* and 10 *H. javanicus* were collected. During period 2, 6 *R. exulans*, 4 *R. rattus*, and 8 *H. javanicus* were collected, and during period 3, 16 *R. exulans*, 1 *R. rattus*, and 4 *H. javanicus* were collected. Trapped *R. exulans* measured  $274 \pm 18$  mm (mean  $\pm$  SD) total length and  $143 \pm 8$  mm in tail length. Trapped *R. rattus* measured  $377 \pm 39$  mm total length and  $208 \pm 16$  mm in tail length. For both species, the sex ratio was biased towards males (1:0.6). Trapped mongooses measured  $531 \pm 39$  mm total length and  $235 \pm 26$  mm in tail length. The sex ratio was biased towards females (1:2.1). We estimated that there were  $5.0 \pm 3.5$  *R. exulans*  $\text{ha}^{-1}$ ,  $3.6$  *R. rattus*  $\text{ha}^{-1}$  and  $4.7 \pm 1.9$  *H. javanicus*  $\text{ha}^{-1}$ . We collected 27 *B. marinus* with a mean SVL of  $80 \pm 16$  mm and a sex ratio biased towards females (1:0.4).

Kraus *et al.* (1999) proposed that rats could be predators of *E. coqui* in Hawaii. This appeared likely because rats are nocturnal and arboreal, which means that they are able to actively hunt *E. coqui* foraging on leaves and trunks. However, we found that not one rat consumed *E. coqui* in an area infested with a dense *E. coqui* population (Figure 1). Rather, we found that rats mostly consumed plant material (fruits). Thus, our data suggest that *E. coqui* is unlikely to bolster these rat populations.

Much of the literature on the diet of *R. rattus* in Puerto Rico describes preferences for plant materials (Muniz-Melendez 1978, Sastre-De Jesus 1979, Weinbren *et al.* 1970). A review of the diet of *R. rattus* in the wet forests of El Verde only described plant material, despite an *E. coqui* density of  $20\,000 \text{ ha}^{-1}$  (Willig & Gannon 1996), although there has been personal observation of *E. coqui* consumption by *R. rattus* in El Verde (Stewart &

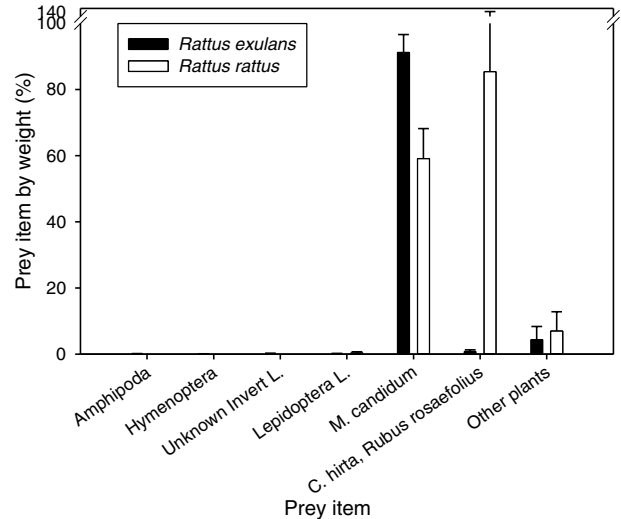


Figure 1. Per cent of prey items by weight ( $\pm$  SE) for *Rattus exulans* and *R. rattus* from Lava Tree State Park, Island of Hawaii (N = 25 and N = 19, respectively). invert. = invertebrate, L. = larvae.

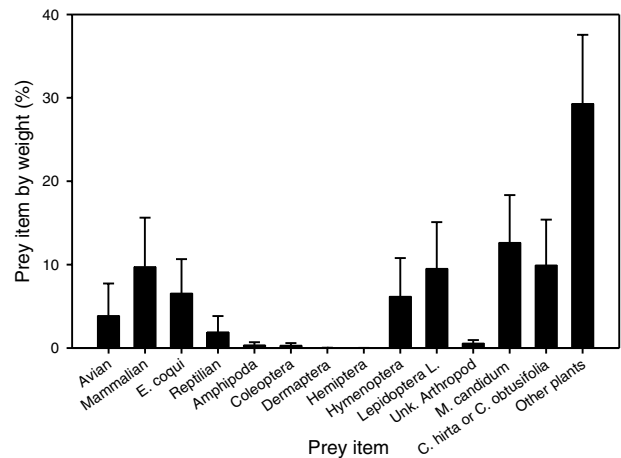


Figure 2. Per cent of prey items by weight ( $\pm$  SE) for *Herpestes javanicus* from Lava Tree State Park, Island of Hawaii (N = 22). L. = larvae, unk. = unknown.

Woolbright 1996). In summary, there are few data from either location suggesting that *E. coqui* will constitute a large proportion of rat diets.

Kraus *et al.* (1999) proposed that mongooses could consume *E. coqui* in Hawaii. We found that, even though mongooses are diurnal and reportedly poor climbers, *E. coqui* constituted  $6.6 \pm 19.2\%$  of their prey items by weight (Figure 2) and three mongooses (14%) had *E. coqui* bones in their digestive tracts. We found that one mongoose consumed 14 *E. coqui* and two others each consumed one *E. coqui*. We expect that mongooses consume *E. coqui* opportunistically in leaf litter or rocks.

Assuming digestive tract contents represent a day of feeding, the results translate to an average of 0.73 *E. coqui* consumed per mongoose  $\text{d}^{-1}$ . Using our estimate of  $4.7$  *H. javanicus*  $\text{ha}^{-1}$ , this converts to a consumption

rate of 1240 *E. coqui* ha<sup>-1</sup> y<sup>-1</sup>. This may impact this *E. coqui* population, which is estimated to recruit over 30 000 *E. coqui* ha<sup>-1</sup> y<sup>-1</sup> (assuming stable populations and using survivorship estimates from Stewart & Woolbright 1996). However, it is difficult from these results alone to determine whether *E. coqui* is influencing this mongoose population.

At the time of sampling, *E. coqui* and *H. javanicus* had only co-existed in this location for 8 y. It may be that *E. coqui* will become an increasingly important component of mongoose diets over time. However, data from Puerto Rico, where *E. coqui* and *H. javanicus* have been co-existing for over 100 y (Wadsworth 1949), also suggest that *E. coqui* represents a small percentage of mongoose diets. Pimentel (1955) found that *Eleutherodactylus* spp. constituted 1% of prey items by volume (N = 56 mongooses), whereas Vilella (1998) found that *Eleutherodactylus* spp. were only consumed by 2 (11%) of the 18 mongooses examined.

We thought that *B. marinus* may be a predator of *E. coqui*, but not one of the *B. marinus* examined consumed *E. coqui*. Of food items found in *B. marinus*, 40% by weight was plant material, which mostly consisted of *F. moluccana* leaves and bark. Of the animal items found, the large majority by weight consisted of Coleoptera (17.4%), Diplopoda (22.8%), Gastropoda (36.5%) and Lepidoptera larvae (9.6%). No other order made up greater than 5% of the weight of animal items consumed. Similarly, *B. marinus* has not been found to consume *E. coqui* in Puerto Rico (N = 301 toads) (Dexter 1932). It does not appear likely that *E. coqui* will bolster populations of *B. marinus* in Hawaii. Rather, the long-term effect of these two species on each other may be through competition for resources (for preliminary *E. coqui* diet data in Hawaii see Beard & Pitt 2005) or the spread of disease (Beard & O'Neill 2005).

We found that research on potential predators of an invasive species in its native range informed the extent of predation by these species in an introduced range. Because the hypothesis that *E. coqui* may bolster predator populations in Hawaii has important implications, we suggest further research be conducted on the topic. First, it would be important to determine if densities of potential predators respond to changes in *E. coqui* densities. It would also be important to determine if there is evidence of sustained use of *E. coqui* by potential predators. Finally, studies should investigate whether at risk endemic populations show reduced survival in the presence of *E. coqui*.

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