



Technical Report HCSU-010

REDUCING FERAL CAT THREATS TO NATIVE WILDLIFE IN HAWAII

Prepared by Steven C. Hess¹, Heidi Hansen^{2,3}, and Paul C. Banko¹

¹ U.S. Geological Survey Pacific Island Ecosystems Research Center,
P.O. Box 44, Hawai'i National Park, HI 96718, USA

² Hawai'i Cooperative Studies Unit, University of Hawai'i at Hilo, Pacific Aquaculture and Coastal Resources Center,
Kilauea Field Station, P.O. Box 44, Hawai'i National Park, HI, 96718, USA

³ Current address: Department of Land and Natural Resources, Division of Forestry and Wildlife,
19 E. Kawili St., Hilo, HI 96720, USA

Hawai'i Cooperative Studies Unit
University of Hawai'i at Hilo
Pacific Aquaculture and Coastal Resources Center (PACRC)
200 W. Kawili St.
Hilo, HI 96720
(808) 933-0706

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¹ USGS Pacific Island Ecosystems Research Center, P.O. Box 44, Hawai`i National Park, HI 96718, USA

² Hawai`i Cooperative Studies Unit, University of Hawai`i at Hilo, Pacific Aquaculture and Coastal Resources Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

³ Current address: Department of Land and Natural Resources, Division of Forestry and Wildlife, 19 E. Kawili St., Hilo, HI 96720, USA

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Hawai`i Cooperative Studies Unit
University of Hawai`i at Hilo
Pacific Aquaculture and Coastal Resources Center
200 W. Kawili St.
Hilo, Hawai`i 96720
(808) 933-0706

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I) Executive Summary

We documented the diet of feral cats (*Felis catus*) on Kīlauea and Mauna Loa within Hawai`i Volcanoes National Park (HAVO), determined the incidence of three feline diseases on Mauna Kea, studied feral cat home range, developed and tested trap-signaling devices, tested food-based baits and attractants, analyzed feral cat population dynamics using genetic techniques, and developed an adaptive strategy for reducing predation on endangered Hawaiian birds.

We documented the diet of feral cats by analyzing the contents of 42 digestive tracts from Kīlauea and Mauna Loa in Hawai`i Volcanoes National Park. Small mammals, invertebrates, and birds were the most common prey types consumed by feral cats. Birds occurred in 27.8–29.2% of digestive tracts. The total number of bird, small mammal, and invertebrate prey differed between Kīlauea and Mauna Loa. On Mauna Loa, significantly more (89%) feral cats consumed small mammals, primarily rodents, than on Kīlauea Volcano (50%). Mice (*Mus musculus*) were the major component of the feral cat diet on Mauna Loa, whereas Orthoptera were the major component of the diet on Kīlauea. We recovered a mandible set, feathers, and bones of an endangered Hawaiian Petrel (*Pterodroma sandwichensis*) from a digestive tract from Mauna Loa. This specimen represents the first well-documented endangered seabird to be recovered from the digestive tract of a feral cat in Hawai`i and suggests that feral cats prey on this species.

We determined prevalence to feline immunodeficiency virus (FIV) antibodies, feline leukemia virus (FeLV) antigen, and *Toxoplasma gondii* antibodies in feral cats on Mauna Kea Hawai`i from April 2002 to May 2004. Six of 68 (8.8%) and 11/68 (16.2%) were antibody positive to FIV and antigen positive for FeLV, respectively; 25/67 (37.3%) were seropositive to *T. gondii*. Antibodies to FeLV and *T. gondii* occurred in all age and sex classes, but FIV occurred only in adult males. Evidence of previous or current infections with two of these infectious agents was detected in eight of 64 cats (12.5%). Despite exposure to these infectious agents, feral cats remain abundant throughout the Hawaiian Islands.

Feral cats in dry subalpine woodland of Mauna Kea, Hawai`i, live in low density and exhibit some of the largest reported home ranges in the literature. While 95% fixed kernel home range estimates for three females averaged 772 ha, four males averaged 1,418 ha, and one male maintained a home range of 2,050 ha. Mean daily movement rates between sexes overlapped widely and did not differ significantly ($P = 0.083$). Log-transformed 95% kernel

home ranges for males were significantly larger than those of females ($P = 0.024$), but 25% kernel home ranges for females were larger than those of males ($P = 0.017$). Moreover, log-transformed home ranges of males were also significantly larger than those of females in this and seven other studies from the Pacific region ($P = 0.044$). Feral cats present a major threat to endangered Hawaiian birds, but knowledge of their ecology can be used for management by optimizing trap spacing and creating buffer zones around conservation areas.

Frequent checks of live traps require enormous amounts of labor and add human scents associated with repeated monitoring which may reduce capture efficiency. To reduce efforts and increase efficiency, we developed a trap-signaling device with long-distance reception, durability in adverse weather, and ease of transport, deployment, and use. Modifications from previous designs include a normally-open magnetic switch and a mounting configuration to maximize reception. The system weighed < 225 g, was effective ≤ 17.1 km, and failed in $< 1\%$ of trap-nights. Employing this system, researchers and wildlife managers may reduce the amount of effort checking traps while improving the welfare of trapped animals.

Successful feral cat control programs require effective baits and lures. Non-targets may interfere with trapping efforts by rapidly consuming bait before feral cats encounter traps, necessitating frequent bait replacement. We compared the effectiveness of baits and lures by analyzing capture rates of feral cats and non-targets and monitoring animal visits to bait stations with remotely-triggered cameras. We tested four different baits and attractants: canned cat food, sardines, catnip, and a bait sausage that we formulated from pork and fat. We trapped for a total of 3,389 trap nights and captured 35 feral cats. There were 323 incidences of trap interferences, reducing the effective trap nights (ETN) to 3,225. The primary cause of trap interference was feral pigs rolling over traps ($n = 185$, 57.3% of interferences). The primary non-target species captured were small Indian mongooses ($n = 74$, 22.9% of interferences). Overall, more cats and mongooses were captured using sardines, although the catch frequencies were not dependent on the bait type used. We obtained photographs of 1,476 small mammals at the bait stations. Mongooses were the principal mammals photographed ($n = 939$, 69.5% of pictures). We also obtained 398 photographs of rats (29.5%) and 9 (0.7%) of mice. Feral cats were photographed only 5 (0.4%) times. We found strong differences between mongooses, rodents, and cats photographed at the four bait types. Sardines were the most visited bait type ($n = 641$, 47.4% of photographs). Pork sausage and cat food accounted for 383 (28.3%) and 322 (23.8%) visits while catnip had only 67 (5.0%)

visits. Feral cats were photographed only at sardine bait. Mongooses were attracted primarily to sardines (49.3%). Pork sausage was the most attractive bait to rats, accounting for 44.5% of photographs. Due to the high rate of non-target interference, other attractants need to be tested for successful feral cat control programs.

Population genetics can provide information about the demographics and dynamics of invasive species that is beneficial for developing effective control strategies. We studied the population genetics of feral cats on Hawai`i Island by microsatellite analysis to evaluate genetic diversity and population structure, assess gene flow and connectivity among three populations, identify potential source populations, characterize population dynamics, and evaluate sex-biased dispersal. High genetic diversity, low structure, and high number of migrants per generation supported high gene flow that was not limited spatially. Migration rates revealed that most migration occurred out of West Mauna Kea. Effective population size estimates indicated increasing cat populations despite control efforts. Despite high gene flow, relatedness estimates declined significantly with increased geographic distance and Bayesian assignment tests revealed the presence of three population clusters. Genetic structure and relatedness estimates indicated male-biased dispersal. Mauna Kea may be a source population that can be targeted for control. However, recolonization seems likely given the great dispersal ability that may not be inhibited by barriers such as lava flows. Genetic monitoring will be necessary to assess the effectiveness of future control efforts.

Despite the long history of feral cats in Hawai`i, there has been little research to provide strategies to improve control programs and reduce depredation on endangered species. Our objective was to develop a predictive model to determine how landscape features on Mauna Kea such as habitat, elevation, and proximity to roads affect the number of feral cats captured at each trap. We used log-link generalized linear models and QAIC_c model ranking criteria to determine the effect of these factors. We found that the number of cats captured per trap was related to trapping effort, habitat type, and whether traps were located on the West or North Slope of Mauna Kea. We recommend an adaptive management strategy to minimize trapping interference by non-target small Indian mongoose (*Herpestes auropunctatus*) with toxicants, to focus trapping efforts in māmane (*Sophora chrysophylla*) habitat on the West slope of Mauna Kea, and to cluster traps near others that have previously captured multiple cats.

II) Diet of Feral Cats in Hawai`i Volcanoes National Park

Steven C. Hess, U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Heidi Hansen¹, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Daniel Nelson, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Roberta Swift², Pacific Cooperative Studies Unit, Hawai`i Volcanoes National Park, P.O. Box 52, Hawai`i National Park, HI 96718, USA

Paul C. Banko, U.S. Geological Survey Pacific Island Ecosystems Research Center, P.O. Box 44, Kīlauea Field Station, Hawai`i National Park, HI, 96718, USA

¹ Current address: Department of Land and Natural Resources Division of Forestry and Wildlife, 19 East Kawili St., Hilo, HI 96720, USA

² Current address: 501 South Cleveland Street, Apt. B, Oceanside, CA 92054, USA

a. Introduction

Domestic cats (*Felis catus*) were brought to Hawai`i on European ships in the late 1700s (King 1984). Subsequently, feral cats were reported to be abundant in forests of Lāna`i and O`ahu by 1892 (Rothschild 1893), and predation of forest birds in the late 1800s was recorded by Perkins (1903). Feral cats currently range throughout the Hawaiian Islands from high-density colonies near coastal areas where pets are frequently abandoned and fed by people (Winter 2003), to remote, low-density populations in montane forests and subalpine areas of Maui (Simons 1983) and Hawai`i Island (Tomich 1986; Hu et al. 2001). Cats are currently important predators of colonial seabirds (Smith et al. 2002) and terrestrial birds that nest near the ground (Kowalsky et al. 2002) and in trees throughout Hawai`i (Hess et al. 2004).

Stomach content analysis of feral cats provides information on the types of prey that sustain cats, and, in conjunction with population dynamics and abundance of prey populations,

the impacts these predators may have on native fauna. Diet may also vary substantially between locations. For example, on Mauna Kea, nearly 80% of cat digestive tracts contained birds, primarily passerines, but also gallinaceous species, reflecting overall bird abundance in this environment (Hess et al. 2004). We examined the diet of feral cats in Hawai`i Volcanoes National Park (HAVO) to determine common diet items, and differences in diet between locations.

b. Study Area

We studied the diet of feral cats over a broad area in HAVO (19° 54' N, 155° 28' W). Several ecological zones were represented in this area, extending from coastal to subalpine environments (Figure 3.1). To determine how prey use changed with elevation, we divided capture locations into two different regions: Kīlauea and Mauna Loa volcanoes. A major highway follows the approximate boundary between these two volcanoes; thus we chose this feature to demarcate populations. Kīlauea extends from the coast to the Kīlauea summit at approximately 1,200 m elevation. This area consists of montane wet forest dominated by `ōhi`a (*Metrosideros polymorpha*) trees with an understory of shade-tolerant trees, shrubs, ferns, and mosses, grading into drier `ōhi`a scrub and shrubland. The Mauna Loa study area extends from above the Kīlauea summit to 2,700 m. This area consists of some old growth koa (*Acacia koa*) and `ōhi`a forests but is primarily characterized as subhumid and subalpine. Vegetative cover can be < 10% at higher elevations and is dominated by shrubby species such as pūkiawe (*Leptecophylla tameiameia*), `a`ali`i (*Dodonaea viscosa*), kūkaenēnē (*Coprosma ernodeoides*), `ōhelo (*Vaccinium reticulatum*), and na`ena`e (*Dubautia ciliolata*) (Wagner et al. 1999). Substrates of both volcanoes are geologically young and interspersed with recent lava flows.

c. Methods

Feral cats were trapped to reduce predation on endangered species such as Nēnē (Hawaiian goose; *Branta sandvicensis*), and `Ua`u (Hawaiian Petrel; *Pterodroma sandwichensis*). Traps were checked daily, minimizing the effect of differential digestion on prey items. Cats were euthanized and entire carcasses were frozen, then stomachs and intestines were removed and preserved in 70% ethanol. We examined the digestive tracts of 24 feral cats collected from Kīlauea and 18 collected from Mauna Loa from 2000–2005. Stomachs were cut

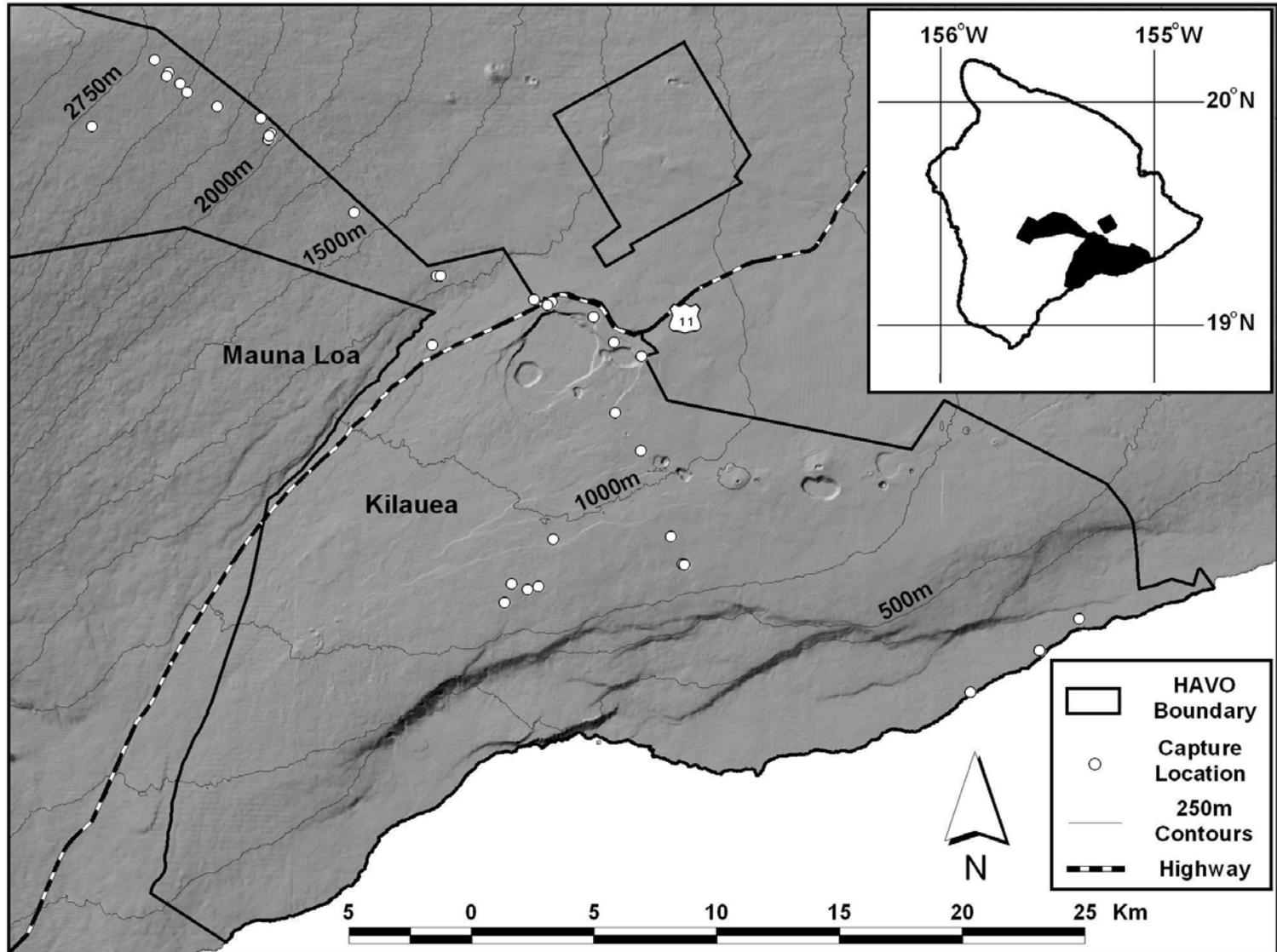


Figure 3.1. Location of feral cats sampled for diet analyses from Kīlauea and Mauna Loa in Hawaiʻi Volcanoes National Park, 2000–2005. Dashed line represents the approximate boundary between Kīlauea and Mauna Loa.

open along the concave side and intestines were opened along their entire length. A continuous, gentle stream of water was used in combination with gentle pressure from fingers to rinse and separate food items through soil sieves (US Bureau of Standards size 5 and 10mesh). Prey items were sorted into respective prey types (e.g., small mammals, invertebrates, birds) and identified to lowest taxonomic level with the help of reference collections. For each prey item, the minimum number of individuals present was estimated from the numbers of unique identifiable parts, such as legs, feet, and insect wings. For example, two left wings of the same taxon indicated a minimum number of two individuals. We did not quantify non-prey items such as plant material, bait, parasitic worms and trapping material. Unmacerated grass blades and shrub leaves were assumed to be consumed as an aid in digestion, a means to eliminate tapeworms in the digestive tract (Neale and Sacks 2001), or secondarily while foraging for grass-dwelling prey (Jones and Barmuta 1998).

For each area, we estimated the mean minimum number and standard error of all major prey types found in cat digestive tracts, the frequency of occurrence (FO; the proportion of digestive tracts containing particular prey items) and its binomial standard error, and the relative frequency of occurrence (RFO). The FO was calculated by dividing the number of digestive tracts containing a particular prey type by the total number of digestive tracts. To assess the importance of each prey type relative to other types, the RFO was calculated by dividing the number of each prey item by the total number of all prey items. We did not determine the index of relative importance because we lacked data on volumetric percentage (Hart et al. 2002). For birds, small mammals, and invertebrates, we conducted a χ^2 test on the total number of each prey type between Mauna Loa and Kīlauea. For the same prey types, we determined post-hoc 95% confidence intervals (CI) for FO and RFO based on each sample.

d. Results

The most frequent prey types found in feral cat digestive tracts were small mammals (88.9% of digestive tracts), and invertebrates (54.2%) from Mauna Loa and Kīlauea, respectively (Table 3.1). Bird remains were found in 27.8 and 29.2% of digestive tracts. More small mammals occurred in samples from Mauna Loa than Kīlauea. Of the cats from Mauna Loa, 89% (16 of 18) consumed small mammals whereas only 50% (12 of 24) from Kīlauea consumed small mammals. Rodents were the primary small mammal prey consumed on Mauna

Table 3.1. Sample size, number of prey, mean minimum number (and SE), frequency of occurrence (with binomial SE), and relative frequency of occurrence of prey items in feral cat digestive tracts from Kīlauea and Mauna Loa in Hawai`i Volcanoes National Park, 2000–2005.

Area	Sample <i>n</i>	Prey <i>n</i>	Mean Minimum Number (SE)	Frequency of Occurrence (SE)	Relative Frequency
Kīlauea	24				
Total Invertebrates		55	2.292 (0.721)	0.542 (0.102)	0.724
Total Birds		7	0.292 (0.095)	0.292 (0.093)	0.092
Procellariiformes		0	--	--	--
Total Small Mammals		14	0.583 (0.133)	0.500 (0.102)	0.184
<i>Mus musculus</i>		4	0.167 (0.078)	0.167 (0.076)	0.053
<i>Rattus</i> spp.		1	0.042 (0.042)	0.042 (0.041)	0.013
Total Scincidae		0	--	--	--
Mauna Loa	18				
Total Invertebrates		25	1.389 (0.677)	0.389 (0.115)	0.439
Total Birds		5	0.278 (0.109)	0.278 (0.116)	0.088
Procellariiformes		1	0.056 (0.056)	0.056 (0.054)	0.018
Total Small Mammals		21	1.167 (0.146)	0.889 (0.074)	0.368
<i>Mus musculus</i>		11	0.611 (0.118)	0.611 (0.115)	0.193
<i>Rattus</i> spp.		3	0.167 (0.090)	0.167(0.088)	0.053
Total Scincidae		6	0.333 (0.333)	0.056 (0.054)	0.105

Loa. Mice (*Mus musculus*) constituted the highest FO in digestive tracts from Mauna Loa (61.1%; Table 3.1) and the second highest RFO (19.3%) from Mauna Loa after Diptera (flies; 28.07%; Table 3.2). Conversely, only 16.7% of digestive tracts from Kīlauea contained mouse remains. Similarly, rats (*Rattus* spp.) occurred in 16.7% of digestive tracts from Mauna Loa and only 4.2% from Kīlauea. One stomach from Mauna Loa contained the remains of both one mouse and one rat.

Invertebrates occurred in 54.2% of digestive tracts from Kīlauea, and 38.9% of digestive tracts from Mauna Loa (Table 3.1). Common invertebrate remains included thoraxes, head capsules, legs, and wings. Orthoptera (crickets, grasshoppers) ranging from 3.0–3.5 cm in length were the most common invertebrate prey item in cat digestive tracts from Kīlauea (27.6% RFO) followed by Diptera, Chilopoda (centipedes) and Hymenoptera (wasps, bees) (Table 3.2). Conversely, Diptera, primarily larvae, were the most frequent invertebrate prey item from Mauna Loa (28.07%) followed by Hymenoptera and Gastropoda (snails). Centipedes were not found in digestive tracts from Mauna Loa.

Most bird prey from both areas was not identifiable to order. Only one passerine, identified by its feet and feathers, was found in a stomach from Kīlauea, comprising of 1.3% of RFO (Table 3.2). One stomach from Mauna Loa, however, contained remains of the procellariid seabird, `Ua`u, identified by a complete mandible set, feathers, and bones (Figure 3.2). Another stomach contained unidentified eggshell fragments. Other vertebrate prey included a sample from Mauna Loa with 6 skinks (Scincidae) comprising 10.5% of RFO for this area. The high RFO value reflects the fact that a single cat fed extensively on skinks while this prey item was found in no other samples from either location.

We found a significant difference between the two areas in the total number of bird, small mammal, and invertebrate prey in the digestive tracts of feral cats ($\chi^2 = 8.39$, $DF = 2$, $P = 0.015$). The 95% CI for FO were not exclusive for birds between Mauna Loa (0.010, 0.535) and Kīlauea (0.055, 0.405) or invertebrates (respectively 0.097, 0.555; 0.394, 0.971), but small mammals were exclusively greater for Mauna Loa (0.659, 1.259) relative to Kīlauea (0.171, 0.530) (Figure 3.3; $DF = 29$, $t = -3.65$, $P < 0.001$). The 95% CI for RFO were also not exclusive for birds between Mauna Loa (-0.001, 0.330) and Kīlauea (0.028, 0.246) or invertebrates (respectively 0.053, 0.371; 0.263, 0.642), but small mammals were exclusively greater for Mauna Loa (0.384, 0.789) relative to Kīlauea (0.088, 0.316) ($DF = 28$, $t = -3.47$, $P < 0.002$).

Table 3.2. Relative frequency of occurrence of prey items in digestive tracts of feral cats from Kīlauea and Mauna Loa in Hawai`i Volcanoes National Park, 2000–2005.

Prey ^a	Area	
	Kīlauea (<i>n</i> =24)	Mauna Loa (<i>n</i> =18)
Unidentified birds	0.079	0.053
Passeriformes	0.013	--
Procellariiformes	--	0.018
Unidentified mammals	0.118	0.123
<i>Mus musculus</i>	0.053	0.193
<i>Rattus</i> spp.	0.013	0.053
Unidentified invertebrates	0.263	--
Chilopoda	0.026	--
Orthoptera	0.276	0.018
Diptera	0.092	0.281
Hymenoptera	0.026	0.088
Lepidoptera	0.013	--
Dermaptera	0.013	--
Gastropoda	0.013	0.035
Unidentified Scincidae	--	0.105

^aComposite categories include only unidentifiable items. Component taxa are listed separately.



Figure 3.2. The remains of an `Ua`u (Hawaiian Petrel; *Pterodroma sandwichensis*) recovered from the digestive tract of a feral cat from Mauna Loa in Hawai`i Volcanoes National Park. Scale is in cm.

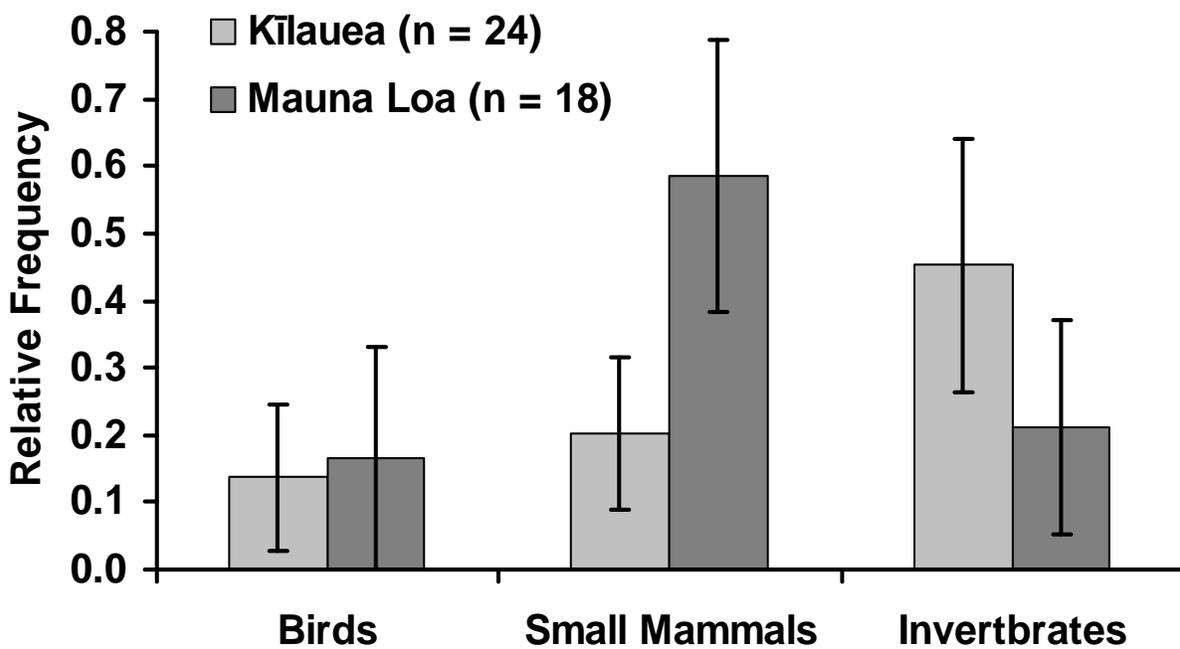
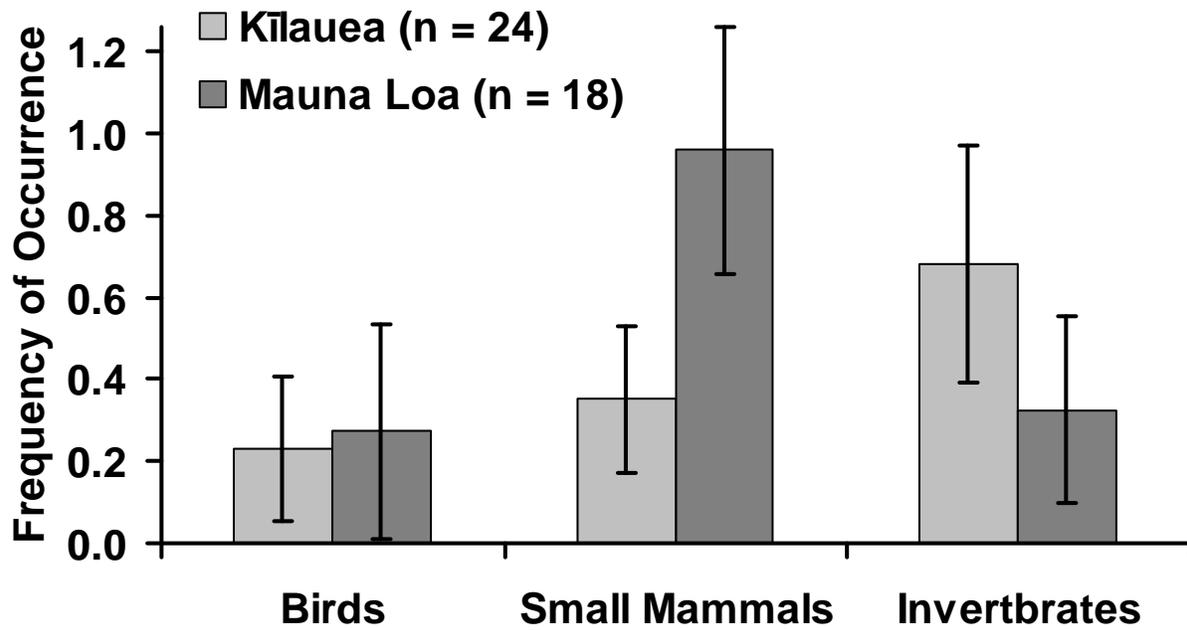


Figure 3.3. Frequency of occurrence (FO) and relative frequency of occurrence (RFO) of birds, small mammals, and invertebrates \pm 95% CI from Kīlauea and Mauna Loa in Hawai`i Volcanoes National Park, 2000–2005.

e. Discussion

We found several key differences in the diet of feral cats between Kīlauea and Mauna Loa volcanoes. The most important prey types from Kīlauea were invertebrates, primarily Orthoptera, while small mammals, primarily mice, were the most important prey types from Mauna Loa. Invertebrate prey occurred more frequently than small mammals on Kīlauea, but on Mauna Loa, where small mammal prey occurred substantially more frequently than invertebrates, perhaps reflecting differences in prey availability between these areas. Birds were less commonly found than other prey types in both locations.

Other researchers have similarly found that birds were relatively rare in the diet of feral cats (Jones and Coman 1981; Molsher et al. 1999; Pontier et al. 2002). Unlike Mauna Kea, where gallinaceous birds are abundant and frequently found in the diet of feral cats (Hess et al. 2004), they are much less common in HAVO (Scott et al. 1986), and their absence from HAVO diet was not surprising.

Invertebrates, particularly Orthoptera, may be an important food source on Kīlauea due to a lack of abundant rodents. With the exception of Mostello (1996), who found arthropods to be the second most important prey type after mice, little significance has been previously attributed to invertebrates in the diet of feral cats in Hawai`i. Although Smucker et al. (2000) found invertebrates in > 47% of scats from Mauna Kea and Kaho`olawe, they stated that introduced rodents may be sustaining predator populations; however, the importance of invertebrates should not be summarily discounted. Snetsinger et al. (1994) found insect remains in 17% of scats from Mauna Kea, and Amarasekare (1994) reported Coleoptera, Orthoptera, and Hymenoptera in scats from Mauna Kea. Studies that rely on scat to quantify diet may systematically underestimate some small prey items composed primarily of soft tissues, such as invertebrates. Some insects, however, (e.g., Hymenoptera, Diptera, Dermaptera) may be attracted to bait and may bias the number of invertebrates found in samples, as evidenced by the Diptera larvae we frequently found.

Dickman (1996), found that invertebrates featured much more prominently in the diets of cats from Pacific region islands ($n = 12$ studies) than from a range of habitats in mainland Australia ($n = 22$ studies). Orthoptera were among the most common invertebrate prey of feral cats in the Pacific region. Mostello (1996) found the katydid *Euconocephalus nasutus* to be the most important invertebrate from scats of Hawaiian cats. Kirkpatrick and Rauzon (1986) found crickets in the stomach contents of cats from Jarvis Island. Jones and Coman (1981) reported

that grasshoppers were among the most frequently eaten groups of invertebrates in southeastern Australia. Molsher (1999) also reported that Orthoptera were the dominant group of invertebrates consumed by cats in New South Wales, Australia, particularly in summer and autumn.

The potential implications of feral cats for native invertebrates may be minimal. Other small mammals, such as early established populations of *R. exulans*, as well as currently abundant *R. rattus*, *R. norvegicus*, mice, and even *Herpestes auropunctatus* (small Indian mongoose) (Kami 1964; Tomich 1986) may have been and continue to be far more important predators of native invertebrates than relatively uncommon feral cats (Atkinson 1977; Stone 1985). Furthermore, the majority of invertebrates found in cat stomachs, particularly Orthoptera, are non-native species (Nishida 2002).

Abundant invertebrate prey, although much smaller than vertebrates, may play an important role in sustaining cats in seasons and environments where other types of prey may not be as common, such as Kīlauea Volcano. Abundant small prey may allow feral cats to survive food shortages, maintain populations at higher densities than they would otherwise be able to maintain, and then exploit seasonally abundant prey such as nesting birds. In the process known as hyperpredation, a prey species well adapted to high predation pressure may serve to maintain large populations of predators which may then drive more vulnerable prey to extinction, such as island endemic birds (Courchamp et al. 2000).

ʻUʻau are highly vulnerable to predation due to their burrow-nesting behavior. The ʻUʻau beak represents the first well-documented endangered species to be recovered from a feral cat in Hawaiʻi, although feathers thought to be from a petrel were found in a digestive tract of a feral cat on Mauna Loa in 1994 (D. Hu, pers. comm.). These findings suggest that feral cats prey on this rare seabird nesting at high elevation (approximately 2,700 m). ʻUaʻu feathers have also recently been recovered from digestive tracts of feral cats in the vicinity of an ʻUaʻu breeding colony on the island of Lānaʻi (J. Penniman, pers. comm.).

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III) Evidence of Feline Immunodeficiency Virus, Feline Leukemia Virus, and *Toxoplasma gondii* in Feral Cats on Mauna Kea, Hawai`i

Raymond M. Danner¹, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Daniel M. Goltz², Pacific Cooperative Studies Unit (University of Hawai`i at Mānoa), c/o U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI 96718, USA

Steven C. Hess, Paul C. Banko, U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI 96718, USA

¹ Current address: Department of Biological Sciences, Virginia Polytechnic Institute and State University, Blacksburg, VA, 24060, USA

² Current address: Wisconsin Department of Natural Resources, Wilson Nursery, 5350 State Road 133, Boscobel, WI 53805, USA

a. Narrative

Infectious diseases of introduced predatory mammals are important for at least two reasons: first, diseases may limit populations, thereby reducing the risk of predation on native wildlife. Second, some infectious agents may be transmitted to native wildlife, domestic pets, or humans. Feral cats (*Felis catus*) in Hawai`i may carry and transmit feline immunodeficiency virus (FIV), feline leukemia virus (FeLV), and the coccidian protozoan *Toxoplasma gondii*. Feline immunodeficiency virus and FeLV have been evaluated as potential biological control agents in island ecosystems (Courchamp and Sugihara 1999), and *T. gondii* is known to cause illness in several Hawaiian bird species (Work et al. 2000; 2002), the endangered Hawaiian monk seal (*Monachus schauinslandi*) (Honnold et al. 2005), and humans (Dubey and Beattie 1988).

Although the pathogenesis of FIV and FeLV in cats has been well studied, and some impacts of *T. gondii* on wildlife are known, seroprevalence of infectious agents in wild felids in remote natural areas of Hawai`i is poorly understood. Our objective was to assess three infectious agents in feral cats on Mauna Kea, Hawai`i. We examined FIV and FeLV to determine whether these potential biological control agents already existed in feral cats, and we examined

toxoplasmosis prevalence to determine whether a risk of transmission exists for native wildlife, particularly for endangered bird species on Mauna Kea, Hawai`i.

We captured feral cats on the north and west slopes of Mauna Kea (19° 50'N, 155° 35'W) from 9 April 2002 to 16 May 2004 as part of efforts to protect and restore an endangered Hawaiian forest bird. Tomahawk® model 106 live traps (23 x 23 x 85 cm) were distributed at 150-m intervals along 15 transects in subalpine woodland from 1,750 to 3,000 m in elevation. We covered traps with a layer of plastic to protect trapped cats from rain and cold, and we placed a cloth rag inside for bedding. We used canned cat food, sardines, and mackerel as bait. We checked set traps daily, and we wired traps open when unattended. Feral cats were euthanized by gunshot to the head according to University of Hawai`i IACUC protocol.

We determined sex, and we determined age by the presence of complete adult dentition. We also collected blood samples. A 21-gauge hypodermic needle was used to draw 10 ml of blood by cardiac puncture postmortem. Snap™ Combo FeLV Antigen/FIV antibody enzyme-linked immunosorbent assays (ELISAs) (IDEXX Laboratories, Inc., Portland, Maine, USA), were used in the field with fresh whole blood according to the manufacturer's instructions. These tests are portable kits that provide results in 10 min when a conjugate is added to whole blood. Remaining whole fresh blood was placed in 2-ml serum collection vials, stored on ice, and centrifuged, and plasma/serum was aliquoted, frozen (-10 C°), and sent to Colorado Veterinary Diagnostic Laboratories (Colorado State University, Fort Collins, Colorado, USA) for analysis of antibodies to *T. gondii*. Immunoglobulin G (IgG) and immunoglobulin M (IgM) ELISAs confirmed past exposure or recent infection with *T. gondii*, respectively, at threshold titers of 1:64 (Lappin et al. 1989).

We took blood samples from 50 feral cats on the west slope and from 21 cats on the north slope of Mauna Kea. We measured exposure/infection to FIV/FeLV in 68 cats and *T. gondii* in 67 cats, with 64 cats assayed for all three agents. Six and 11 of 68 cats were serologically or antigen positive to FIV or FeLV, respectively (Table 4.1). FIV occurred only in adult males. Twenty-five of 67 cats were seropositive to toxoplasmosis. Of 64 cats tested for all three agents, four cats were positive to *T. gondii* and FIV, three cats to *T. gondii* and FeLV, and one cat was positive for both viruses.

Feral cats on Mauna Kea lead a solitary existence, frequently roaming over great distances far from human habitation (Tomich 1986). In contrast to urban animals leading more colonial lifestyles, Mauna Kea cats probably do not have as much conspecific contact, thereby

limiting opportunities for horizontal disease transmission. The presence of FIV only in adult males, however, is consistent with the suspected primary mode of transmission, which is biting (Yamamoto et al. 1988). We failed to detect FIV in eight adult males among 21 cats from the north slope, but this result is probably due to inadequate sample size rather than a lack of geographic mixing. We suspect mixing is not a problem because a male cat with a radio collar repeatedly traveled 25 km between the west and northeast slopes, and the genetic structure of feral cats suggests substantial gene flow between Mauna Kea and Mauna Loa populations, which are > 50 km apart (USGS-BRD, unpubl. data).

Table 4.1. Prevalence of feline immunodeficiency (FIV) antibodies, feline leukemia (FeLV) antigen, and *Toxoplasma gondii*-specific immunoglobulin G (IgG) and immunoglobulin M (IgM) antibodies in feral cats on Mauna Kea, Hawai`i, 2002–2004 (percentage with number of age/sex class sample in parentheses).

	FIV	FeLV	<i>T. gondii</i>		
			IgG	IgM	IgG & IgM
Adult					
Male	16.7 (6/36)	16.7 (6/36)	28.6 (10/35)	0 (0/35)	2.9 (1/35)
Female	0 (0/23)	13.0 (3/23)	27.3 (6/22)	9.1 (2/22)	9.1 (2/22)
Juvenile					
Male	0 (0/3)	33.3 (1/3)	66.7 (2/3)	0 (0/3)	0 (0/3)
Female	0 (0/6)	16.7 (1/6)	28.6 (2/7)	0 (0/7)	0 (0/7)
Overall	8.8 (6/68)	16.2 (11/68)	29.9 (20/67)	3.0 (2/67)	4.5 (3/67)

FeLV may be transmitted through contaminated saliva, blood, or other body fluids (Maruyama et al. 2003), but the virus is extremely labile, surviving only 24–48 hr in a moist environment at room temperature (Cotter 1998). Therefore, close contact is required for transmission. Kittens also may be infected transplacentally, through nursing, or by licking (Cotter 1998). The cat flea (*Ctenocephalides felis*) is also a potential vector of FeLV (Vobis et al. 2003), but it has yet to be recovered from cats on Mauna Kea, possibly due to the cool, dry

climate. These modes may explain how the virus is maintained in a population that lacks a colonial social structure.

Cats are the definitive host of *T. gondii* (Wallace 1973), and toxoplasmosis has been reported in Hawai`i since the 1950s (Tilden 1953). Toxoplasmosis has caused mortality of native Hawaiian birds such as captive Nēnē (*Branta sandvicensis*), wild Red-footed Booby (*Sula sula*) (Work et al. 2002), and critically endangered `Alalā (*Corvus hawaiiensis*) (Work et al. 2000). Moreover, *T. gondii* oocysts may enter marine environments in municipal sewage or storm water runoff, sporulate in seawater (Lindsay et al. 2003) and thereby infect a variety of marine mammals, including dolphins (Migaki et al. 1990) and seals (Holshuh et al. 1985).

Vertical transmission of *T. gondii* to offspring can occur transplacentally or when kittens consume infected milk (Kenny et al. 2002), but cats typically become infected by eating raw meat, birds, or rodents containing cysts (Acha and Szyfres 1980). Humans and wildlife primarily develop toxoplasmosis after ingesting sporulated oocysts shed in cat feces (Dubey and Beattie 1988). Work et al. (2000) suspected that free-ranging `Alalā may have contracted fatal toxoplasmosis by ingesting *T. gondii* oocysts from infected feral cat feces, tissue cysts from transport hosts, or from invertebrates that ingested oocysts. Other ground-feeding species such as the endangered Nēnē and the introduced Erckel's Francolin (*Francolinus erckelii*) may become infected after consuming forage contaminated with sporulated oocysts (Work et al. 2002).

The degree to which feline diseases limit or regulate cat populations in the wild is not well understood. FeLV may strongly limit populations (Courchamp and Sugihara 1999). Cat populations were depressed by 7–20% in a population model with only 4.3–12.4% FeLV prevalence (Fromont et al. 1997). The cat population on Mauna Kea may be depressed by feline diseases, but their presence suggests they would not be effective agents in a biological control strategy. Feral cats remain abundant throughout most of the Hawaiian Islands (Winter 2003), and they are difficult to control in the remote wilderness.

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IV) Home Range and Movements of Feral Cats on Mauna Kea, Hawai`i

Daniel Goltz¹, Pacific Cooperative Studies Unit (University of Hawai`i at Mānoa), U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Steve Hess, U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Kevin Brink, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Paul Banko, U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

Raymond Danner², Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

¹ Current address: Wisconsin Department of Natural Resources, Wilson Nursery, 5350 State Road 133, Boscobel, WI 53805, USA

² Current address: Department of Biological Sciences, Virginia Polytechnic Institute and State University, 2125 Derring Hall, Blacksburg, VA 24061, USA

a. Introduction

Feral cats (*Felis catus*) became abundant in forests of the Hawaiian Islands soon after their introduction in the late 1700s (King 1984; Rothschild 1893; Perkins 1903). They were subsequently reported to be notorious predators of birds that contributed to the decline and extinction of some Hawaiian species (Perkins 1903; Berger 1981; Ralph and van Riper 1985; Stone 1985; Snetsinger et al. 1994). In Hawai`i, cats are currently important predators of terrestrial birds that nest near the ground (Kowalsky et al. 2002), and in trees (Hess et al. 2004). Cats are also important predators of colonial seabirds (Smith et al. 2002). Feral cats range throughout the Hawaiian Islands from high-density colonies near coastal areas where pets are frequently abandoned and fed by people (Winter 2003), to remote, low-density populations in montane forests and subalpine areas of Maui (Simons 1983) and Hawai`i Island

(Hu et al. 2001). Despite the long history of feral cats in Hawai`i, there has been little research about their spatial arrangement or basic ecological organization, such as home range and movements.

Several studies have documented home range and movements in the Pacific region where introduced cats have established feral populations. Fitzgerald and Karl (1986) reported linear home ranges in the steep Orongorongo Valley of New Zealand. Konecny (1987) found the home ranges of male cats to be larger than those of females in the Gálapagos Islands; however, Norbury et al. (1998) found no difference in home range size between sexes for feral cats living in dry tussock grassland in New Zealand. Edwards et al. (2001) documented the largest home ranges (2,210.5 ha) in semiarid woodland of central Australia. The only study of feral cat home ranges in Hawai`i was from a wet montane forest on windward Mauna Kea, at Hakalau Forest National Wildlife Refuge (Hakalau), Hawai`i Island (Smucker et al. 2000). These studies have contributed to the basic knowledge of the species, but have also provided information for better management of non-native predators and conservation of native fauna (Fitzgerald and Karl 1986).

The distribution and abundance of feral cats may be controlled by a number of factors including territorial behavior, social interactions, or food resources. Marked differences between individuals may exist in landscape use patterns due to foraging, mate-seeking, denning, and rearing behaviors. The spatial arrangement of feral cats can be used to gauge the timing and spacing distance of control units (e.g., traps or poisoned baits), to understand the epidemiology of diseases, and to delineate the total area over which feral cats need to be controlled in order to remove resident animals and confine immigration to buffer zones on the perimeter of core conservation areas (Veitch 1985; Norbury 1998; Short et al. 1997; Edwards et al. 2001). Our objectives were to determine home range, territoriality, and daily movement rates of feral cats in the dry subalpine woodland of Mauna Kea, Hawai`i, Island as part of a larger study of feral cats in Hawai`i that included diseases (Danner et al. 2007), diet (Hess et al. 2007), and population genetics (Hansen et al., 2007). The overall goal of this research was to provide basic information on the impacts of feral cats on native wildlife as well as strategic considerations for developing and improving control methodology.

b. Study Area

Study area was located in subalpine woodland on Hawai`i Island on the West Slope of Mauna Kea (19° 49' N, 155° 36' W), a dormant volcano. Subalpine woodland occurs between

1,750–3,000 m elevation. Overall canopy cover averaged 30% and canopy height was generally short (3–8 m) with interspersed lower-stature shrubs and larger areas of shrublands. Dominant trees include māmane (*Sophora chrysophylla*) and naio (*Myoporum sandwicense*) and extensive shrublands are dominated by pūkiawe (*Leptecophylla tameiameia*), and `a`ali`i (*Dodonaea viscosa*). The semi-arid aspect of subalpine woodland is due to severely drained volcanic substrates and rain shadow effects. Rainfall averaged 511 mm and temperature averaged $11.1^{\circ} \pm 1.5^{\circ}$ C annually (Juvik et al. 1993). There are no natural sources of perennial standing water in the study area.

c. Methods

Since feral cats are difficult to observe, we calibrated 35–37g Holohil Systems Ltd. model MI-2 radio transmitters with AOR AR8200 digital receivers to determine location accuracy during close-range tracking. We simulated cat behavior by handling transmitter collars both in motion and motionless at < 0.5 m height through vegetation. Observers that did not know the true location of transmitters monitored the receiver's LCD signal meter and achieved approximately 50 m accuracy. We maintained this distance during tracking to avoid disturbing the subjects.

We captured seven male and three female cats, anaesthetized them with methoxyfluorane, and fitted them with transmitters. Cats were allowed to fully recover in traps and were not tracked for ≥ 1 d after release. Six male cats were fitted with transmitters in July 1999. One cat (M-4) was opportunistically recaptured after 14 mo and fitted with a new transmitter to extend data collection. Three females and one additional male cat were fitted with transmitters in September 2000. The weight of males ranged from 2.4–3.65 kg ($\bar{X}=2.99$ kg) while females ranged from 1.75–2.2 kg ($\bar{X}=1.86$ kg). All cats were adults, based on dentition. We recorded ≤ 3 locations per cat per day with Garmin GPS 12 (Lenexa, Kansas, USA) global positioning system receivers. Cats were periodically tracked on consecutive days to determine daily movements. Data collection ended in February 2002.

To investigate the temporal autocorrelation of observations, we calculated $t2/r2$ statistics for a range of minimum times between observations (Swihart and Slade 1985; 1986). A minimum separation of 2.85 d was needed to achieve quasi-independent observations. We excluded data points that were taken ≤ 2.85 d after the first observation, and points that were ≤ 2.85 d after subsequent observations. We calculated 95%, 50%, and 25% fixed kernel (Worton 1995) home ranges using the animal movement extension (Hooge and Eichenlaub 1997) for ArcView GIS (ESRI 1999). We used least squares cross-validation to estimate a kernel

smoothing parameter for each cat, and used the median value (378 m) for all cats to produce the final home range estimate as recommended by Seaman and Powell (1996). We then examined the effect of reduced sample size on kernel home range estimates with 1,000 bootstrap minimum convex polygons (MCP) from the remaining locations using the animal movement extension (Hooge and Eichenlaub 1997). We plotted MCP area against sample size to determine if sufficient observations existed to stabilize MCP area. We compared log-transformed 95% and 25% kernel home range estimates between sexes and the simple mean rate of speed (m/d) between all successive observations with *t*-tests. We also compared log-transformed home range estimates between sexes with seven other published studies from the Pacific region.

d. Results

Fixed kernel home range estimates were determined to be reliable for seven of the 10 subjects through bootstrap minimum convex polygon analysis. Estimates of 95% fixed kernel home ranges for these seven cats ranged from 610–2,050 ha and averaged 1,418 ha for males and 772 ha for females (Table 5.1). Excluding points to achieve quasi-independence resulted in an average increase of 28% in the 50% core activity areas, although overall 95% kernel home ranges were smaller when compared to analyses using all observations. Log-transformed 95% kernel home ranges for males were significantly larger than those of females (Equal variance 2-sample *t*-test; $DF = 5$, $t = -3.20$, $P = 0.024$), but 25% kernel home ranges of females were larger than those of males ($DF = 5$, $t = 3.53$, $P = 0.017$; Figs 5.1–5.2). Female F-3 raised two litters of kittens during the study and had the smallest home range and daily movement rates. F-2 had two core 25% activity areas and males M-2, M-4, and M-5 had multiple 50% activity areas. M-1 exhibited sequentially clustered observations separated by 14.8 km and was therefore treated as having two separate home ranges, although sample size was insufficient for reliable kernel estimates for either home range. M-3 periodically travelled from the West to the North Slope of Mauna Kea, making a 45 km roundtrip in a 2-week period, but had insufficient sample size for reliable kernel estimate. Mean daily movement rates between sexes overlapped widely and did not differ significantly (Unequal variance 2-sample *t*-test; $DF = 6$, $t = -2.08$, $P = 0.083$). Log-transformed home ranges for males were significantly larger than those of females in this and seven other studies from the Pacific region ($DF = 12$, $t = -2.25$, $P = 0.044$; Table 5.2).

Table 5.1. Fixed kernel home range estimates (ha) and mean daily movement rates of feral cats (*Felis catus*) on the West Slope of Mauna Kea, Hawai`i, 1998–2001. Non-independent observations were eliminated based on estimated average of 2.85 d to quasi-independence. Home range was calculated by the median least square cross validation (LSCV) smoothing parameter (H) value of 378.

ID	Days to Quasi-Independence	Observations		LSCV H	Home Range (ha)			Mean Distance (m/day)
		<i>n</i> (all)	<i>n</i> (2.85)		95%	50%	25%	
M-1	0.25	26	9 ^a	600	637	55	24	--
M-1	0.05	30	8 ^a	298	428	60	26	4521.5
M-2	0.04	66	29	416	1167	98	37	6014.3
M-3	4.00	70	18 ^a	399	739	80	29	4901.4
M-4	6.65	179	43	542	2050	152	29	153.7
M-5	0.05	112	46	358	1279	116	32	109.8
M-6	1.00	19	6 ^a	386	432	80	24	90.9
M-7	8.00	73	30	428	1176	84	32	124.9
F-1	4.95	135	31	291	875	169	61	108.6
F-2	1.70	130	27	292	831	202	58	112.4
F-3	6.90	183	44	195	610	100	39	183.7

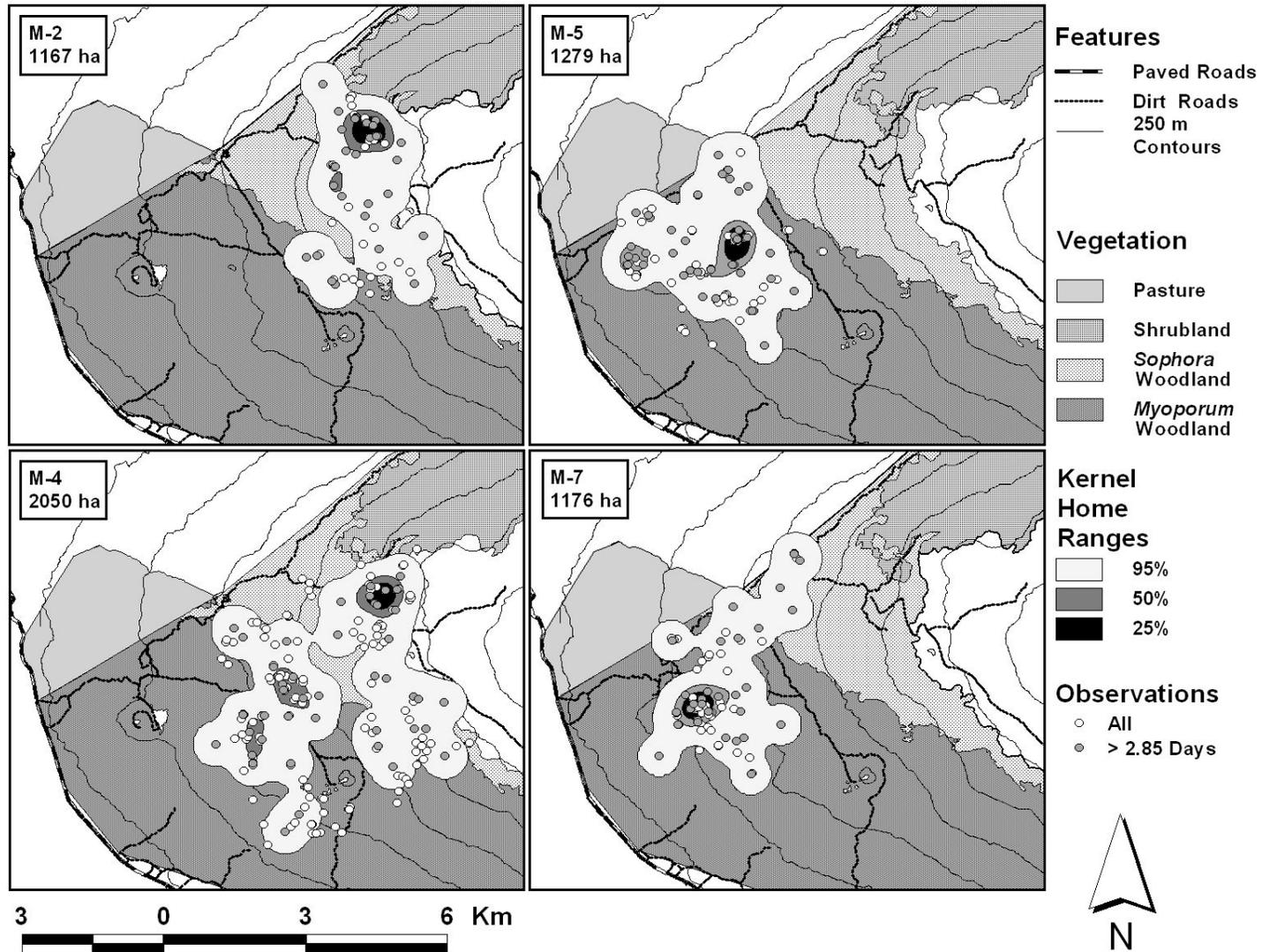


Figure 5.1. Telemetry observations and fixed kernel home ranges of male feral cats on the West Slope of Mauna Kea, island of Hawai`i in 1998–2001. Non-independent observations were eliminated based on estimated average of 2.85 d to quasi-independence.

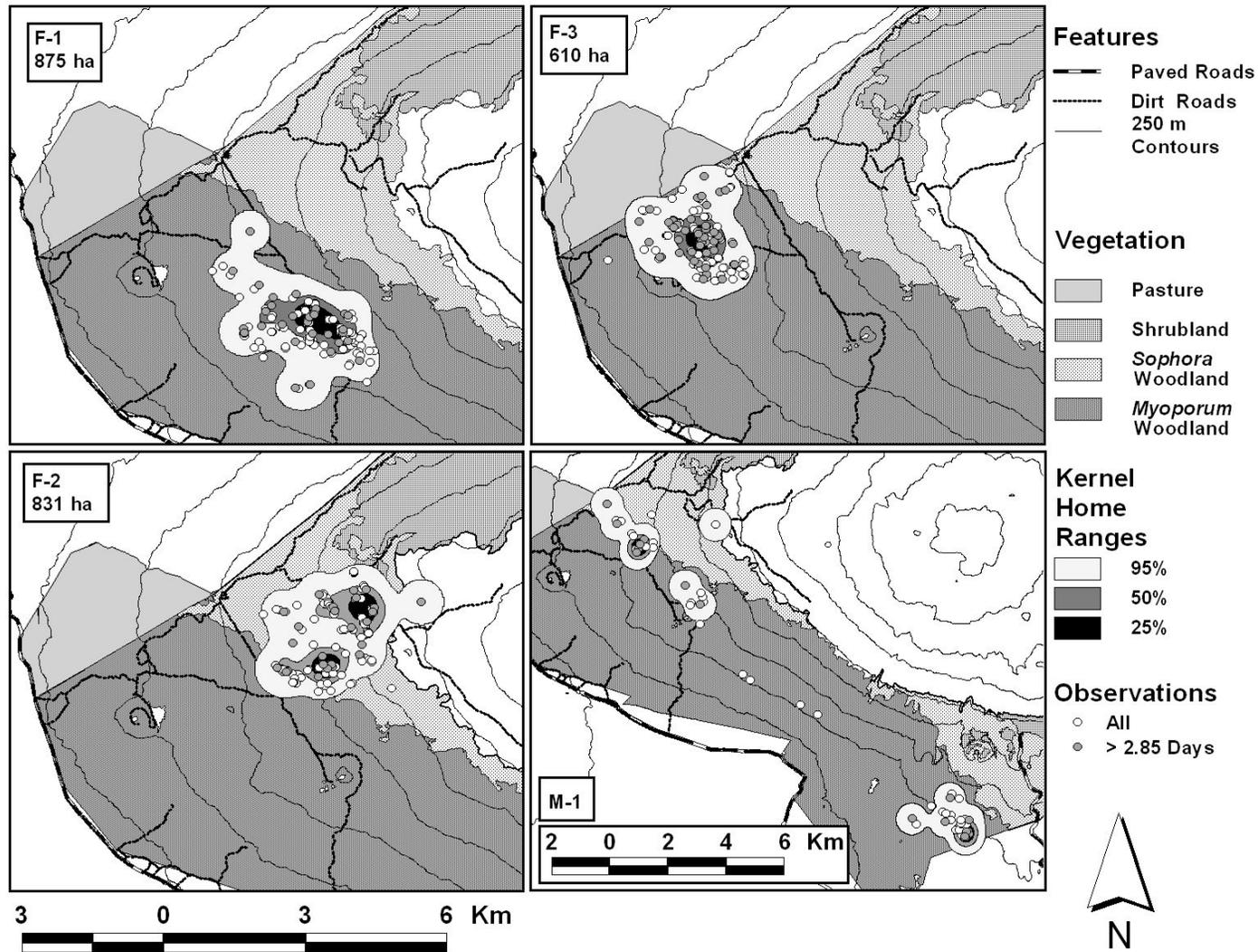


Figure 5.2. Telemetry observations and fixed kernel home ranges of three female and one male feral cat on the West Slope of Mauna Kea, island of Hawai'i 1998–2001. Non-independent observations were eliminated based on estimated average of 2.85 d to quasi-independence.

Table 5.2. Comparison of home range estimates from the West Slope of Mauna Kea, Hawai`i, with seven other published studies from the Pacific region. Percent by which Mauna Kea home range was > another study area = $(1 - (HR_{study\ x}/HR_{Mauna\ Kea})) * 100$.

Habitat and Location	Home Range		Percent Mauna	
	Size (ha)		Kea > by	
	F	M	F	M
Subalpine woodland, Mauna Kea, Hawai`i	772	1418	--	--
Victorian Mallee, South-eastern Australia ^a	170	620	78.0%	56.3%
Orongorongo Valley, North Island, NZ ^b	80	140	89.6%	90.1%
Galápagos Islands ^c	82	304	89.4%	78.6%
Dry tussock grassland, South Island, NZ ^d	225	225	70.9%	84.1%
Open forest, New South Wales, Australia ^e	140	288	81.9%	79.7%
Wet montane forest, Hakalau, Hawai`i ^f	223	574	71.1%	59.5%
Semiarid woodland, Central Australia ^g	--	2211	--	-55.9%
Mean	241.7	722.4	80.1%	56.1%

^aJones and Coman (1982)

^bFitzgerald and Karl (1986)

^cKonecny (1987)

^dNorbury et al. (1998)

^eMolsher et al. (2005)

^fSmucker et al. (2000)

^gEdwards et al. (2000)

e. Discussion

Mean home ranges of feral cats on Mauna Kea were the largest reported among seven other studies from the Pacific region for females (\bar{X} = 79.9% larger) and the second largest for males (\bar{X} = 56.1% larger; Table 5.2). We found that home ranges of male feral cats on Mauna Kea were 60% larger, and females were 71% larger than those reported from Hakalau, which is approximately 25 km in distance from our study site (Smucker et al. 2000). Moreover, home

ranges of males were also significantly larger than those of females throughout the Pacific region.

Although Konecny (1987), Jones and Coman (1982), and Norbury et al. (1998) used minimum convex polygon analyses, the larger home ranges on Mauna Kea represents more than a methodological discrepancy. Home ranges were comparably sized in the environment most similar to Mauna Kea; semiarid woodland of the Northern Territory of Australia (Edwards et al. 2001). Edwards et al. (2001) and Smucker et al. (2000) also based their estimates on kernel methodology, but our estimates of overall home ranges may be comparatively smaller in area because we eliminated non-independent points. No other studies explicitly accounted for the independence of locations; however, Norbury et al. (1998) examined the effect of sample size on home range estimates and found that home range size began to stabilize after about 10 locations. We found that subjects with ≤ 18 locations did not provide reliable home range estimates. Only two of five subjects in the Smucker et al. (2000) study had > 18 locations, therefore sample sizes may have been inadequate, resulting in underestimation of true home range size.

We found some evidence that the spatial arrangement and low density of feral cats on Mauna Kea was tied to social organization. There were strong differences between sexes in home range size, which we attributed primarily to mate-seeking behavior in males, and denning behavior in females. Although females on Mauna Kea occupied large areas relative to other study locations, their overall mean home ranges were only 54% as large as males, but their core activity areas were larger than those of males indicating they may use smaller areas more intensively while males use larger areas more extensively. Because mean daily movements on Mauna Kea did not differ between sexes, this indicates that males did not always return to the same location on a daily basis, whereas females may have been tied to a central location. Konecny (1987) also found differences between sexes and that female cats with kittens occupied extremely restricted ranges. Norbury et al. (1998) and Molsher (2005), however, found no difference between sexes in home range for cats in dry tussock grassland in New Zealand and central-western New South Wales, Australia, respectively. The relative sizes of male and female home ranges may differ among these study environments due to the breeding frequency of females and the mate-seeking and foraging behaviors of males in different habitats.

In studies where food sources were abundant and concentrated, a large degree of spatial overlap occurred between male cat home ranges, but activity centers were discrete and encounters with conspecifics were rare (Konecny 1987; Short et al. 1997). Smucker et al. (2000) reported that male home ranges had minimal overlap in montane forest at Hakalau. In subalpine Mauna Kea, however, male cat home ranges overlapped extensively, including the 25% core activity areas of M-2 and M-4. There was no apparent seasonal pattern to home range overlap; however, the range of M-4 extended to lower elevation primarily during March–August 2001, and subsequently returned to higher elevation. Evidence for territorial encounters between males on Mauna Kea comes from the prevalence of Feline Immunodeficiency Virus (FIV) which is primarily transmitted by biting and scratching (Yamamoto et al. 1988). While 17% of 39 males from Mauna Kea tested positive for FIV, all 29 females were negative (Danner et al. 2007). Overall female home ranges on Mauna Kea overlapped only slightly, and core activity areas did not overlap.

Feral cats are wide-ranging predators with negative effects on the native fauna throughout the Pacific region (Dickman 1996). Hansen et al. (2007) estimated that 17.6% of cats per generation on Mauna Loa, primarily males, had migrated > 53 km from Mauna Kea. There is evidence that feral cats in Hawai`i prey on endangered forest birds (Laut et al. 2003; Hess et al. 2004) as well as nesting seabirds (Smith et al. 2002; Hess et al. 2007). Male cats tend to be more wide-ranging than females and may therefore encounter more endangered birds while foraging. For example, the mean home range of male cats on Mauna Kea represents more than 10% of the entire range (140 km²) occupied by the endangered Hawaiian finch, the Palila (*Loxioides bailleui*) (Scott et al. 1986). Because male home ranges were also non-exclusive, Palila may be simultaneously exposed to predation by several different individual cats.

Trap spacing should account for minimum daily movements and differences between sexes in home range. The radius of a circular area equivalent to the home range of female feral cats on Mauna Kea is 1,568 m and males is 2,125 m. Cats may be expected to encounter traps at some point in time when traps are spaced less than these distances, however, daily movements may be used to dictate minimum spacing between transects during short-term (2–3 d) trapping sessions. Most subjects made daily movements > 100 m. The implications of large home ranges in male cats is that large trapping areas are needed to confine immigration to buffer zones on the perimeter of endangered species habitat in this environment in Hawai`i.

Other considerations are also important for effective trapping programs. Short et al. (2002) found that feral cats that did not use rubbish dumps were more likely to be caught in concealed foot-hold traps than cage traps. No such human subsidies currently exist within the range of cats on the West Slope of Mauna Kea.

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V) Design and Evaluation of a Simple Signaling Device for Live-Traps

Francis L. Benevides Jr., FB Engineering, P.O. Box 5023, Hilo, HI 96720, USA

Heidi Hansen¹, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey
Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i
National Park, HI, 96718, USA

Steven C. Hess, U.S. Geological Survey Pacific Island Ecosystems Research Center, P.O. Box 44,
Kīlauea Field Station, Hawai`i National Park, HI, 96718, USA

¹ Current address: Department of Land and Natural Resources Division of Forestry and Wildlife,
19 East Kawili St., Hilo, HI 96720, USA

a. Introduction

Frequent checks of live traps mandated by many research institutions and wildlife management agencies require enormous amounts of labor and expense. This limits the effective management of larger landscapes inhabited by vulnerable endangered species. Vertebrate pest control in remote areas can benefit from simple trap-signaling devices that minimize physical trap checks and improve the welfare of trapped animals by reducing response time (Johnston et al. 2005). Moreover, fewer checks may improve capture efficiency by reducing associated human scents that deter wary vertebrates from approaching traps (Marks 1996).

Previous designs for trap-signaling devices include a radio-alarm-equipped bownet with a mini magnetic-reed contact switch (Proudfoot and Jacobs 2001) and a radiotransmitter for corral traps with an alligator clip switch (Hayes 1982). Nolan et al. (1984) also described a radiotransmitter activated by a mechanical trip pin for monitoring snares. More sophisticated designs include a radiotelemetry system for treadle snares to monitor a large number of alarm units, each with a unique identity code (Marks 1996), and an electronic signaling system for leg-hold traps based on cellular telephone technology (Larkin et al. 2003).

Key features of effective trap-signaling devices are long-distance reception, durability in adverse weather and ease of transport, deployment and use. Another important feature is design simplicity which reduces the potential for failure. Our objectives were to design, develop and evaluate the reliability of a signaling device for live traps. We sought to improve previous

designs with a magnetic security switch, which has the potential to increase system reliability and long-distance reception by allowing transmitters and antennas to be mounted high above traps.

b. Study Area

Feral cats (*Felis catus*) and small Indian mongooses (*Herpestes auropunctatus*), which prey on the endangered Hawaiian birds Palila (*Loxioides bailleui*) and `Ua`u (Hawaiian Petrel; *Pterodroma sandwichensis*), were livetrapped in Mauna Kea Forest Reserve (MKFR) and on the south-eastern flank of Mauna Loa in Hawai`i Volcanoes National Park (HAVO), Hawai`i Island. The MKFR study area (19° 54' N, 155° 28' W) was located between 1,750–3,000 m elevation and characterized as dry subalpine woodland. Mean annual temperature and rainfall were 11.1° C and 511 mm respectively. Overall canopy cover averaged 30% and canopy height was generally short (3–8 m) with interspersed lower-stature shrubland. The HAVO study area (19° 29' N, 155° 25' W) extended from 2,000–2,700 m elevation. Mean annual temperature and rainfall were 10.5° C and 1,250 mm respectively. This area was primarily characterized as low stature (2–8 m) dry subalpine woodland and shrubland (Wagner et al. 1999). Vegetation cover was ≤ 10% at higher elevations and dominated by shrubs. The MKFR study area contained abundant feral pigs (*Sus scrofa*) and sheep (*Ovis* spp.), but the HAVO study area was free of ungulates. Small mammals (*Mus musculus* and *Rattus* spp.) were abundant at both sites. Substrates of both volcanoes were geologically young and interspersed with major lava flows.

c. Methods

We designed our trap signaling devices in part from components made by Merlin Systems Inc.; including a Very High Frequency (VHF) potted radiotransmitter powered by an external 3.6 volt Tadiran lithium battery. Both components came encased in a 3 x 6 cm weatherproof plastic housing with a screw detachable cover, along with a 40 cm rigid transmitting antenna extending out of the housing. Transmitters with unique frequencies in the range of 164–166 MHz had a pulse repetition rate of approximately 40 per min. We attached a 3 m long, 28-gauge, stranded 2-conductor insulated cable (Qualtron, Inc. 89Z0198) to the transmitter and battery components through the bottom of the housing. We routed the other end of the cable to a 3.5 x 5.5 cm weatherproof plastic box (Radio Shack 270-288). The box provided strain relief and protection for a splice between the transmitter cable and a pigtail end of a cable from a normally-open Sentrol 1045W surface mount industrial magnetic contact switch typically used for home and office security systems (Figure 6.1). One

end of the pigtail cable was potted to the magnetic switch by the manufacturer. The other end of the cable was connected to the positive terminal of the battery. Components were either factory sealed (e.g., radiotransmitter, magnetic switch) or made weatherproof (e.g., transmitter and battery housing, strain relief box) during assembly using a common silicon based sealant to ensure reliable operation in rain and dust conditions. The entire length of the cable was encased by camouflage pattern duct tape. We evaluated the cost per complete assembled unit.

The transmitter, battery and antenna were designed to be mounted high in a tree to facilitate long-distance line of sight (LOS) reception. The switch contact was mounted on the exterior of the trap and the magnet was mounted on the door such that the switch and magnet were aligned when the door was open, thus activating the transmitter. A sprung trap, bad battery, failed transmitter, defective antenna, or switch stuck in the open position, as well as any open circuit type of damage to the cable deactivated the transmitter. Our system's safeguard was similar to designs by Hayes (1982) and Nolan et al. (1984), whereby the absence of a signal from the device indicated that the trap was sprung or required maintenance.

Devices were attached to Tomahawk® model 106 live traps (23 x 23 x 85 cm) with cable ties. Transmitters were mounted to the highest point of nearby trees. We deployed 10 trap-signaling devices on the North Slope of the MKFR between November 2004 and December 2005. We set traps and activated trap-signaling devices at the beginning of 10-d trapping sessions, and monitored signals daily using a 3-element Yagi directional antenna. We also physically checked traps daily according to University of Hawai'i Animal Care and Use protocol. Similarly, we deployed 11 trap-signaling devices in HAVO between April and October 2005 at the beginning of 5-d sessions and monitored signals daily. We physically checked traps when no signal was received. We also checked signals periodically from the furthest available LOS location. Traps were closed at the end of each trapping session to maintain battery longevity.

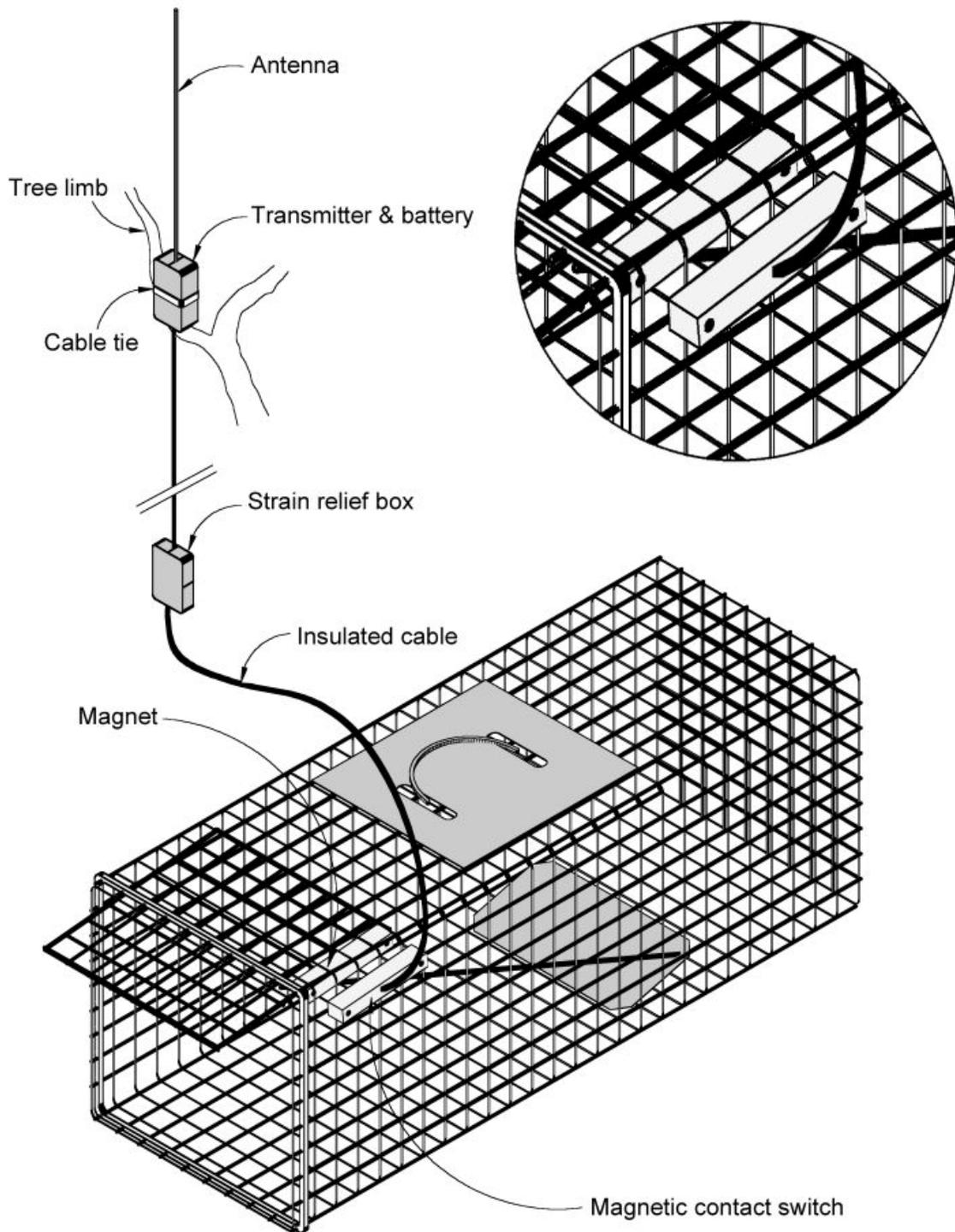


Figure 6.1. A simple trap-signaling device on an armed live trap, including transmitter in a sealed 3 x 6 cm plastic housing with a rigid 40 cm external antenna and a cable leading to a strain relief box and a magnetic switch. Total weight of the device is < 225 g.

d. Results

Between November 2004 and December 2005, we monitored trap-signaling devices at distances ranging from 0.5–4.6 km ($\bar{X} = 1.9 \pm 1.2$ SD). Between April and October 2005, we monitored signals at distances ranging from 12.9–15.8 km ($\bar{X} = 13.8 \pm 1.1$ SD). Furthest LOS distances ranging from 16.1–19.0 km ($\bar{X} = 17.1 \pm 1.1$ SD).

We tested the devices for a total of 646 trap-nights of which they operated properly during 631 (97.7%) trap-nights (Table 6.1). We received signals with no capture or change in status during 570 (88.2%) trap-nights. The signaling system alerted us to the capture of 30 target animals: 5 mongooses, 4 feral cats, and 4 rats in the HAVO study area; and 8 mongooses, 7 feral cats, and 2 rats in MKFR. Non-target captures included an Erkel's Francolin (*Francolinus erckelii*) and a feral pig in MKFR. Feral pigs rolled over traps 21 times in MKFR without capture, as evidenced by extensive rooting. In 16 cases, trap doors closed and transmitters ceased transmitting. However, in 5 cases, doors remained open and transmitters continued to transmit while traps were inoperable. In 8 other cases in MKFR and HAVO, trap doors closed without capturing an animal and devices stopped transmitting. In 15 (2.3%) trap nights, devices stopped transmitting without a change in trap status. These cases were caused by strained electrical wires that interrupted the switch circuit and rodents chewing through cables. There were no cases where devices continued to transmit after an animal was captured in a trap. Of the 2 units that expired during the study, batteries lasted 39 and 41 trap-nights over the course of 11 months of operation. Assembled trap-signaling units weighed < 225 g. The components of each device cost approximately \$270 US plus 2-4 hr labor for assembly and testing.

f. Discussion

The high reliability and long range reception of our trap-signaling devices indicated that the system was an effective tool for trapping vertebrate pests. Our trap-signaling system weighed only < 225 g, but was effective at ≤ 17.1 km. These devices failed to function < 1% of trap-nights despite being exposed to periods of heavy rainfall during testing. Failures were entirely due to interference by non-target species. Compared to the relatively complex mobile signaling system designed by Larkin et al. (2003) priced at \$300–700 US per trap set, the cost for our system was \$270 US plus labor.

Table 6.1. Number of nights that trap-signaling devices were tested on Mauna Loa, Hawai`i Volcanoes National Park (HAVO), April–October 2005 and Mauna Kea Forest Reserve (MKFR), November 2004–December 2005.

	Trap-nights		
	HAVO	MKFR	Total
Success			
No change in status	241	329	570
Animal captured	13	19	32
Trap sprung, no capture	3	21 ^a	24
No signal, no change in status	12	3	15
Failure			
Signal received, trap rolled over	0	5 ^b	5
Total	269	377	646

^aTrap sprung without capture (5) or rolled over by feral pigs (16).

^bOnly where feral pigs were present, doors closed and the devices ceased transmitting.

Other key features of our system include design simplicity, ease of transport, and deployment. This simple design requires only a few off-the-shelf components that are light, small, and durable. Light weight and small size allowed several devices to be carried at a time. Each device can be deployed in approximately 15 min on virtually any type of trap that has distinct armed and sprung positions. Because the system has few components, there are few opportunities for failure.

Although batteries failed in 2 cases, we do not recommend a replaceable battery cartridge because contacts may corrode in a humid environment. Battery leads are soldered to tabs and sealed in a weatherproof box, therefore replacement requires technical skill. More sophisticated designs may require active power sources such as solar panels, increasing costs as well as the likelihood of failure. Battery life may be extended by programming the transmitter with a slower pulse rate (E. Levine, Merlin Systems Inc., pers. comm.). Other problems may include limited LOS reception, cable strain, and interference by some wildlife species. In topographically complex areas, the device may not function well due to LOS limitation; however, we found that strategic placement of traps, devices, and receivers can overcome many obstacles to reception.

We strongly recommend encasing the entire length of cables in flat letter-folded duct tape for 2 reasons: 1) to reduce cable strain during transport and operation; and 2) to reduce damage by rodents or other animals. The entire system should be regularly inspected for damage, particularly where abundant non-target animals may disturb traps. Solutions to reduce feral pig disturbances may include placing traps away from disruptive non-target species or securing traps to the ground. All traps should be periodically checked to prevent animals from being held longer than care and use protocols specify.

f. Management Implications

Our design for a trap-signaling device can be used to determine the status of distant traps at any time, allowing staff to respond only to signaling traps, thereby reducing time spent checking traps and improving animal welfare (Johnston et al. 2005). This is particularly important in situations where there is a possibility of capturing endangered species, allowing managers to quickly release these as well as non-target species.

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VI) Bait Preference of Feral Cats and Interference by Small Mammals in Hawai`i Volcanoes National Park

Heidi Hansen¹, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey
Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i
National Park, HI, 96718, USA

Steven C. Hess, and Paul C. Banko, U.S. Geological Survey Pacific Island Ecosystems Research
Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

¹ Current address: Department of Land and Natural Resources Division of Forestry and Wildlife,
19 East Kawili St., Hilo, HI 96720, USA

a. Introduction

Feral cats (*Felis catus*) became abundant in forests of the Hawaiian Islands soon after their introduction in the late 1700s (King 1984; Rothschild 1893; Perkins 1903). They were subsequently reported to be notorious predators of birds that contributed to the decline and extinction of some Hawaiian species (Perkins 1903; Berger 1981; Ralph and van Riper 1985; Stone 1985; Snetsinger et al. 1994). Currently, feral cats range throughout the Hawaiian Islands from high-density colonies near coastal areas where pets are frequently abandoned and fed by people (Winter 2003), to remote, low-density populations in montane forests and subalpine areas of Maui (Simons 1983) and Hawai`i Island (Hu et al. 2001).

In Hawai`i, cats are currently important predators of terrestrial birds that nest near the ground (Kowalsky et al. 2002), and in trees (Hess et al. 2004). Cats are also important predators of colonial seabirds (Smith et al. 2002), and may be particularly destructive to some burrow nesting seabird species which have delayed maturity and low reproductive potential (Simons and Hodges 1998). An individual cat may kill as many as 200 seabirds per year (van Aarde 1980). Cats may also carry and transmit toxoplasmosis which is known to cause mortality in a number of bird species in Hawai`i (Danner et al. 2007; Work et al. 2000; 2002).

Feral cats are especially difficult to control in the wild due to their wariness of traps and humans. The success of cat control programs worldwide has been mixed, often requiring continuous effort due to the lack of effective bait and trap lures (Algar and Burrows 2004). Current predator control programs on Hawai`i Island use live traps to protect threatened

nesting birds, including the Nēnē (Hawaiian goose; *Branta sandvicensis*), ʻUaʻu (Hawaiian Petrel; *Pterodroma sandwichensis*), and Palila (*Loxioides bailleui*). However, most trapping efforts occur in remote areas and require tremendous effort in hiking or expense in flying by helicopter to these sites.

Non-target species such as rats (*Rattus* spp.) and small Indian mongooses (*Herpestes auropunctatus*), while not benign, may not pose as urgent a threat to some endangered species as feral cats. For example, mongooses depredate ground-nesting birds, but are poor climbers and therefore seldom threaten tree-nesting passerines (Hays and Conant 2007). Non-target species may also be more abundant and easier to attract and capture than feral cats, thereby interfering with the capture of cats by pre-empting single-capture traps and reducing capture efficiency. Such non-target small mammals may also rapidly consume bait before cats encounter traps, necessitating frequent bait replacement. Therefore, baits are needed which are 1) highly effective in attracting feral cats to traps, 2) relatively species specific, 3) easy to transport, and 4) individually prepackaged to maintain freshness during long field sessions.

Many studies report problems of non-targets consuming bait and getting caught in traps (Seymour et al. 2005; Veitch 2001), and target-specific bait and scent attractants have been tested for feral cats using captive animals or biomarkers (Algar and Burrows 2004; Marks et al. 2006). However, few studies quantify the interferences of non-targets. Moreover, baiting studies using captive animals and biomarkers are labor intensive and require large sample sizes (Hegglin et al. 2004). In contrast, the evaluation of trapping data and bait uptake using camera traps provides detailed data on trap interferences by other species and bait preferences of feral cats. Our objectives were to 1) investigate bait preferences of feral cats, 2) quantify trap interferences by non-targets, 3) and evaluate bait uptake and preferences of non-targets.

b. Study Area

Trapping

Study areas were located on Mauna Loa (19° 29' N, 155° 25' W) in Hawaiʻi Volcanoes National Park (HAVO), and on North Mauna Kea (NMK; 19° 54' N, 155° 27' W), Hawaiʻi Island. NMK extends from 1,700–2,850 m elevation and is characterized as dry subalpine woodlands consisting of mamane (*Sophora chrysophylla*) and naio (*Myoporum sandwicense*) trees. HAVO extends from 1,175–2,700 m elevation. This area consists of some old growth koa (*Acacia koa*) and ʻōhiʻa (*Metrosideros polymorpha*) forests (e.g., Kīpuka Puaulu and Kīpuka Kī) but is primarily characterized as subhumid and subalpine. Vegetative cover can be < 10% at higher

elevations and is dominated by shrubby species such as pūkiawe (*Leptecophylla tameiameia*), `a`ali`i (*Dodonaea viscosa*), kūkaenēnē (*Coprosma ernodeoides*), `ōhelo (*Vaccinium reticulatum*), and na`ena`e (*Dubautia ciliolata*) (Wagner et al. 1999). NMK contained abundant feral pigs (*Sus scrofa*) and sheep (*Ovis* spp.), but HAVO was free of ungulates. Small mammals (*Mus musculus* and *Rattus* spp.) were abundant at both sites. Substrates of both volcanoes were geologically young and interspersed with major lava flows.

Bait Stations

Two paired bait stations were located on the southeast slope of Mauna Loa in Kīpuka Kī and along the Mauna Loa Strip Road. Kīpuka Kī is a mesic forest consisting of large koa (*Acacia koa*), `ōhi`a, and mānele (soapberry; *Sapindus saponaria*) trees at approximately 1,300 m (Mueller-Dombois et al. 1981). Understory vegetation consists of primarily native ferns and grasses where the overstory is dense and non-native plants such as blackberry (*Rubus argutus*) where the canopy is more open. The other site was located along Mauna Loa Road at 1,600 m. The overstory consists primarily of koa and the understory consists of native shrubs such as pūkiawe and `a`ali`i and non-native grasses and shrubs. Another single bait station was located on a dirt two-track road that bisects the southeast slope of Mauna Loa. Vegetation at this location consists primarily of koa and non-native grasses.

c. Methods

The relative effectiveness and attractiveness of baits and lures for trapping feral cats were compared using: 1) capture rates of feral cats in HAVO and NMK, and 2) active infrared cameras set up at bait stations in HAVO. The bait stations and trapping occurred concurrently from November 2004 to December 2005.

Trapping

We deployed 130 Tomahawk[®] model 106 live traps (66 x 23 x 23cm) in HAVO and NMK (58 and 72, respectively). In compliance with University of Hawai`i Institutional Animal Care and Use Committee protocols, we covered all traps with plastic sheets and placed a cloth inside to serve as bedding. We checked traps daily and wired the trap doors open when not in use. To quantify the effectiveness of popular baits, we baited the traps with commercial canned cat food (Friskies[®], Nestlé Purina PetCare Company, St. Louis, Missouri) or sardines in soybean oil (Beach Cliff[®], Brunswick[®], ©Bumble Bee Seafoods, LLC, San Diego, California). Bait was replaced as needed or every 5 d of trapping. Interferences in trapping were defined as non-target captures (e.g., rat, mongoose), TRO (trap rolled over by feral pigs), TNC (trap triggered

with no capture), and trap missing. Traps were considered operational if either a cat was captured or there was no change in trap status. We used simple linear regression to test whether trap interferences increased with the number of operational traps.

To determine bait effectiveness and degree of interference, we analyzed differences in capture rates for cats and mongooses using canned cat food and sardine bait. Similarly to Seymour et al. (2005), we estimated effective trap nights (ETN) by assuming that trapping interferences reduced trapping effort by half a night. The number of cats caught per trap was not correlated with the number of days the trap was set (Spearman's rank correlation: $r = 0.32$, $n = 29$, $P = 0.09$), so we analyzed catch frequencies for cats and mongooses rather than animals captured per trap night. The catch frequency was defined as the proportion of traps per ETN baited with either bait that caught animals (Caughley 1980). We used χ^2 analyses to test for associations between catch frequency and bait tested.

Bait Stations

Animal visits to two paired bait stations separated by at least 100 m at each site were monitored using active infrared cameras from August to December, 2005. We placed stations in areas where feral cats had been seen previously in open corridors and along trails that appeared to be natural travel routes for cats. All stations were within 50 m of the Mauna Loa Road. We tested 4 different types of food-based baits and attractants: canned cat food; sardines; a sausage that we formulated and produced from pork, lard, preservatives, and sheep casings; and approximately 2.3 g of catnip in 5 cm square burlap pouches (Catnipetc.com, Needham, Massachusetts). Each bait type was tested at each station for at least 15 d. The same type of bait was not tested at the other station simultaneously. Animal visits to the stations were monitored using active infrared trail monitors (TrailMaster[®] TM1550; Goodson and Associates, Inc., Lenexa, Kansas) that emit an infrared beam between a transmitter and receiver. Each trail monitor was attached to a camera kit (TrailMaster[®] TM35-1) equipped with an automatic, weatherproof, 35-mm Canon[®] film camera with an automatic light flash. When an animal passed through the beam, it was photographed and an event was recorded.

We used the following settings for TrailMaster[®] monitors: the camera delay (minimum time between photos) was set from 30 seconds to 2 minutes depending on activity of the station; the sensitivity (the amount of time required for the beam to be broken) of the trail monitors were set between five and seven in order to reduce photographs of insects and vegetation movement; and activated for 24 hr/d. Trail monitors were set at a height of

approximately 10 cm so that small mammals near the bait station would always be recorded (Tolle and Kéry 2003). Stations were maintained daily. The alignment of the cameras and monitors were inspected daily. We recorded the approximate proportion of bait take, number of events, number of photographs, time and date of photographs taken, and changes made to the infrared unit (cables, cleaning) in order to identify potential equipment failure. Bait was replaced when more than 60% of the bait was gone. Film and batteries were replaced as necessary. The entire lengths of all cables were encased by camouflage pattern duct tape to provide strain relief, protection from the environment, and alternative material for rodents to gnaw on instead of cable insulation.

We also monitored animal visits to a separate station that was baited with sardines for 40 d during November and December 2005. The station was located 3 m from the road. Visits were documented using a passive infrared digital camera (Bushnell® Digital Trail Scout™ with Night Vision Pro 11-9900; Lenexa, Kansas). When an animal entered the infrared zone of detection, a 15-second digital video image was recorded and stored on an SD flash memory card. We maintained this station once a week and replaced bait as necessary.

We estimated the proportion of bait take each time the bait station was checked from a scale of 0 (no bait consumed) to 1 (all bait consumed) in 0.1 increments. The mean bait take was calculated for all baits and individual baits, excluding catnip. We also calculated the mean number of mammals photographed for all baits and individual baits. We tested whether mean bait take or the number of photographed mammals increased over time using simple linear regression. Because catnip could not be consumed, we only analyzed the number of photographed animals at the bait stations. We used chi-squared analyses to test for differences between the attractiveness of the different bait types.

d. Results

Trapping

We trapped for a total of 3,389 trap nights in HAVO and NMK (1,032 and 2,357, respectively) and captured 35 feral cats (Table 7.1). There were 323 incidences of trap interferences, reducing the number of traps available to catch cats to 3,227 ETN (95.2% of total trap nights). The primary cause of trap interference was due to TRO by feral pigs in NMK ($n =$

Table 7.1. The number of total trap nights, effective trap nights (ETN) and captures of feral cats and trap interferences due to capture of mongooses, rats, other non-targets (e.g. rats, feral pig, and Erkel's Francolin), traps rolled over (TRO) by feral pigs, traps tripped with no capture (TNC), and traps missing while trapping in Hawai`i Volcanoes National Park (HAVO) and North Mauna Kea (NMK), Hawai`i, between November 2004 to December 2005.

Location	Trap Nights	Effective Trap Nights	Feral Cats caught	Trap Interferences				
				Mongoose	Other Non-			Trap
				caught	targets caught	TRO	TNC	Missing
HAVO	1032	1008.5	11	37	4	0	6	0
NMK	2357	2216.5	24	37	8	185	41	5
Total	3389	3225	35	74	12	185	47	5

Table 7.2. The number of captures (n) and capture frequencies of feral cats and mongooses per effective trap night (ETN) using sardines and canned cat food as bait in Hawai`i Volcanoes National Park (HAVO) and North Mauna Kea (NMK), Hawai`i, November 2004–2005.

Location	Bait type	Effective Trap Night (ETN)	Feral Cats		Mongooses	
			n caught	catch/ETN	n caught	catch/ETN
HAVO	sardines	85	2	0.024	15	0.176
	cat food	923.5	9	0.01	22	0.024
NMK	sardines	92.5	3	0.032	0	0
	cat food	2124	21	0.01	37	0.017
Total		3225	35	0.011	74	0.022

185, 57.3% of interferences). The primary non-target species captured were mongooses ($n = 74$, 22.9% of interferences) (Table 7.1). TNC, other non-target captures [rats, feral pig, and Erkel's Francolin (*Francolinus erckelii*)], and missing traps accounted for 47 (14.6%), 12 (3.72%), and 5 (1.55%) of trap interferences, respectively. The number of trap interferences increased significantly with the number of open traps (slope = 0.185 interference/open trap, $F = 54.9$, $P < 0.001$). Because mongooses accounted for the majority of non-target captures, all other trap interferences were excluded from further analyses.

Sardines were tested for a total of 177.5 ETN (85 in HAVO, 92.5 in NMK) and canned cat food was tested for a total of 3,047.5 ETN (923.5 in HAVO, 2124 in NMK) (Table 7.2). The greatest frequency of cat and mongoose captures occurred when using sardines (5 cats and 15 mongooses for 177.5 trap nights) rather than canned cat food (30 cats and 59 mongooses for 3047.5 trap nights). However, we found no significant difference between the two bait types and catch frequencies for cats and mongooses ($\chi^2 = 0.0011$, $DF = 1$, $P \leq 1$). The 95% CI for the catch frequencies of traps baited with sardines or canned cat food were not exclusive for cats, but were exclusively greater for mongooses trapped with sardines than with cat food (Figure 7.1).

Bait Stations

There was a significant increase in the mean proportion of bait take for all baits as the bait testing proceeded. Individually, mean bait take increased significantly with the number of nights that canned cat food and sardines were tested, but not for pork sausage (Table 7.3). There was a significant decrease in the mean number of photographs taken at all baits as the bait testing proceeded. Photographs taken at sardines, canned cat food, and pork sausage significantly declined as bait testing proceeded, but not for cat nip (Table 7.4).

In total, we obtained photographs of 1476 small mammals (Table 7.5). Mongooses were the principal mammal photographed, accounting for 939 (69.5%) pictures. Other mammals recorded at the bait stations included 398 photographs of rats (29.5%) and 9 (0.7%) of mice (*Mus musculus*). Feral cats were photographed only 5 (0.4%) times. Individual animals may have been photographed multiple times and there was greater than one conspecific individual in many photographs.

Statistical analysis of the number of photographs of each species by bait type was problematic because there were five or fewer photographs of some species at particular baits. Because we were specifically interested in feral cats and there were also few photographs of

mice, we combined mice with rats into a larger group of rodents for analysis. Nonetheless, we found strong differences between mongooses, rodents, and cats photographed at the 4 bait types ($\chi^2 = 47.79$, $DF = 6$, $P < 0.001$). We then censored pork sausage and catnip from the data to test for differences between the attractiveness of sardines and canned cat food. There was no significant difference in the number of cats, mongooses, and rodents attracted to these 2 baits ($\chi^2 = 3.09$, $DF = 2$, $P \leq 1$).

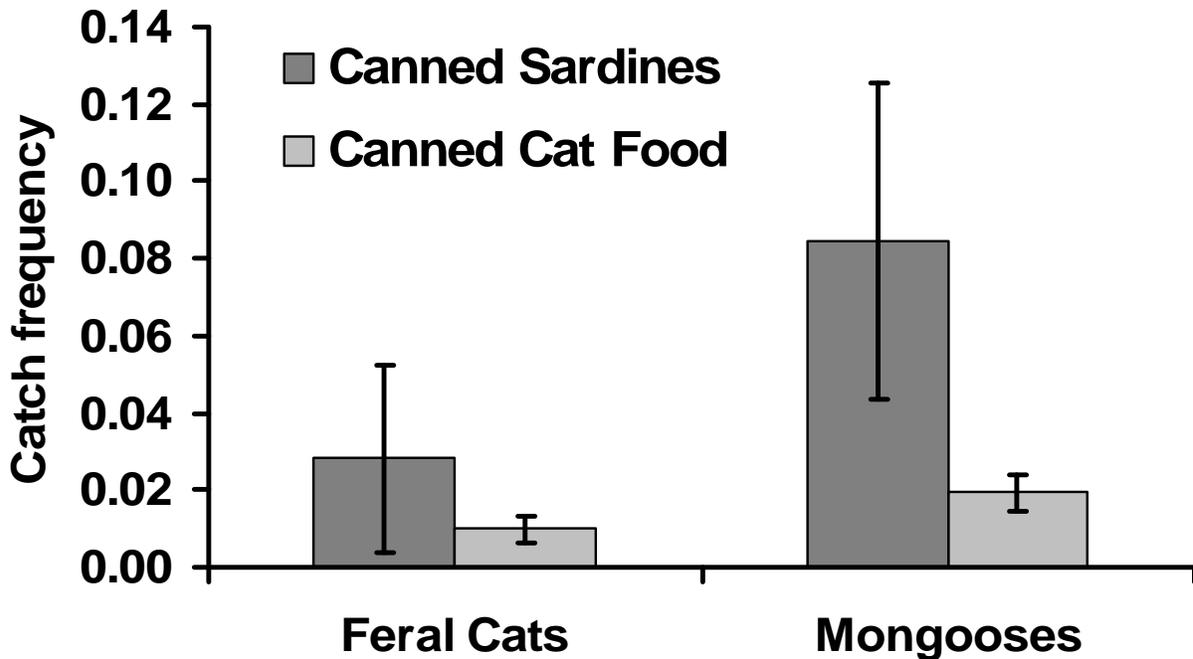


Figure 7.1. Catch frequencies (proportion of traps catching animals per effective trap night [ETN] \pm 95% CI) of feral cats and mongooses for canned sardines and canned cat food baits in Hawai`i Volcanoes National Park (HAVO) and North Mauna Kea (NMK), Hawai`i, November 2004–2005. The 95% CI for the catch frequencies of traps baited with sardines or canned cat food were not exclusive for cats (0.0286 sardine-baited traps catching cats /ETN; 95% CI: 0.00383, 0.0525 vs 0.00989 cat food-baited traps catching cats/ETN; 95% CI: 0.00634, 0.0133) but were exclusively greater for mongooses trapped with sardines (0.0883 sardine-baited traps catching cats /ETN; 95% CI: 0.0436, 0.125) than with cat food (0.0196 cat food-baited traps catching cats/ETN; 95% CI: 0.0145, 0.0243).

Sardines were the most visited bait type with 641 (47.4%) photographs. Pork sausage and canned cat food accounted for 383 (28.3%) and 322 (23.8%) visits while catnip had only 67 (5.0%) visits. Feral cats were photographed only at sardine bait. One feral cat was video-

Table 7.3. Regression analysis of mean proportion of bait take per trap night for three food baits tested in Hawai'i Volcanoes National Park (HAVO), August–December, 2005.

Bait	<i>n</i>	DF	Slope	<i>F</i> -value	<i>P</i> -value
Pork sausage	12	11	0.018	4.24	0.067
Sardines	19	18	0.015	8.20	≤ 0.011
Cat food	17	16	0.022	4.63	≤ 0.048
Overall	17	16	0.017	12.60	≤ 0.003

Table 7.4. Regression analysis of mean number of photographs per camera night for four baits tested in Hawai'i Volcanoes National Park (HAVO), August–December, 2005.

Bait	<i>n</i>	DF	Slope	<i>F</i> -value	<i>P</i> -value
Catnip	24	23	-0.031	1.72	0.203
Pork sausage	16	15	-0.529	6.82	≤ 0.002
Sardines	21	20	-0.407	16.61	≤ 0.001
Cat food	20	19	-0.027	5.91	≤ 0.026
Overall	24	23	-0.26	23.97	≤ 0.001

Table 7.5. The number of photographs and number of animals per photograph, by species, at four different types of bait on Mauna Loa Hawai'i Volcanoes National Park, 2004–2005.

Species	Number of Photos				Total Photos with Animals	Total Animals Photographed	Percent of Photos
	Canned Sardines	Canned Cat Food	Pork Sausage	Catnip			
Feral Cat (<i>Felis catus</i>)	5	0	0	0	5	5	0.37
Mouse (<i>Mus musculus</i>)	2	6	0	1	9	9	0.67
Rat (<i>Rattus</i> spp.)	171	89	177	23	398	487	29.5
Small Indian Mongoose (<i>Herpestes auropunctatus</i>)	463	227	206	43	939	975	69.5
Total Photos with Animals	641	322	383	67	1351	1476	--

recorded licking a can of sardines for 15 seconds. Sardines accounted for most (49.3%) mongoose photographs, and canned cat food accounted for 24.2% of mongoose photographs. Pork sausage accounted for most (44.5%) rat photographs. Multiple conspecifics appeared in 32 (3.4%) and 26 (6.5%) mongoose and rat photographs, respectively.

e. Discussion

The conservation of endangered ground-nesting birds and tree-nesting passerines in Hawai`i requires effective means for controlling feral cats which climb trees and depredate eggs, nestlings, and adults during this vulnerable stage of life (Hess et al. 2004). Our results suggest that non-targets, especially small Indian mongooses, interfered with feral cat control programs. The high rates of trap interferences and bait uptake by other species suggested that the baits we tested were not highly effective for trapping feral cats. Non-targets, primarily rats, have been reported to reduce the number of available traps on Little Barrier Island by 1.5% (Veitch 2001). We reported a higher rate of trap interferences with a reduction of trap availability of 4.8%, primarily due to mongoose captures and traps rolled over by feral pigs. Not surprisingly, the number of interferences was correlated with the number of open traps.

The food-based baits and catnip we tested were olfactory attractants to small mammals, signaling the presence of food or novel odors in the environment. With exception to the pork sausage, the baits we tested are commonly used in current predator control programs in Hawai`i and by trappers (Moruzzi et al. 2002). As such, the attractive distance of the odor must lie within the home range of the target species to be effective. The home ranges of mongooses and rodents in Hawai`i are much smaller than that of feral cats (Lindsey et al. 1999; Smucker et al. 2000), and feral cats also live in much lower densities than other small mammals. Also, feral cats may not be as adept as finding baits as other species (Algar and Burrows 2004).

Infrared-triggered cameras are becoming a widely used tool that provides data about elusive animals that are difficult to capture (Swann et al. 2004) and can also be used to monitor visitation rates and bait uptake of different species (Hegglin et al. 2004). Although bait stations may produce biases in detection because of baits used to attract animals (Moruzzi et al. 2002), they simulate trap visitation with commonly-used baits and provide data on non-target interferences encountered while trapping. Infrared-triggered photography enabled us to evaluate the bait preference and interference of animals without the deterrence of human presence (Wolf et al. 2003).

As bait testing proceeded, there was a significant increase in bait take which may be directly related to the significant decrease in photographs taken. This could be due to a few individual animals rapidly consuming the bait and therefore making the bait station unattractive to other animals. Similarly to trapping data, mongooses interfered the most with baiting efforts. Because 3–6% of bait visits by mongooses and rats involved more than one individual, multiple-capture traps may be an effective way to quickly reduce densities of these species such that cats may be more effectively trapped.

There was no statistical difference in the attractiveness of sardines and canned cat food between feral cats, rodents, and mongooses in either trapping or bait station testing, although our only observations of cats occurred at sardine bait and more cats were caught on sardines, suggesting that sardines may be a more attractive bait to cats than the other baits we tested. The degree to which cats were attracted to sardine baits was surprising considering that fish were completely foreign to these montane and subalpine inhabitants who primarily consume invertebrates, small mammals, and birds (Hess et al. 2004; 2007). However, both catch frequencies and visitation rates showed that sardines were also the most attractive bait to mongooses. Therefore, using sardines as feral cat bait in locations where the two species co-occur could reduce trap effectiveness and require an increase in trapping effort. It is possible that fresh fish may be more attractive than either of these canned baits. Fresh baits, however, would require refrigeration which is not available at many remote field camps, rendering these types of baits impractical for many feral cat control programs. Also, sardines packed in oil may not decay as quickly as fresh baits. A potential solution to reduce interference by mongooses is to use an alternate control method such as a toxicant around trap sites. Standard rodent bait stations with Eaton's Bait Blocks® containing 0.005% diphacinone and fish flavorizer were found to be effective at controlling mongooses over small areas (Smith et al. 2000).

Sausages are potentially effective baits for remote areas because they can 1) be specifically formulated, 2) contain preservatives, and 3) be partially cooked to further enhance longevity. Algar and Burrows (2003) reported that feral cats are highly attracted to a patented kangaroo meat sausage, however, the pork sausage we made appeared to be only attractive to rats and mongooses. The addition of chicken fat and flavor enhancers could increase the attractiveness of our pork sausage to feral cats (Algar and Burrows 2003). We found catnip to be neither attractive nor specific to feral cats, perhaps a neophobic reaction to an odor they were unlikely to encounter naturally in this environment. Moreover, mongooses, rats, and even

mice were attracted to catnip to a limited extent. We obtained 2 photographs of a mongoose apparently scent marking a pouch of catnip. Other attractants, such as a mixture of cat feces and urine with a preservative glycerin (B. Wood et al., unpublished data), may be effective at attracting feral cats to traps and are worthy of testing.

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VII) Using population genetic tools to develop a control strategy for feral cats (*Felis catus*) in Hawai`i

Heidi Hansen¹, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey
Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i
National Park, HI, 96718, USA

Steven C. Hess, U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field
Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

David Cole², Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey Pacific
Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National
Park, HI, 96718, USA

Paul C. Banko, U.S. Geological Survey Pacific Island Ecosystems Research Center, P.O. Box 44,
Kīlauea Field Station, Hawai`i National Park, HI, 96718, USA

¹ Current address: Department of Natural Resources Division of Forestry and Wildlife, 19 East
Kawili St., Hilo, HI 96720, USA

² Current address: University of Hawai`i at Mānoa, 3190 Maile Way, Room 101, Honolulu, HI
96822, USA

a. Introduction

Population genetic tools have proven beneficial for the management and conservation of rare and endangered wildlife (Paetkau et al. 1997; Spong et al. 2000). Only recently have these tools been applied in invasive species management (Robertson and Gemmell 2004; Abdelkrim et al. 2005; Rollins et al. 2006). Invasive species are the main cause of species extinctions in island ecosystems (Courchamp et al. 1999) and the second main cause of biodiversity loss after habitat destruction (Vitousek et al. 1997). Successful invasive species management requires identifying a target population of manageable size that has a low recolonization risk (Robertson and Gemmell 2004). Attempts to control only a fraction of the population or a sink population could result in rapid recolonization (Robertson and Gemmell 2004; Abdelkrim et al. 2005). Identifying routes of potential migration is difficult using direct observations, but vital for controlling invasive species (Robertson and Gemmell 2004; Abdelkrim et al. 2005; Rollins et al. 2006). Population genetics can provide valuable information about the demographic status and

dynamics of invasive species and may provide an alternative approach for developing control strategies (Robertson and Gemmell 2004; Abdelkrim et al. 2005; Pontier et al. 2005). Feral cats (*Felis catus*) are currently listed as one of the '100 world's worst invasive alien species' (Lowe et al. 2000). Domestic cats were brought to Hawai`i on European ships in the late 1700s (King 1984) and feral animals were reported by 1840 in remote montane areas of Hawai`i Island (Brackenridge 1841). Currently, feral cats occur in low densities in montane forests and subalpine areas of Maui (Simons 1983) and Hawai`i Island (Hu et al. 2001) and are frequent predators of endangered Hawaiian birds including colonial seabirds (Smith et al. 2002), ground-nesting waterfowl (Banko 1992), and tree-nesting passerines (Hess et al. 2004). Cats also carry *Toxoplasma gondii*, which has caused fatal toxoplasmosis in endangered Hawaiian birds (Work et al. 2000; 2002) and Hawaiian monk seals (*Monachus schauinslandi*) (Honnold et al. 2005).

The behavior of feral cats in remote subalpine and alpine environments of Hawai`i makes traditional methods to understand population dynamics problematic; they are solitary, elusive, hard to capture, and inhabit areas that are difficult to survey (Hess et al. 2004). Their dispersal patterns and the inaccessibility of remote locations also make them logistically difficult to manage. Research on the genetic structure of feral cats in island ecosystems is rare (Pontier et al. 2005), but can provide valuable information for formulating control strategies and determining the scale and location of control efforts (Rollins et al. 2006). Here, we describe the use of seven highly polymorphic microsatellite markers to estimate the genetic structure of three feral cat populations on Hawai`i Island. The objectives of our research were to 1) evaluate genetic diversity and population structure; 2) assess levels of gene flow and connectivity between populations; 3) identify potential source populations; 4) characterize population dynamics; and 5) evaluate evidence for sex-biased dispersal. We present results that can be used to formulate an island-wide control strategy.

b. Study Area

Study areas were located on Hawai`i Island in Hawai`i Volcanoes National Park (HAVO), and on North Mauna Kea (NMK), and West Mauna Kea (WMK) in designated critical habitat for Palila (*Loxioides bailleui*) (Figure 8.1). NMK and WMK were characterized as dry subalpine woodlands from 1,701–2,835 m elevation. HAVO extended from 800–2,700 m elevation and consisted of montane wet forest grading into drier `ōhi`a (*Metrosideros polymorpha*) scrub and subalpine shrubland. Substrates in HAVO were geologically young and interspersed with recent

lava flows. HAVO and Mauna Kea were separated by extensive recent lava flows. The linear distances separating WMK from NMK, WMK from HAVO, and NMK from HAVO were 18 km, 50.2 km, and 53.5 km, respectively, and greater than the average home ranges reported for feral cats on Mauna Kea (1,418 ha and 770 ha, respectively; Goltz et al., in press). The approximate areas from which cats were captured varied from 8 km² at NMK and 32 km² at WMK, to 87 km² at HAVO. Traps were arranged on transects perpendicular to elevation contours on NMK and WMK, and parallel to contours in HAVO. The population density of cats was unknown.

c. Methods

Population sampling and microsatellite analysis

Feral cats were trapped from 2000–2005 to reduce predation on endangered species such as Nēnē (Hawaiian goose; *Branta sandvicensis*) and `Ua`u (Hawaiian Petrel; *Pterodroma sandwichensis*) in HAVO, and Palila on NMK and WMK. Traps were checked daily and cats were euthanized according to University of Hawai`i IACUC protocol. We collected muscle tissue samples from 85 feral cats (49 males, 36 females) and stored samples in lysis buffer (0.1M Tris-HCL pH 8.0, 0.1M sodium EDTA, 2% SDS) at -20°C until extraction. We extracted genomic DNA from the tissue using the QIAGEN® DNeasy™ Tissue Kit (Qiagen, Inc., Valencia, California, USA).

We generated complete genotypes for all feral cat samples using seven polymorphic short-tandem repeat (STR) microsatellite markers previously developed and optimized for multiplexing of domestic cats (Menotti-Raymond et al. 2005) and included a gender-identifying sequence tagged site (STS) from the domestic cat Y-chromosome SRY gene. The markers were labeled with D2, D3 and D4 WellRED fluorescent dyes (Beckman Coulter) at the five-prime end of either the forward or reverse primers. This allowed multiplexing of all eight marker loci in one sequence run by distinguishing loci and associated alleles that overlapped in size.

A single PCR (Polymerase Chain Reaction) was performed for each sample in a 20.0 µL volume containing 1X PCR Gold buffer (Applied Biosystems) with final reaction concentrations (adapted from Menotti-Raymond et al. 2005): 1.5 mM MgCl₂, 0.8 mM dNTPs, 1 unit *Amplitaq*

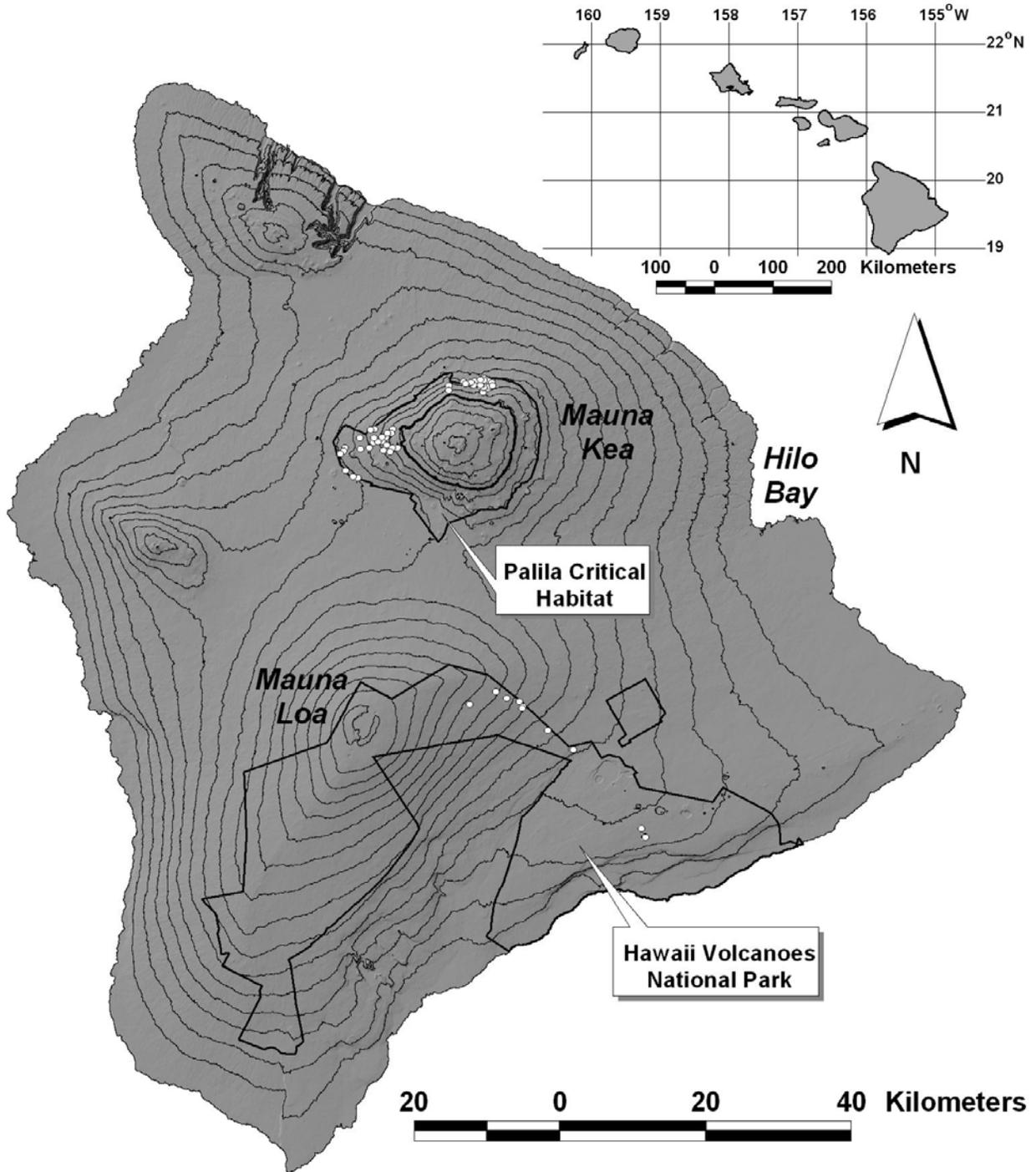


Figure 8.1. Capture locations of feral cats sampled for genetic analyses (white dots) on Mauna Loa within Hawai'i Volcanoes National Park ($19^{\circ} 26' N$, $155^{\circ} 19' W$) and on the north ($19^{\circ} 54' N$, $155^{\circ} 27' W$) and west ($19^{\circ} 49' N$, $155^{\circ} 36' W$) slopes of Mauna Kea within the critical habitat of Palila (*Loxioides bailleui*). Elevation contour intervals are at 250 m.

gold DNA polymerase (Applied Biosystems), 0.16 mg/mL bovine serum albumin, 8 μ L microsatellite primer mix (see final concentrations in Table 8.1), and ~20 ng genomic DNA. PCRs were performed using a MJ Research PTC-200 DNA thermocycler using conditions optimized by Menotti-Raymond et al. (2005). The eight-primer pair multiplexes were then visualized on a Beckman-Coulter CEQ 8000 automated capillary sequencer (Beckman-Coulter; Fullerton, CA), one lane for each DNA sample (Core Genetics Facility, University of Hawai`i at Hilo). Allele sizes were estimated using CEQ 8000 v 7.0 and then visually inspected taking into consideration the expected allele size in base pairs for each of the eight loci and the original DNA clones from which the microsatellite loci were developed (Menotti-Raymond et al. 2005).

Genetic diversity

Genetic diversity for each population was summarized as the average number of alleles per locus (A) and average observed (H_o) and expected (H_e) heterozygosities (Nei 1978) using Fstat 2.9.3 (Goudet 2001). We calculated the unbiased inbreeding coefficient f (similar to F_{IS} ; Weir and Cockerham 1984) for each population-locus combination and tested for deviations from Hardy-Weinberg equilibrium (HWE) using Genepop 3.4 (Raymond and Rousset 1995). Departure from Hardy-Weinberg expectations was assessed by exact tests with unbiased P -values based on a Markov chain simulation (with 1000 as dememorization number, 500 batches, and 1000 iterations per batch; Guo and Thompson 1992). Loci were combined using Fisher's method (Raymond and Rousset 1995) to examine departure from equilibrium for each population. A Bonferonni correction was used to adjust significance levels across multiple tests (Goudet 2001). All loci occurred on different chromosomes or different linkage groups on the same chromosome (Menotti-Raymond et al. 2005) and were considered independent markers.

Population structure

We investigated population structure between the three populations using several methods with analyses for all feral cats, and separate analyses for males and females. We calculated the unbiased estimator θ (analogous to F_{ST} ; Weir and Cockerham 1984) for all population pairs using Fstat (Goudet 2001). In addition to θ , ρ (analogous to R_{ST} ; Slatkin 1995) was estimated using R_{ST} Calc 2.2 (Goodman 1997) after standardization of allele sizes in R_{ST} Standardize 2.2 (Goodman 1997). We tested for correlation between the values of θ and ρ using a Mantel test (Pearsons correlation, with 10,000 iterations; Manly 1991) conducted in Mantel 2.0 (Liedloff 1999). We tested the significance levels of θ and ρ for each population pair by calculating P -values and 95% confidence intervals (CI) using a bootstrap procedure. We

compared pairwise θ estimates with log-transformed geographic distances [\ln (km)] to assess whether dispersal is limited spatially for feral cats. Significance of the relationships was tested using a Mantel test (Pearson's correlation, with 10,000 iterations; Manly 1991) conducted in Mantel 2.0 (Liedloff 1999). Geographical distances between individuals were calculated with ArcView GIS 3.2 (ESRI 1999).

We used a Bayesian Markov chain Monte Carlo (MCMC) approach to cluster individual genotypes from all cats ($n = 85$), male cats ($n = 49$), and female cats ($n = 36$) into respective populations and determine the most likely number of populations (K) as implemented in the program Structure 2.0 (Pritchard et al. 2000). Posterior probability values for K (log likelihood; $\ln L$) were estimated for pre-assigned number of clusters ($K = 1-6$ for all cats, $K = 1-3$ for males and females) using the mixed ancestry (admixture) model, a burn-in of 30,000 iterations (checking that parameters α and likelihood had converged), and 100,000 MCMC repetitions for three to four independent runs. We used only genetic information and excluded geographic location from analyses. The K value where the likelihood plateaued was chosen as the number of populations (Pritchard et al. 2000).

We then conducted individual-based assignment tests that assign an individual to the population in which its genotype is most likely to occur to identify possible migrants (Aspi et al. 2006). First, we assigned samples into respective populations based upon the highest proportion of membership (q) from results obtained in Structure. Second, we used the Rannala and Mountain (1997) Bayesian individual assignment method to estimate the likelihood that a cat originated from a given population as implemented in GeneClass2 (Piry et al. 2004). The probability of an individual being a resident was compared to randomly generated genotypes (10,000 replicates) and an individual was rejected from the population if the value was below $P < 0.01$.

Effective population size

To estimate the current number of successfully breeding individuals per population, we used Bartley et al.'s (1992) method of linkage disequilibrium to estimate the effective population size ($N_e(D)$) and 95% CI for each population as implemented in NeEstimator (Peel et al. 2004). To investigate short-term trends of $N_e(D)$, we identified temporal sampling periods that corresponded to trapping intervals in HAVO and NMK and estimated the $N_e(D)$ and 95% CI for each sampling period using Hill's (1981) one sample method. Due to the small numbers of cats caught each year in HAVO, the first and second temporal sampling periods occurred from

2000–2002 ($n = 6$) and 2003–2005 ($n = 9$), respectively, with samples pooled to increase sample sizes. The first and second sampling periods for NMK occurred in 2004 ($n = 18$) and 2005 ($n = 21$), respectively. This analysis was not available for WMK as there was only one sampling period in 2003 ($n = 31$).

Gene flow and migration rate

To obtain indirect measures of gene flow between populations, we estimated the number of migrants per generation (N_m) where N is the effective population size, m is the proportion of migrants per generation, and $N_m = (1/F_{st} - 1)/4$ (Slatkin 1995). N_m estimates are based on historical rates of gene flow and include only individuals that successfully reproduce (Pearse and Crandall 2004).

To determine possible source populations that could be targeted for control (Rollins et al. 2006), we estimated recent migration rates among populations using the Bayesian approach as implemented in BAYESASS+ 1.3 (Wilson and Rannala 2003) and approximated 95% CI. The program was run using a MCMC length of 3,000,000 iterations, a burn-in period of 100,000 (checking that the chains had converged and the log-likelihood values had peaked), and the input parameters (Δp = allele frequency, Δm = migration, Δf) set at 0.15, 0.10, and 0.20, respectively. This analysis included all migrants regardless of success at reproduction (Rollins et al. 2006).

Sex-biased dispersal

To assess sex-biased dispersal, we examined potential differences between sexes in genetic structure and relatedness. We performed an assignment t -test in FSTAT as described by Goudet et al. (2002) and calculated mean assignment indices (mA_i), θ estimates, and f estimates for females and males among the populations (Favre et al. 1997). We used program IDENTIX 4.03 (Belkhir et al. 2002) to estimate pairwise relatedness (r_{xy}) of males and females by applying the method of Lynch and Ritland (1999) and approximated 95% CI. We compared the relatedness between female and male cats within individual populations and relatedness of females among all populations. To investigate potential differences in relatedness between populations, we examined r_{xy} estimates and geographic distances between individuals by using the Mantel test (Pearsons correlation, with 10000 iterations; Manly 1991) conducted in MANTEL 2.0 (Liedloff 1999).

d. Results

Genetic diversity

The mean number of alleles (A) ranged from 7.57 ± 2.99 (mean \pm SD) to 9.00 ± 3.83 according to population (Table 8.2). Mean H_0 ranged from 0.30 to 0.95 according to locus, and ranged from 0.63 ± 0.03 (WMK) to 0.76 ± 0.04 (HAVO) according to population. Mean H_E and H_0 were not significantly different in any population but mean H_E was larger than H_0 (0.72 ± 0.07) in WMK. Except for F124 and FCA731 ($P < 0.002$) in NMK and WMK, the single-locus f values did not differ from zero ($P > 0.05$). Multilocus f values ranged from -0.01 ($P = 0.16$) in HAVO to 0.09 ($P < 0.0007$) in WMK with an overall value of 0.03 ($P = 0.0001$) between all loci and populations. After Bonferonni corrections, f values showed a significant departure from HWE in NMK and WMK, suggesting that either: inbreeding may have increased; we unintentionally sampled parent-offspring pairs; or null alleles are present in these populations (Kyle and Strobeck 2001; Schwartz et al. 2003; Aspi et al. 2006).

Population structure

Overall estimates of ρ (0.023; $P = 0.008$) showed results similar to those of θ (0.028; $P = 0.0001$). Pairwise θ estimates were all significant ($P < 0.05$) and calculated (with 95% CI) as 0.038 (HAVO-NMK; 0.008–0.071), 0.028 (HAVO-WMK; 0.013–0.041) and 0.023 (NMK-WMK; 0.006–0.043). Pairwise ρ -values (with 95% CI) were significant ($P < 0.05$) for HAVO-WMK and NMK-WMK (0.047, 0.017–0.135; and 0.025, 0.008–0.083, respectively) but not for HAVO-NMK (0.004, $P = 0.274$, -0.009–0.084). Pairwise values of ρ and θ were not correlated (Mantel test $r = -0.649$, $P = 0.853$) and pairwise ρ -values had extremely large 95% CI (almost twice as large as θ estimates). When populations are weakly structured with high rates of gene flow, θ provides a more accurate estimator than ρ (Balloux and Goudet 2002). Therefore, ρ -values were not used in further analyses. Genetic differentiation between population pairs was not significantly correlated with distance [ln (km)] (Mantel test $r = 0.789$, $P = 0.167$).

Bayesian clustering of all feral cats suggested the presence of three clusters (ln L = -2037; not shown) with membership in each cluster ranging from $q = 0.35$ to $q = 0.95$ (mean $q = 0.75$). Females were split into two clusters (ln L = -818) with membership ranging from $q = 0.59$ to $q = 0.98$ (mean $q = 0.88$). Female cats from HAVO and NMK created one cluster while WMK females created another. For males, a single cluster ($K = 1$) was the most likely (ln L = -1261).

Assignment tests correctly assigned the majority of individuals to the area in which they were trapped (GeneClass2: 62.4%; Structure: 56.4%). Considering the two closest populations (18 km apart), 23.1% (GeneClass2) and 10.3% (Structure) of individuals captured in NMK were assigned to WMK. Of those misassigned in WMK, 25.8% (GeneClass2) and 41.9% (Structure) were assigned to HAVO (Table 8.3). In GeneClass2, two individuals were identified as samples that could not be grouped into a population, and thus were likely to be migrants from an unsampled population (Cegelski et al. 2003). Another two individuals were identified in both tests as being likely migrants (probability of assignment (P) > 0.80) from WMK into both HAVO and NMK.

Effective population size

The overall $Ne(D)$ estimates for HAVO, NMK, and WMK (with 95% CI) were 24.2 (19.2–54.2), 35 (27.5–46.4), and 26.9 (21.2–35.4), respectively. In HAVO, the $Ne(D)$ estimates from 2000–2002 samples ($n = 6$) and 2003–2005 samples ($n = 9$) increased from 6.4 (4.0–12.2) to 18 (10.0–55.3), respectively. In NMK, the $Ne(D)$ estimates from 2004 samples ($n = 18$) and 2005 samples ($n = 21$) increased from 29.2 (19.1–54.2) to 30.8 (20.8–53.5), respectively. Although not significant, these estimates may suggest population growth for HAVO and NMK (Figure 8.2).

Gene flow and migration rate

Nm estimates suggested high gene flow between feral cat populations with the effective number of migrants per generation ranging between 6.3 and 10.6 (Table 8.4). Number of migrants per generation was lowest between HAVO and NMK and highest in the adjacent populations on Mauna Kea. The mean posterior probabilities of migration rates showed that the majority of individuals were native to their capture locations in all populations with the most originating in WMK (0.708–0.927) (Table 8.5). There was a relatively high degree of migration between populations from WMK to NMK ($m = 0.248$; 95% CI = 0.032–0.325) and HAVO ($m = 0.176$; 95% CI = 0.034–0.312). In contrast, migration rates from both HAVO and NMK into the other populations were very low with the smallest migration rate from HAVO to WMK ($m = 0.015$; 95% CI = 0.0004–0.048).

Table 8.1. Measurements of genetic diversity for feral cats on Hawai`i Island. Expected (H_E) and observed (H_O) heterozygosities, average number of alleles per locus (A) and the unbiased inbreeding coefficient, f , were estimated using seven microsatellite loci in the three feral cat populations sampled in Hawai`i Volcanoes National Park (HAVO), North Mauna Kea (NMK) and West Mauna Kea (WMK).

Locus	HAVO ($n=15$)				NMK ($n=39$)				WMK ($n=31$)			
	H_E	H_O	A	f	H_E	H_O	A	f	H_E	H_O	A	f
FCA441	0.550	0.600	4	-0.091	0.489	0.564	5	-0.154	0.446	0.516	4	-0.158
FCA742	0.698	0.800	8	-0.147	0.786	0.744	9	0.054	0.778	0.774	7	0.006
FCA733	0.883	0.867	8	0.019	0.833	0.949	9	-0.139	0.850	0.806	8	0.051
FCA723	0.890	0.867	10	0.027	0.769	0.744	14	0.032	0.829	0.806	15	0.027
F124	0.888	0.800	11	0.099	0.800	0.795	9	0.007*	0.826	0.774	12	0.063*
FCA740	0.507	0.533	3	-0.052	0.610	0.513	5	0.159	0.454	0.452	6	0.005
FCA731	0.879	0.800	9	0.089	0.862	0.757	11	0.122*	0.829	0.296	11	0.643*
Mean	0.756	0.752	7.57	-0.008	0.736	0.723	8.86	0.012	0.716	0.632	9.00	0.091

*Values indicate a significant ($P < 0.002$) deviation from Hardy-Weinberg equilibrium after Bonferroni corrections.

Table 8.2. Assignment of feral cats to populations on Hawai`i Island. Assignment of feral cats to the three sampled populations (capture locations) in Hawai`i Volcanoes National Park (HAVO), North Mauna Kea (NMK) and West Mauna Kea (WMK) using GENECLASS2 (Piry et al. 2004) and STRUCTURE assignment tests (Pritchard et al. 2000).

Location	<i>n</i>	Number assigned to population					
		HAVO		NMK		WMK	
		GENECLASS2	STRUCTURE	GENECLASS2	STRUCTURE	GENECLASS2	STRUCTURE
HAVO	15	11	12	2	1	2	2
NMK	39	7	13	22	22	9	4
WMK	31	8	13	3	4	19	14

Table 8.3. Genetic structure and connectivity of feral cat populations on Hawai`i Island. Pairwise $F_{ST}(\theta)$ estimates above the diagonal, estimated number of migrants per generation (N_m) between populations below the diagonal, and expected heterozygosities (H_E) along the diagonal from three feral cat populations sampled in Hawai`i Volcanoes National Park (HAVO), North Mauna Kea (NMK) and West Mauna Kea (WMK), Hawai`i.

	HAVO	NMK	WMK
HAVO	0.756	0.038	0.028
NMK	6.30	0.736	0.023
WMK	8.81	10.56	0.716

Table 8.4. Migration rates of feral cats on Hawai`i Island. Means of the posterior distributions of the migration rate (with 95% confidence intervals) into each of the three feral cat populations sampled in Hawai`i Volcanoes National Park (HAVO), North Mauna Kea (NMK) and West Mauna Kea (WMK), Hawai`i. Migration rates are estimated as the proportion of individuals in column populations that originated from populations in rows. Values along the diagonal are the proportions of individuals within a population derived from that population.

	HAVO	NMK	WMK
HAVO	0.708 (0.667–0.835)	0.116 (0.004–0.277)	0.176 (0.034–0.312)
NMK	0.015 (0.0004–0.048)	0.737 (0.668–0.953)	0.248 (0.032–0.325)
WMK	0.038 (0.001–0.104)	0.035 (0.002–0.127)	0.927 (0.843–0.986)

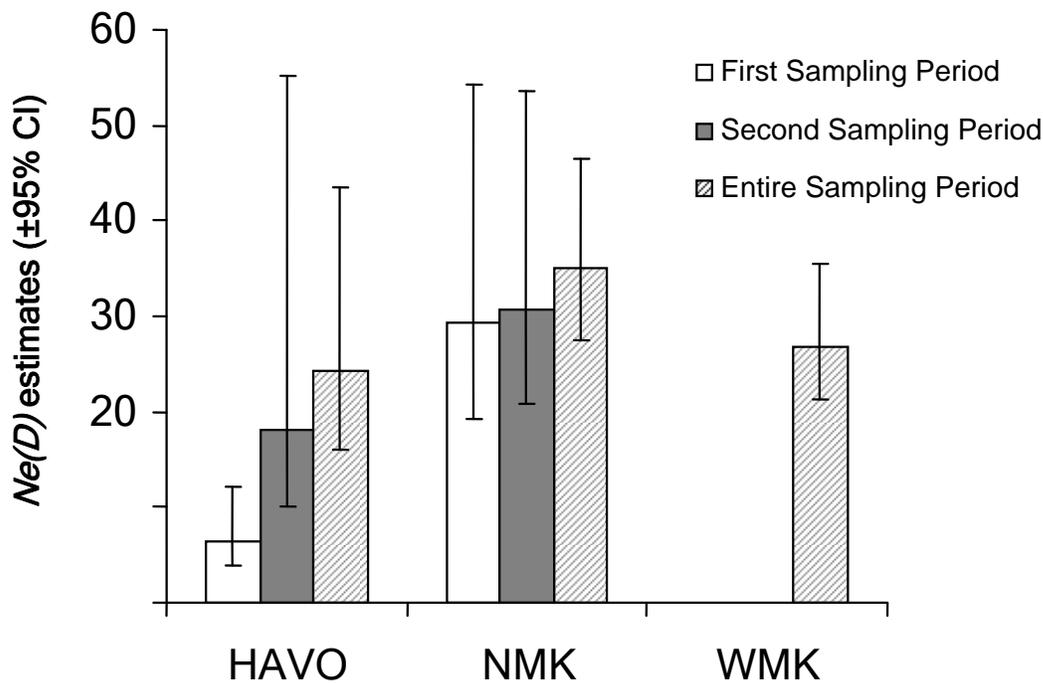


Figure 8.2. Effective population size ($N_e(D)$) estimates for three feral cat populations sampled in Hawai'i Volcanoes National Park (HAVO), North Mauna Kea (NMK) and West Mauna Kea (WMK), Hawai'i, with corresponding 95% confidence intervals for first (white bars), second (shaded bars) and entire (hatched bars) sampling periods. The first and second sampling periods for HAVO were between 2000–2002 and 2003–2005. The first and second sampling periods for NMK were 2004 and 2005, respectively. The entire sampling period for WMK was in 2003.

the other populations were very low with the smallest migration rate from HAVO to WMK ($m = 0.015$; 95% CI = 0.0004–0.048).

Sex-biased dispersal

Assignment t -test results supported male-biased dispersal and female philopatry. Relatedness of males and females differed significantly ($P < 0.05$) and the mAI_c of males among populations was significantly lower than that of females ($mAI_c = -0.900, 1.23, P = 0.007$, respectively). The estimate of f was significantly higher for males ($f = 0.085$) than for females ($f = -0.007$; $P = 0.008$) and the average θ estimate across the populations for males ($\theta = 0.013$) was significantly lower than that calculated for females ($\theta = 0.053$; $P = 0.028$).

We found no differences in relatedness between female and male cats within and among populations (Figure 8.3). Mean r_{xy} values suggested little to no relatedness within and

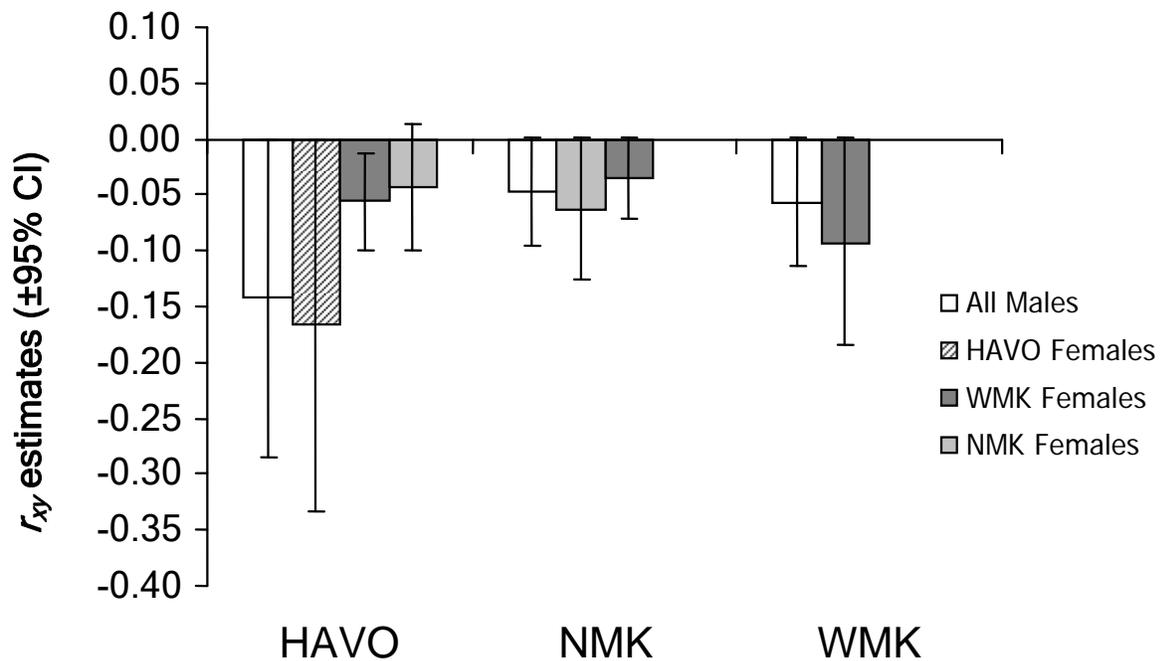


Figure 8.3. Estimated pairwise relatedness (r_{xy} – Lynch and Ritland 1999) with 95% confidence intervals of male feral cats (white bars) and female feral cats (hatched and shaded bars) within and among three populations from Hawai'i Volcanoes National Park (HAVO), North Mauna Kea (NMK) and West Mauna Kea (WMK), Hawai'i.

among populations (as shown by the negative estimates) and both male and female cats in HAVO were less related to each other than cats in other populations, most likely due to the larger sampling area of HAVO compared to the other populations. Among populations, pairwise r_{xy} values were significantly correlated with geographic distance ($P < 0.0001$), having a greater inverse correlation for female cats (Mantel test $r = -0.161$) than for all cats (Mantel test $r = -0.131$) and very little inverse correlation for male cats (Mantel test $r = -0.095$). Within populations, r_{xy} values and distance were not correlated for male cats ($P > 0.5$ in all populations). For females, r_{xy} values declined significantly with distance in HAVO and WMK (HAVO, Mantel test $r = -0.576$, $P = 0.005$; WMK, Mantel test $r = -0.386$, $P = 0.002$), although not in NMK (Mantel test $r = -0.05$, $P = 0.587$).

The most related individuals ($r_{xy} = 0.86$), an adult and a juvenile female, were captured within 15 d of each other at a distance of 1.4 km in WMK. It is likely this was a mother-daughter pair. The most related individuals ($r_{xy} = 0.62$) separated by the greatest distance (65.9 km) were two females captured in HAVO and NMK and were likely siblings or a mother-daughter pair.

e. Discussion

Genetic structure suggests that feral cats on Hawai`i Island exhibit long-distance dispersal between populations and their movements are not inhibited by current control efforts or barriers such as extensive lava flows. As expected, given their European ancestry, the genetic diversity of feral cats we examined in Hawai`i ($A =$ from 7.57 to 9.00, $H_0 = 0.70$) was similar to European domestic cats ($A = 14.2$, $H_0 = 0.70$; Pierpaoli et al. 2003) but was greater than that reported for feral cats recently introduced from France to a sub-Antarctic Island ($A =$ from 3.67 to 7.00, $H_0 = 0.53$; Pontier et al. 2005) and also for cat colonies in France ($A =$ from 4.38 to 7.78, $H_0 = 0.61$) that experience low dispersal rates due to barriers such as heavy-traffic roads (Say et al. 2003). We also found very little genetic differentiation ($\theta = 0.028$) between populations and dispersal was not spatially limited. Generally, F_{ST} values below 0.05 suggest high levels of gene flow and population connectivity (Thulin et al. 2006). Therefore, we conclude that the three populations were not founded independently, there was rapid colonization from initial founders due to high growth rates, and ongoing gene flow has occurred (Abdelkrim et al. 2005; Pontier et al. 2005). Apparently, feral cats in Hawai`i have not differentiated markedly from their initial founders (Pontier et al. 2005). Although we have no definitive evidence to rule out the possibility that domestic house cats were recruited into nearby feral populations, all of the cats we captured lacked diversity in coat coloration and had reverted back to pelage characteristics similar to European wildcats (*Felis silvestris*; Lowe et al. 2000; Beaumont et al. 2001).

We found some evidence of inbreeding in the Mauna Kea populations ($f > 0.09$) and between all populations ($f = 0.03$), although low compared to estimates of isolated feral cat populations in urban France ($f = 0.14$; Say et al. 2003) and on a sub-Antarctic island ($f > 0.11$; Pontier et al. 2005). Inbreeding may be the result of factors that cause low population density, such as kitten mortality due to feline leukemia virus, feline immunodeficiency virus, and toxoplasmosis (Molsher 1999), all of which have been documented in feral cats on Mauna Kea (Danner et al. 2007). Inbreeding may also reduce fitness and resistance to diseases (Coltman et al. 1999), mating success (Slate et al. 2000), and juvenile survival (Coltman et al. 1998).

Despite the positive inbreeding coefficients, WMK and NMK, separated by the shortest distance (18 km), had high levels of gene flow between them with 10.6 cats migrating per generation. Kaeuffer et al. (2004) estimated that the mean generation time for feral cats in France was 3.38 yr. This suggests that > 10 cats migrated between WMK and NMK and successfully reproduced in 3 yr. Given the home range sizes of feral cats reported from similar environments in Australia (Edwards et al. 2001), it is not surprising that cats disperse long distances. Often populations in close proximity receive a greater number of migrants than populations farther apart (Whitlock and McCauley 1999), and accordingly, less gene flow occurred between Mauna Kea and HAVO. It is surprising, though, that an estimated 6.3 to 8.8 cats per generation may have migrated between these populations given the harsh environment between these areas. However, our estimates of indirect gene flow are presented only as an index of relative measures of connectivity between populations and some assumptions of this analysis may have been violated (Whitlock and McCauley 1999; Cegelski et al. 2003).

Similar to other carnivores, the spatial organization of cats is determined by the abundance and distribution of available prey and receptive mates and the proximity to human habitation (Pontier et al. 1995). The density of cats may be higher on Mauna Kea, particularly WMK, than in HAVO due to abundant prey (Hess et al. 2007) which may facilitate increased reproduction, survival, and dispersal to other locations. Accordingly, migration rates showed the majority of dispersing cats originated and immigrated from WMK into the other populations with little migration occurring into WMK. Also, only 45 to 61% of individuals captured in WMK were correctly assigned to their origin. The 'misassigned' individuals were likely migrants from an unsampled population or offspring of migrants (Cegelski et al. 2003; Rollins et al. 2006). In fact, two migrants were captured in both HAVO and NMK and assigned to WMK. This suggests that cats may not be able to easily integrate with some residents and may disperse long distances to fill empty territories created by current predator removal efforts. Also, cats may disperse more during the seabird breeding season to depredate these burrow-nesting endangered species (Hu et al. 2001).

Mauna Kea populations had significantly more successfully breeding individuals than in HAVO as shown in the larger N_e estimates (DeYoung and Honeycutt 2005). Because we trapped consistently between 2000 and 2005, our samples had overlapping generations and immediate evaluation of N_e would have been difficult using more common temporal methods that require at least two non-overlapping generations (Waples 1989). Bartley's linkage disequilibrium method (Bartley et al. 1992) has an advantage in that population trends can be evaluated

(Leberg 2005) and only one sampling period is required regardless of generation length (Bartley et al. 1992). Both HAVO and NMK apparently experienced population expansion with increasing N_e estimates, indicating insufficient control efforts between trapping periods. However, our sample sizes were small and our results should be used only to evaluate control strategies, as sample sizes over 90 may be necessary to obtain precise estimates of N_e when using the linkage disequilibrium method (Bartley et al. 1992). Also, Mauna Kea populations may be experiencing introgression from adjacent unsampled populations which would explain both the increase in N_e estimates and misassigned individuals (Roman and Palumbi 2003; Spencer et al. 2005).

Although our results support high gene flow, population structuring was evident in the Bayesian clustering results that showed three clusters of cats. Cats captured in HAVO had a greater proportion of individuals correctly assigned to that population whereas individuals in Mauna Kea had more ambiguous assignments. The discrepancies between genetic structure and assignment tests could be due to several reasons such as the distance and terrain separating populations, predator control efforts reducing encounter rates, and sampling individuals that have mixed or indistinct ancestry (Cegelski et al. 2003). When analyzing individual sexes, population structuring suggested two clusters of females and only one of males. Also, assignment t -tests and relatedness estimates supported male-biased dispersal which is common in many mammal species (Goudet et al. 2002). More unusual was the similar relatedness we found between males and females. However, relatedness of females, but not males, decreased significantly with geographic distance further supporting female philopatry. Similar to other research, we found that sex-biased dispersal estimates derived using genetic-based assignment techniques mirrored radio-telemetry findings, and did not require as much field effort (DeYoung and Honeycutt 2005). Goudet et al. (2002) suggested that short-distance dispersal may be sex-biased to avoid inbreeding. Alternatively, long-distance dispersal may be a means to colonize empty patches and is unlikely to be sex-biased. In fact, Spong and Creel (2001) reported that approximately 20% of female lions (*Panthera leo*) in Africa emigrate to new territories. This is consistent with our finding that the most related individuals separated by the furthest distance were two females.

Our results can be used to design an effective plan for feral cat control. Although complete eradication of feral cats has occurred on several small islands (Nogales et al. 2004), Hawai`i Island would require large-scale control programs which are typically difficult to implement (Pontier et al. 2005). One major limitation for control efforts is the ability of the

invasive species to disperse from neighboring populations and recolonize (Abdelkrim et al. 2005). The genetic structure of feral cats in Hawai`i indicates great dispersal ability, therefore, control will be very difficult and recolonization seems highly likely. The reproductive biology and life history of the invasive species can also determine the ease of control and potential for recolonization (Myers 2000). Female cats reach sexual maturity between 6 and 8 months and males between 8 and 10 months and can breed two to three times a year. However, reproduction can be delayed (Jones and Coman 1982; Say et al. 1999) as shown by Bester et al. (2002) who reported that a drastic decrease of feral cats following eradication efforts on sub-Antarctic Marion Island caused a decline in pregnancy rates and fecundity, possibly due to a lower encounter rate between sexes. Control efforts need to minimize gene flow to no more than one migrant per generation in order to decrease genetic diversity (Wright 1969) and thus create small or fragmented populations in which inbreeding is more likely to occur. Targeting the source population for control is important when the animal disperses long-distances on unknown pathways (Rollins et al. 2006). On Hawai`i Island, control efforts on Mauna Kea may benefit endangered wildlife at other locations by reducing dispersing feral cats. Also, the sex dispersing the most could be targeted for control (Rollins et al. 2006); however, we do not recommend targeting male feral cats as both sexes exhibited some dispersal.

To minimize risk of failure in controlling invasive species and reduce management cost and loss of native species, genetic monitoring of the invasive species should be a preliminary step in the management process in order to assess the effectiveness of control efforts (Robertson and Gemmell 2004; Abdelkrim et al. 2005; Rollins et al. 2006). For example, a decrease in genetic diversity, increase in inbreeding, and decrease in effective population size would all indicate successful control strategies (DeYoung and Honeycutt 2005). Assignment tests can be used to identify captured individuals as either control survivors or recolonizers following control efforts (Robertson and Gemmell 2004). Management of other invasive vertebrates may benefit by employing these population genetic tools.

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VIII. Adaptive Strategies for Reducing Predation on Endangered Hawaiian Birds

Steven C. Hess, and Paul C. Banko, U.S. Geological Survey Pacific Island Ecosystems Research Center, P.O. Box 44, Kīlauea Field Station, Hawai`i National Park, HI, 96718, USA

Heidi Hansen¹, Hawai`i Cooperative Studies Unit (PACRC, UH Hilo), U.S. Geological Survey Pacific Island Ecosystems Research Center, Kīlauea Field Station, P.O. Box 44, Hawai`i National Park, HI, 96718, USA

¹ Current address: Department of Land and Natural Resources, Division of Forestry and Wildlife, 19 E. Kawili St., Hilo, HI 96720, USA

a. Introduction

Feral cats (*Felis catus*) became abundant in forests of the Hawaiian Islands soon after their introduction in the late 1700s (Rothschild 1893; Perkins 1903; King 1984). They were subsequently reported to be notorious predators of birds that contributed to the decline and extinction of some Hawaiian species (Perkins 1903; Berger 1981; Ralph and van Riper 1985; Stone 1985; Snetsinger et al. 1994). There is strong evidence that feral cats in Hawai`i currently prey on endangered forest birds (Laut et al. 2003; Hess et al. 2004), terrestrial birds that nest near the ground (Kowalsky et al. 2002), as well as nesting colonial seabirds (Smith et al. 2002; Hess et al. 2007). Feral cats range throughout the Hawaiian Islands from high-density colonies near coastal areas where pet cats are frequently abandoned and maintained by people (Winter 2003), to remote, low-density populations in montane forests and subalpine areas of Maui (Simons 1983) and Hawai`i Island (Hu et al. 2001). Despite the long history of feral cats in Hawai`i, there has been little research to provide strategies to improve control programs and reduce their depredation of endangered species.

Many research institutions and wildlife management agencies mandate daily checks of live traps, the primary method of trapping in Hawai`i, due to care and treatment concerns (Stone 1995). Consequently, this method requires enormous amounts of labor. Wary cats may be reluctant to enter live traps, especially if there are associated human scents, which further reduces captures (Marks 1996). Moreover, many traps may be located in places where cats are never captured. Trap placement relative to different habitats, elevation, and proximity to roads or other landscape features may affect the capture of both target and non-target species. In addition to controlling cats within core conservation areas, buffer zones of > 2 km may also be

necessary to restrict immigrants, which are primarily males (Goltz et al., in press). All of these considerations may limit the efficiency of managing larger landscapes inhabited by vulnerable endangered species. Efficiency of cat control may be improved by taking advantage of certain landscape features to increase overall capture rates, the number of cats captured per trap, and to reduce captures of non-target species.

Non-target species such as rats (*Rattus* spp.) and small Indian mongooses (*Herpestes auropunctatus*), while not benign, may not pose as urgent a threat to some endangered species as feral cats. For example, mongooses depredate ground-nesting birds (Banko 1992), but are poor climbers and therefore seldom threaten tree-nesting passerines (Hays and Conant 2007). However, non-target species may rapidly consume bait before cats encounter traps, necessitating frequent bait replacement. Many non-target small mammals may also be more abundant and easier to attract and capture than feral cats, thereby interfering with the capture of cats by pre-empting single-capture traps. The additional effort required to control a suite of exotic species may also add unnecessary costs in comparison to strategies focused on single species.

Hansen et al. (2007) determined that feral cat control efforts focused on Mauna Kea may benefit endangered wildlife at other locations on Hawai`i Island by reducing dispersing individuals. Our objective was to develop an adaptive management strategy based on a predictive model of landscape features on Mauna Kea such as habitat, elevation, and proximity to roads that affect the number of feral cats captured at each trap. We sought to identify effective trap locations that can be used to reliably capture cats, and if possible, to reduce captures of non-target mongooses. The total number of captures per trap may be more informative than capture rates (captures per trap-night) because capture rates may decline as a result of reduced population density. This work was part of a larger study of feral cats in Hawai`i that investigated diseases (Danner et al. 2007), diet (Hess et al. 2004; 2007a, b), population genetics (Hansen et al. 2007), and home range (Goltz et al., in press). The overall goal of this research was to provide better information on the impacts of feral cats on endangered species as well as strategies for developing and improving control methodology.

b. Study Area

Study areas were located on North Mauna Kea (NMK; 19° 54' N, 155° 27' W), and West Mauna Kea (WMK; 19° 49' N, 155° 36' W), a dormant volcano on Hawai`i Island (Figure 9.1). Subalpine woodland occurs between 1,750–3,000 m in elevation (Hess et al. 1999). Large areas of woodlands were dominated by māmane (*Sophora chrysophylla*) or naio (*Myoporum*

sandwicense) trees, while extensive shrublands were dominated by pūkiawe (*Leptecophylla tameiameia*), and `ā`ali`i (*Dodonaea viscosa*). Overall canopy cover averaged 30%, canopy height was generally short (3–8 m) with interspersed lower-stature shrubs, and other large areas were dominated by shrubs. Small patches of barren ground > 30 m² were interspersed throughout other habitats. The semi-arid aspect of subalpine woodland is due to severely drained volcanic substrates and rain shadow effects. Rainfall averaged 511 mm and temperature averaged 11.1° ± 1.5° C annually (Juvik et al. 1993). The study area contained no natural sources of perennial standing water.

c. Methods

We captured feral cats (*Felis catus*) and small Indian mongooses (*Herpestes auropunctatus*) on WMK from April 2002–August 2003 and on NMK from September 1998–July 2005 as part of efforts to restore Palila (*Loxioides bailleui*), an endangered Hawaiian forest bird. Tomahawk® model 106 live traps (23 x 23 x 85 cm) were distributed at 150 m intervals primarily along 14 transects at 152 different locations (Figures 9.1–9.2). The trapping areas encompassed approximately 50 km² on WMK and 14 km² on NMK. We covered traps with a layer of plastic to protect trapped cats from rain and cold, and placed a cloth rag inside for bedding. We used canned cat food, sardines, and mackerel as bait, checked set traps daily, and wired traps open when unattended. Feral cats and mongooses were euthanized according to University of Hawai'i IACUC Protocol.

We determined proximity to roads from a GIS layer of roads and elevation from a 10 m digital elevation model at each trap (ESRI 1999). All GIS layers used North American Datum 1983 and were projected in Universal Transverse Mercator zone 5. Habitat type (māmane, naio, shrubland, or barren) at each trap was determined from a classified Landsat Enhanced Thematic Mapper image from 5 February, 2000 (USGS, unpubl. data). Image resolution was 30 m². We randomly chose 337 of 505 40-m² field plots to derive the thematic classification and used the remainder to assess its accuracy. Within māmane dominated woodlands, classification correctly predicted the presence of māmane trees in 96.7–100% of plots. Within mixed woodlands, comprised mainly of naio and māmane trees, the classification characterized 94.4–100% of plots correctly for both species. In shrubland communities, the model assigned pūkiawe shrubs to 71.4–73.3% of plots correctly.

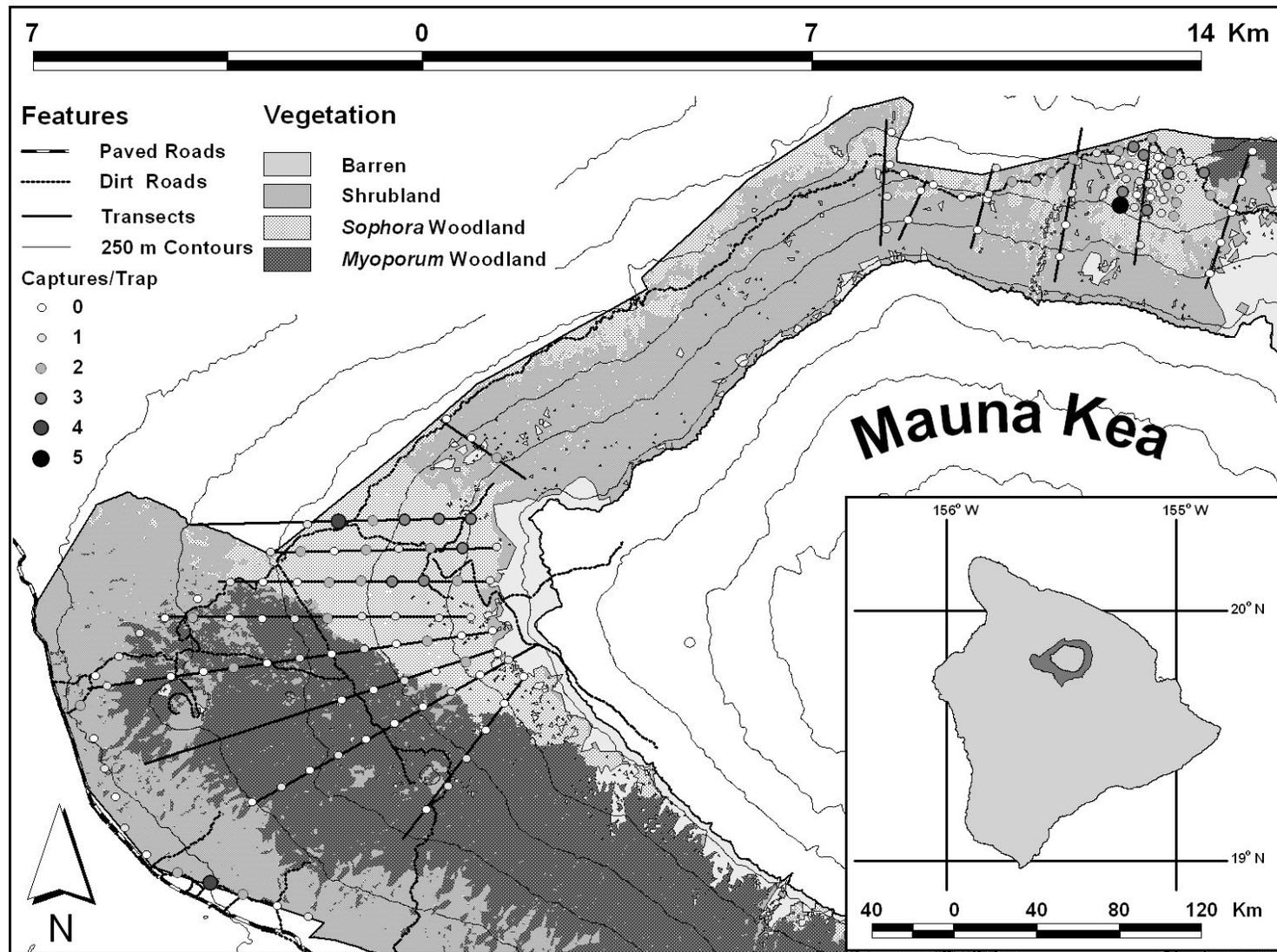


Figure 9.1. Feral cat (*Felis catus*) captures within endangered species habitat on Mauna Kea, Hawai`i, 2002–2005.

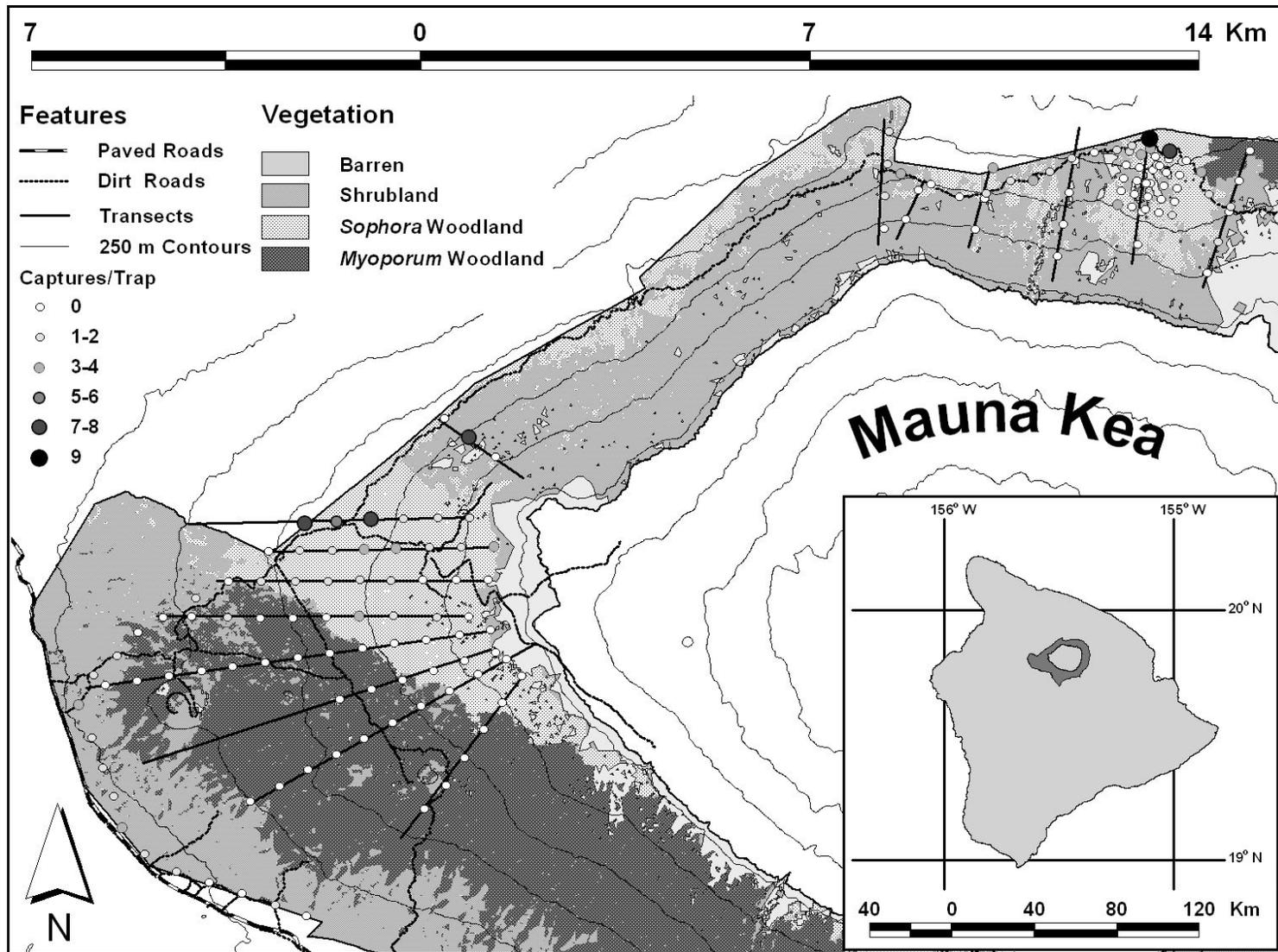


Figure 9.2. Small Indian mongoose (*Herpestes auropunctatus*) captures within endangered species habitat on Mauna Kea, Hawai`i, 2002–2005.

We used analysis of variance (PROC GLM; SAS Institute 1999) to determine if there were differences between the proximity of traps to roads among habitat types and between locations. We used log-link generalized linear models (PROC GENMOD; SAS Institute 1999) to examine the relationship between the number of cats captured at each trap and a suite of predictors. The number of cat captures per trap was treated as a Poisson distributed response variable after finding the mean (0.993) was similar to the variance (1.132). Predictors included effort (number of trap-nights), side of the volcano (NMK vs. WMK), habitat type, elevation (m), proximity to roads (m), and the number of mongooses captured. We did not examine proximity to human habitation because Goltz et al. (in press) determined that the home range of feral cats on WMK did not include areas of human influence; NMK was even more isolated. Models including all combinations of main effects were ranked with Akaike's Information Criterion with quasi-likelihood modifications and correction for small sample size (QAIC_c), recommended when count data may be overdispersed and $n/K < 40$ (Burnham and Anderson 1998). Interaction terms were not considered due to the relatively small sample size and large numbers of degrees of freedom fitting such interactions would require. We used orthogonal contrasts to estimate differences in captures between habitat types and side of the volcano from the final estimating model.

d. Results

A total of 5,888 traps nights were represented in the data. No cats were captured at 62 of the 152 trap locations (40.8 %). The maximum number of cats captured at a trap was five, which occurred only at one trap location. The number of cats captured was positively correlated with the number of mongooses captured in the same trap (Spearman correlation coefficient $\rho = 0.3123$, $P < 0.0001$). The number of mongooses captured at each trap was positively correlated with overall effort (Spearman correlation coefficient $\rho = 0.2260$, $P = 0.0051$). Proximity to roads was not related to habitat type ($F_{3, 144} = 0.28$, $P > 0.84$; Table 9.1) or side (NMK vs. WMK) of the volcano alone ($F_{1, 144} = 0.31$, $P = 0.58$), but was significantly related to the interaction between habitat type and side of the volcano ($F_{3, 144} = 3.82$, $P = 0.0114$). Habitat type was not balanced by elevation ($F_{3, 147} = 9.34$, $P < 0.0001$; Table 9.2). Barren ground and māmane woodland occurred at higher elevation than naio dominated woodland and shrublands (Tukey multiple comparison; $P < 0.05$).

Table 9.1. Number of captures and capture rates of feral cats (*Felis catus*) and small Indian mongooses (*Herpestes auropunctatus*), and mean elevation and proximity to roads by side (North vs West slope) of the volcano for 152 traps on Mauna Kea, Hawai`i, 2002–2005.

Location	<i>n</i> Traps	Trap Nights	<i>n</i> Cats	Cats/ 100 Trap Nights	<i>n</i> Mongooses	Mongoose/ 100 Trap Nights	Mean Elevation (m ± SE)	Mean Road Proximity (m ± SE)
West	86	780	84	10.77	77	9.87	2330.4 ± 38.3	398.1 ± 38.0
North	66	5108	67	1.31	60	1.17	2385.6 ± 20.7	417.8 ± 51.3
Total/Mean	152	5888	151	2.56	137	2.33	2354.4 ± 23.5	406.7 ± 30.9

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Table 9.2. Number of captures and capture rates of feral cats (*Felis catus*) and small Indian mongoose (*Herpestes auropunctatus*), and mean elevation and proximity to roads for 152 traps by habitat type on Mauna Kea, Hawai`i, 2002–2005.

Habitat	<i>n</i> Traps	Trap Nights	<i>n</i> Cats	Cats/ 100 Trap Nights	<i>n</i> Mongooses	Mongoose/ 100 Trap Nights	Mean Elevation (m ± SE)	Mean Road Proximity (m ± SE)
Barren	9	449	7	1.56	0	0.00	2478.2 ± 92.2	475.6 ± 67.4
Māmane	62	2940	87	2.96	84	2.86	2477.4 ± 27.9	398.9 ± 47.2
Naio	27	269	14	5.21	4	1.49	2262.0 ± 36.7	446.2 ± 74.7
Shrubland	54	2230	43	1.93	49	2.20	2238.6 ± 46.4	384.3 ± 56.3
Total/Mean	152	5888	151	2.56	137	2.33	2354.4 ± 23.5	406.7 ± 30.9

The two highest ranked generalized linear models $< 2 \Delta\text{QAIC}_c$ units included effort, habitat type, side (NMK vs. WMK), and the number of mongooses captured (Table 9.3). The highest ranked model included effort ($\chi^2 = 16.15$, $\text{df} = 1$, $P < 0.0001$), habitat type ($\chi^2 = 11.08$, $\text{df} = 3$, $P = 0.0113$), and side ($\chi^2 = 9.95$, $\text{df} = 1$, $P = 0.0016$). Effort appeared in the 15 highest ranked models, side in 11, and habitat type in five. The number of mongooses captured, which was correlated with effort, was not a significant predictor in the second highest ranked model [effort ($\chi^2 = 13.06$, $\text{df} = 1$, $P = 0.003$), habitat type ($\chi^2 = 8.95$, $\text{df} = 3$, $P = 0.0299$), side ($\chi^2 = 8.32$, $\text{df} = 1$, $P = 0.0039$), and number of mongooses captured ($\chi^2 = 2.27$, $\text{df} = 1$, $P = 0.1323$)]. Therefore, we chose model 1 as the final estimating model because it contained the essential predictors common to the five highest ranked models. The full model, ranked seventh, was $3.22 \Delta\text{QAIC}_c$ units. We assessed goodness of fit for the full model by comparing its deviance (152.38) with its asymptotic χ^2 with 143 df, and found $P > 0.28$; therefore, the null hypothesis of adequate fit was not rejected.

The mean number of cats captured per trap was significantly higher in māmane woodland than in naio (Contrast; $\chi^2 = 8.85$, $\text{df} = 1$, $P = 0.0029$; Figure 9.3), higher in māmane woodland than in shrubland (Contrast; $\chi^2 = 3.90$, $\text{df} = 1$, $P = 0.0482$), and higher in māmane woodland than the other three habitats combined (Contrast; $\chi^2 = 7.95$, $\text{df} = 1$, $P = 0.0048$). The mean number of cats captured per trap was also significantly higher on WMK than on NMK (Contrast; $\chi^2 = 9.95$, $\text{df} = 1$, $P = 0.0016$; Figure 9.4). The interpretation of negative parameters from log-link models indicates that the response was a fraction of the number relative to the reference category; positive parameters indicate the response was a multiple of the reference category. The parameter for māmane woodland from the final model was 0.378, therefore, $\exp(0.378) = 1.46$ times more captures per trap than traps located in shrubland, whereas traps in naio had 0.604 as many captures per trap as those located in shrubland (Table 9.4). Traps on NMK had 0.354 as many captures per trap as those located on WMK.

e. Discussion

Our overall modeling effort indicated that effort, habitat type, and side of the volcano (WMK vs. NMK) provided some ability to predict the number of cats captured in a trap. Other factors in the four highest ranked models included number of mongooses captured per trap, elevation, and proximity to roads. These factors appeared in one, two, and one model, respectively. Effort, habitat type, and side were highly supported. Models containing a positive

Table 9.3. Generalized linear models < 10 ΔQAIC_c for feral cat (*Felis catus*) captures from Mauna Kea, Hawai`i, 2002–2005.

Rank	<i>n</i>	K	Log Likelihood	QAIC _c	ΔQAIC_c	Deviance	df	\hat{c}	GOF <i>P</i> -value	Model
1	152	6	-132.92	263.06	0.00	156.08	146	1.07	0.27	Effort, habitat, side
2	152	7	-131.86	263.25	0.20	153.95	145	1.06	0.29	Effort, habitat, side, n mongooses
3	152	8	-131.63	265.07	2.01	153.50	144	1.07	0.28	Effort, habitat, side, elevation, road proximity
4	152	7	-132.85	265.12	2.07	155.94	145	1.08	0.25	Effort, habitat, side, elevation
5	152	8	-131.85	265.47	2.41	153.93	144	1.07	0.27	Effort, habitat, side, elevation, n mongooses
6	152	4	-136.65	265.75	2.70	163.54	148	1.11	0.18	Effort, side, n mongooses
7 ^a	152	9	-131.07	266.28	3.22	152.38	143	1.07	0.28	Effort, habitat, side, elevation, road proximity, n mongooses
8	152	5	-136.02	266.71	3.65	162.28	147	1.10	0.18	Effort, side, elevation, n mongooses,
9	152	5	-136.39	267.39	4.33	163.01	147	1.11	0.17	Effort, side, elevation, road proximity
10	152	3	-138.95	267.96	4.91	168.14	149	1.13	0.14	Effort, side
11	152	4	-138.50	269.23	6.17	167.24	148	1.13	0.13	Effort, side, elevation
12	152	6	-136.76	270.26	7.20	163.75	146	1.12	0.15	Effort, habitat, n mongooses
13	152	7	-136.29	271.58	8.52	162.82	145	1.12	0.15	Effort, habitat, road proximity, n mongooses
14	152	5	-138.86	272.04	8.99	167.96	147	1.14	0.11	Effort, habitat
15	152	7	-136.56	272.08	9.02	163.35	145	1.13	0.14	Effort, habitat, elevation, n mongooses
16	152	5	-139.36	272.98	9.92	168.96	147	1.15	0.10	Habitat, n mongooses

^aFull model

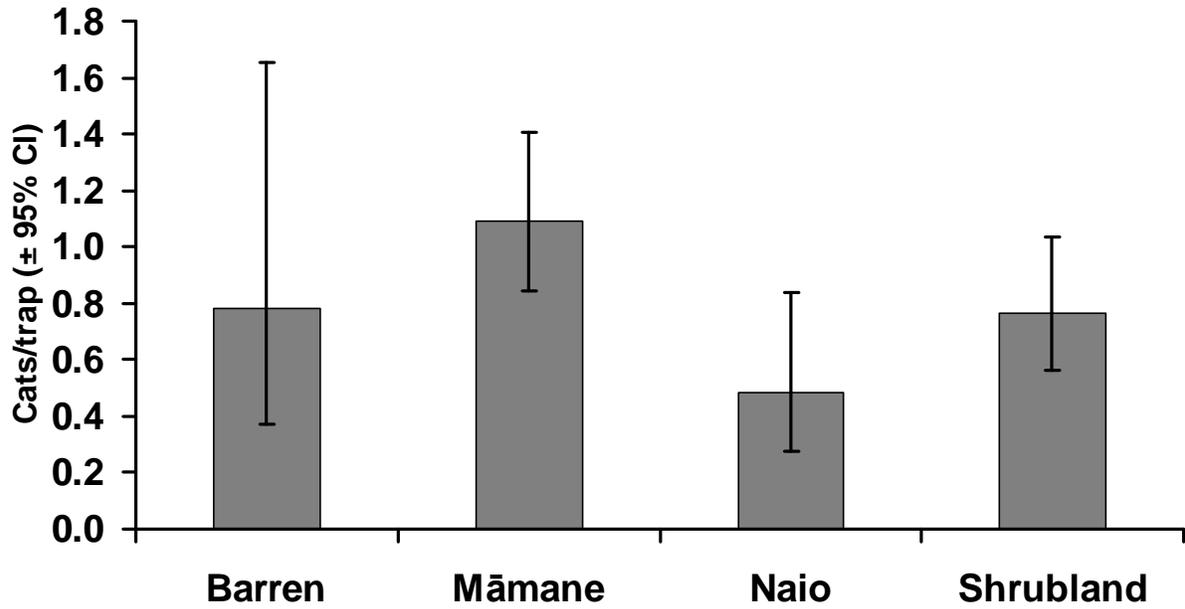


Figure 9.3. Least square means from a generalized linear model of the number of feral cat (*Felis catus*) captures per trap by habitat type from Mauna Kea, Hawai`i, 2002–2005.

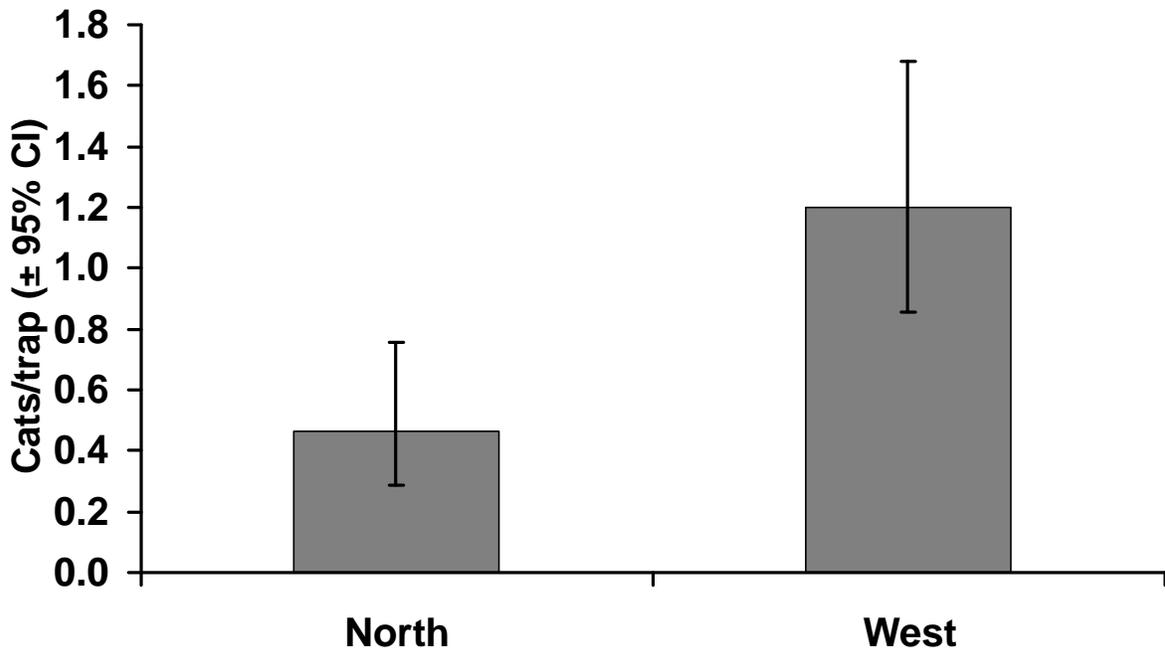


Figure 9.4. Least square means from a generalized linear model of feral cat (*Felis catus*) captures from the North and West slopes of Mauna Kea, Hawai`i, 2002–2005.

Table 9.4. Parameter estimates from the highest ranked generalized linear model of the number of feral cat (*Felis catus*) captures per trap from Mauna Kea, Hawai`i, 2002–2005.

Parameter	df	Exponent			95%		χ^2	P-value
		Estimate	(Estimate)	SE	95% LCI	UCI		
Intercept	1	-0.2098	0.811	0.177	-0.5556	0.1361	1.41	0.2345
Effort	1	0.012	1.012	0.0029	0.006	0.0175	16.15	< 0.0001
Habitat	3	--	--	--	--	--	11.08	0.0113
Barren	1	-0.038	0.963	0.4079	-0.8375	0.7614	0.01	0.9257
Māmane	1	0.378	1.460	0.1914	0.0031	0.7534	3.90	0.0482
Naio	1	-0.505	0.604	0.3176	-1.1274	0.1176	2.53	0.1119
Shrubland	0	0.000	1.000	0.000	0.000	0.000	--	--
Side	1	--	--	--	--	--	9.95	0.0016
North	1	-1.038	0.354	0.3291	-1.6832	-0.3931	9.95	0.0016
West	0	0.000	1.000	0.000	0.000	0.000	--	--

relationship with effort and number of cat captures provided a convenient check for plausibility because traps are unlikely to capture multiple cats (or mongooses) without greater effort. Our findings suggest that traps located in māmane woodland on WMK were more likely to capture multiple cats than other habitats and locations in the study areas.

More cats per trap were captured in māmane woodland than in naio even though the overall capture rate (cats/100 trap-nights) was high in naio. This is because cats were captured over extensive areas dispersed in naio woodland, but in māmane, multiple cats were frequently captured in many traps. Overall capture rates do not provide a good basis for comparison between habitats because the density of cats may have been reduced after extended periods of trapping, thereby reducing capture rate. The coarse habitat layer available to us did not address differences in microhabitat use by cats. Woody material, grasses and other structural aspects of habitat are likely to be a strong determinant of how cats and their prey interact and move

through their environment. More detailed measurement of habitat structure would provide better insights about what components are favored by cats.

Both capture rates and the number of captures per trap were higher on WMK than NMK. These findings corroborate population genetics research that determined there was a net dispersal of cats from WMK to NMK and Mauna Loa, which is over 50 km away (Hansen et al. 2007). A possible mechanism driving dispersal from WMK is that abundant birds, including nine introduced gamebird species on Mauna Kea, provide plentiful food resources (Hess et al. 2004; 2007a, b) necessary for high reproduction and survival, thereby creating a potential source population. Cats may reach high densities in such prime habitats and then disperse to other locations during seasonal food shortages to areas where birds are less abundant (Scott et al. 1986). Trapping in source areas may be an effective strategy to reduce dispersal to other locations (Robertson and Gemmell 2004).

Other factors supported in the highest ranking models included the number of mongooses captured at a trap, elevation and proximity to roads. The information contained in number of mongooses captured may have been redundant with effort and therefore provided little in the way of model improvement over the essential predictors. Elevation was also confounded with habitat type and therefore redundant. Proximity to roads appeared in the third and seventh ranked (full) models in conjunction with elevation, but proximity to roads was not a highly ranked factor overall. This may be due to several high elevation roads within māmane woodland on WMK where multiple cats per trap were captured. Mongooses, however, were captured in high elevation māmane woodland in close proximity to roads.

Unfortunately, the number of mongooses captured at a trap was positively related to the number of cat captures despite the fact that mongooses pre-empted traps; therefore, reducing interference by this non-target species through trap placement is unlikely. A potential solution is to use an alternate control method such as toxicants in the vicinity of trap sites. Standard rodent bait stations with Eaton's Bait Blocks® containing 0.005% diphacinone and fish flavorizer were found to be effective at controlling mongooses over small areas (Smith et al. 2000) and would effectively control bait-consuming rats as well.

One strategy to improve capture success is to deploy more traps in the vicinity of traps where cats have been captured and discontinue trapping (at least temporarily) at locations where cats have not been captured. Such a scenario may result in an active adaptive clustering of traps in areas where features of the landscape make cats more likely to repeatedly visit a

particular location due to travel routes, high prey density, habitat structure conducive to hunting, or other unknown reasons. In this respect, the trapping strategy might resemble an adaptive cluster sampling approach, where samples are taken from adjacent units until subjects are no longer encountered (Thompson 1992; 2004). The results of such efforts should be periodically reevaluated in such a sampling framework and efforts should be adjusted accordingly.

Although we found no other references to adaptive management exclusively involving predator control of feral cats, two relevant studies advocated adaptive management designs in recovery programs for rare birds (Innes et al. 1999; Keedwell et al. 2002). Innes et al. (1999) advocated adaptive management approaches for properly targeting intensive control efforts of key pests on mainland sites (i.e., where eradication is not feasible due to continuous recolonization). Keedwell et al. (2002) demonstrated the difficulty in evaluating the success of predator control regimes on the fledging success of endangered birds because of inherently small sample sizes. They recommended, however, altering standard trapping protocols over time, monitoring predator and prey abundance, and monitoring the relationship of these factors to the survival of the subject endangered species. We would also recommend evaluating the level of trapping effort or cost in such an analysis to determine the most effective management regimes.

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