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Locking Horns with Hawai'i's Non-native Ungulate Issues

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ABSTRACT: Conservation and management interests for sustained-yield hunting of non-native ungulates in Hawai'i have conflicted with the conservation of native biota for several decades. Hawaiian ecosystems evolved in the absence of large mammals and all currently hunted animals in Hawai'i are non-native species. The best-studied aspects of Hawai'i's ungulates have dealt primarily with direct negative effects on native biota in natural areas, but there has been little research in population dynamics for sustained-yield management. Ungulates have been removed from approximately 750 km² throughout the Hawaiian Islands to protect these natural areas, thereby reducing the amount of land area available for hunting activities and the maintenance of game populations. At the same time, unauthorized introductions of additional wild ungulate species between Hawaiian Islands have recently increased in frequency. The majority of hunting activities are of feral domestic livestock species for subsistence purposes, which typically do not generate sufficient revenue to offset costs of game management. Moreover, bag limits and seasons are generally not determined from biological criteria because harvest reporting is voluntary and game populations are rarely monitored. Consequently, ungulate populations cannot be managed for any particular level of abundance or other objectives. Research and monitoring which emphasize population dynamics and productivity would enable more precisely regulated sustained-yield game management programs and may reduce potential conflicts with the conservation of native biota.

KEY WORDS: eradication, Hawai'i, hunting, invasive species, island conservation, non-native ungulates, ungulates

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INTRODUCTION

The Hawaiian Islands, like many other isolated oceanic islands, developed in the complete absence of large herbivorous mammals (Carlquist 1970). The discovery of the Hawaiian Archipelago by ocean-voyagers and the introductions of several mammal species forever altered the ecosystems of these islands. Although thorn-like prickles developed in a group of at least 10 Hawaiian plant species, probably as a defense against Hawai'i's principal vertebrate herbivores a diverse group of now-extinct flightless geese and goose-like ducks (Olson and James 1982, Givnish et al. 1994, Duncan et al. 2013), other plants that evolved on these islands had no specific physical or chemical defenses against mammalian herbivory (Carlquist 1970, Bowen and Van Vuren 1997). Further, porous volcanic soils were not well-suited against disturbance caused by hoof action (Yocom 1967).

Polynesians brought the first ungulate species to the Hawaiian Islands in sailing canoes. These were domestic pigs (*Sus scrofa*) known as *pua'a*, which originated from Island Southeast Asia (Larson et al. 2005, Larson et al. 2007). Archaeological evidence documented *pua'a* more than 1,000 years ago from permanent Polynesian settlements on the islands of O'ahu (Pearson et al. 1971), Moloka'i (Kirch and Kelly 1975, Kirch 1982), and Kaua'i (Burney et al. 2001). Both skeletal remains and early historic observers indicated that *pua'a* were smaller than contemporary Hawaiian feral pigs, weighing only 27-45 kg (Ziegler 2002), and were close commensals that did not stray far from human settlements (Maly 1998). It was not until European explorers discovered the Hawaiian Archipelago and initiated another wave of mammalian introductions that larger European swine interbred with *pua'a*, the first being a boar and a sow

brought to the island of Ni'ihau by Captain James Cook in 1778 (Tomich 1986). Swine repeatedly interbred with multiple introduced domestic varieties and European wild boars, escaping from captivity to become the most abundant large mammal throughout the Hawaiian Islands.

Pigs were only one of several introduced ungulate species that became widespread after Europeans colonized the islands. Viewing remote islands as strategic resupply outposts for ships on worldwide voyages, explorers introduced familiar livestock such as cattle (*Bos taurus*), domestic goats (*Capra hircus*), and sheep (*Ovis aries*) throughout islands of the Pacific region. Captain James Cook brought goats to Hawai'i in 1778-1779 (Tomich 1986). Captain George Vancouver strategically liberated livestock throughout the Hawaiian archipelago in 1793 and 1794 and meticulously documented these introductions (Tomich 1986). Vancouver was able to convince King Kamehameha I, who had recently unified his authority over Hawai'i Island, to enforce a 10-year *kapu*, which forbade the killing of livestock. Vancouver wrote that the *kapu*, "...cannot fail to render the extirpation of these animals a task not easily to be accomplished." Livestock became feral and proliferated during this period without any predators or competitors. Sheep were reported at the summit of Mauna Kea, the highest peak in the Pacific Ocean, only 32 years after their introduction (Vancouver 1798:64).

Cattle brought to Hawai'i Island by Vancouver in 1793 were also protected by royal order for almost 30 years and allowed to roam freely (Kramer 1971). By 1822, "immense herds" of wild cattle occupied the Mauna Kea area (Ellis 1917). Between Kawaihae and Mauna Kea, the destruction of the once densely forested Waimea Plains was described by an anonymous writer just 63 years after their introduction: "It is in the memory

of many foreigners now living there when the whole of these plains were covered with a thick wood, to the very edge of the slope. Where hardly a tree is to be seen for miles, we were informed by an old resident, that twenty five years ago he lost himself with his team in the woods.” (Baldwin and Fagerlund 1943).

Among the wild ungulates introduced to Hawai‘i that had never been domesticated were axis deer (*Axis axis*), which are native to India, Sri Lanka, and Nepal (Graf and Nichols 1966). Axis deer from India were given to King Kamehameha V in 1867 and released in early 1868 (Kramer 1971). Several deer from Moloka‘i were moved to O‘ahu before 1898, and to L na‘i in 1920. Axis deer were later released on Maui in 1959 where they have become widespread (Anderson 2003). European mouflon sheep (*Ovis gmelini musimon*) from the Mediterranean Islands were first introduced to the Island of L na‘i in 1954 for sport hunting (Tomich 1986). Feral domestic sheep were intentionally hybridized with mouflon and released on Mauna Kea from 1962-1966 (Tomich 1986). Another Hawai‘i Island population of mouflon was founded at Kahuku Ranch on Mauna Loa from only 11 individuals between 1968-1974 (Hess et al. 2006). Three separate mouflon populations have become established in the Hawaiian Islands to date. The introduction of axis deer to Hawai‘i Island was debated for many years, but was opposed by ranchers and environmentalists (Titcomb 1969, Walker 1969). Nonetheless, unauthorized translocations of deer and mouflon between islands have occurred more recently. Axis deer were introduced to Hawai‘i Island from Maui (Tummons 2011a) and mouflon from Hawai‘i Island were introduced to Maui in 2009 (Tummons 2011b), prompting an investigation and efforts to eradicate axis deer from Hawai‘i Island.

CONSEQUENCES OF LIBERATION

The introduction of non-native ungulates has been one of the primary causes of endangerment of Pacific Island biota. The native vegetation of Hawai‘i and the Pacific Islands has suffered severe degradation and extinction of many plant species, including roughly 9% of all Hawaiian flora since the time of contact with the Western society (Sakai et al. 2002). More than half of the remaining native plants of Hawai‘i have been designated with some type of state, federal, or international conservation status. Several species of non-native ungulates are known to directly inhibit regeneration and cause mortality in many native tree and understory plant species through herbivory, digging, and bark stripping (Spatz and Mueller-Dombois 1975, Scowcroft and Sakai 1983, Merlin and Juvik 1993, Drake and Pratt 2001, Chynoweth et al. 2013). Although not as well-studied, indirect and interactive processes that have reinforced ecosystem change include increased erosion (Yocom 1967, Stone and Loope 1987), facilitation of invasive species (Weller et al. 2011), enhanced nutrient cycling (Vitousek 1986), water interception by grasses (Hughes et al. 1991, Cordell and Sandquist 2008), and alteration of fire regimes (D’Antonio and Vitousek 1992). The combination of these direct and indirect effects have been associated with the replacement of native forests and shrublands by novel hybrid grazing systems dominated or co-dominated by

non-native species, primarily grasses, and the degradation of habitat for native wildlife.

One of the best-known cases of habitat degradation by ungulates leading to the endangerment of native wildlife is that of a once common bird in the endemic honeycreeper subfamily (Drepanidinae), the palila (*Loxioides bailleui*), on Mauna Kea. Palila inhabited ~1,300 km² of Hawai‘i Island when European explorers arrived, but their range has since been reduced to a core area of ~65 km² due mainly to browsing of endemic m mane tree (*Sophora chrysophylla*), whose seeds are the main food of palila (Banko et al. 2002, Banko et al. 2013). While cattle converted low-elevation m mane forest to pasture, goats and sheep degraded high-elevation habitat (Scowcroft 1983, Scowcroft and Giffin 1983), restricting the elevation range of m mane forest available to palila (Scott et al. 1984). Improved habitat conditions and vegetation recovery was reported following the federal court-mandated removal of “virtually all feral sheep and feral goats” by 1982 from designated Palila Critical Habitat (USFWS 1986, Scowcroft and Conrad 1988, Scott et al. 1984, Juvik et al. 1992, Hess et al. 1999). Mouflon and feral sheep hybrids, however, continued to occupy the area (USFWS 1986) because they had been excluded from the original court ruling, but in 1987 the court ordered the removal of all *Ovis* spp. (Pratt et al. 1997). The effects of long-term browsing have recently been compounded by a severe, prolonged drought, which has caused the further deterioration of habitat and a sharp decline of palila abundance (Banko et al. 2013). Although fence construction and habitat restoration has recently accelerated, palila may respond slowly to improvements due to their restricted diet, low rate of reproduction, and preference for larger, older m mane trees (van Riper et al. 1978, Scott et al. 1984, Pratt et al. 1997, Banko et al. 2009).

CONTEMPORARY CONSERVATION OF NATURAL AREAS

There is now substantial evidence that non-native ungulates have degraded native ecosystems throughout Hawai‘i, and that recovery of native plant communities and associated wildlife cannot occur in the continued presence of ungulates (Price et al. 2009). Experimental exclosures and large-scale ungulate removals have demonstrated beneficial effects for the recovery of native ecosystems in Hawai‘i (Loope and Scowcroft 1985, Loh and Tunison 1999, Cole et al. 2012, Cole and Litton 2013). Land managers have developed highly effective control methods for large-scale removal of feral ungulates (Taylor and Katahira 1988, Anderson and Stone 1993). Ungulates have been completely excluded or removed from roughly 750 km² of important terrestrial ecosystems throughout the Hawaiian Islands to date, primarily on federal lands (Hess and Jacobi 2011). Feral goats have also been entirely eradicated from the islands of Ni‘ihau (189.1 km²) in 1911, L na‘i (361.3 km²) in 1981, 554 km² of Hawai‘i Volcanoes National Park on Hawai‘i Island in 1984, and the island of Kaho‘olawe (116.5 km²) in 1990, the largest Hawaiian Island where all ungulates have been entirely removed. In comparison, the largest land area from which goats have been eradicated on any

Pacific island was the 585 km² Galápagos island of Santiago, Ecuador, in 2005 (Cruz et al. 2009). The removal of ungulates, however, conflicts with sustained-yield hunting programs in that these lands become excluded from public use for the purpose of mammal hunting. Societal values for hunting necessitate the construction and maintenance of expensive barriers to exclude ungulates from pest-free refuges on larger multi-tenure islands (Hess and Jacobi 2011).

SUSTAINED-YIELD GAME MANAGEMENT

Hawai'i is the only state in the U.S. where no native species are hunted and all hunted species are those that have been introduced (Lepczyk et al. 2011). Feral domestic livestock species are the most popular subsistence game but have low trophy value relative to wild continental species. Consequently, revenue generated by licenses, lotteries, and guided hunts is concomitantly low and does not contribute to reinvestment in game management. Game abundance and recruitment is generally not monitored in Hawai'i, and reporting of hunter-killed game is voluntary and incomplete; therefore, biological criteria are not available to establish bag limits. Seasons are often open year-round, but hunters are typically limited to one large mammal per day. Arbitrary seasons and limits, non-mandatory reporting, and the lack of monitoring make it virtually impossible to know the effect of hunting on game abundance or to regulate game populations for any particular level of abundance.

One of the most fundamental pieces of information needed to manage game populations for sustained-yield hunting is the precise total number of animals harvested by hunters. Other vital population parameters include abundance or density in the wild, and annual productivity or recruitment. In the absence of precise data, bag limits cannot be reliably assigned to populations, and managers cannot know if populations are remaining stable, increasing, or decreasing as a result of hunting. There are only a few examples where the abundance of ungulate populations have been determined in Hawai'i, mostly through retrospective analyses of eradication efforts (Anderson and Stone 1994, Hess et al. 2007, Stephens et al. 2008).

In rare cases where data on harvest, abundance, recruitment, and density of wild ungulates are available, useful analyses can be conducted to understand and advance the management of natural resources. An approach which may be useful in addressing the necessary minimum amount of land area allocated for hunting in Hawai'i is to determine the amount of land area necessary for maintaining populations of hunted animals to meet current levels harvested by the public. Hess and Jacobi (2014) demonstrated a general approach to answer the question: What is the total amount of land necessary to provide sustained-yield hunting of game meat for food at the current harvest level on Hawai'i Island if only feral pigs were to be harvested?

Data for this heuristic exercise came from a retrospective analyses of eradication effort at Hakalau Forest National Wildlife Refuge (Hess et al. 2007) and from a study of Hawai'i County Food Self-Sufficiency, which estimated that 187,333.6 kg of meat from all wild game animals was harvested annually by public hunters,

although the report acknowledged uncertainty in reported harvest (Delparte and Melrose 2012). Using estimates of 41.3-43.2% of a population that can be repeatedly harvested annually, a population density of 12.1 pigs per km² (Hess et al. 2007), and dressed weight range of 22.7-27.2 kg per whole pig (Delparte and Melrose 2012), Hess and Jacobi (2014) determined that 1,318 -1,651 km² would be necessary to maintain a pig population large enough to meet the estimated current annual harvest. Although this amount of land area represents only 12.6-15.8% of the total land area Hawai'i Island, it more accurately represents 27.6-43.5% of 4,780 km² (45.7%) of available lands which may be compatible with sustained-yield hunting. Moreover, it does not account for annual variability in productivity or variation in local population density and assumes public access is comparable to current conditions. This simplistic analysis nonetheless provides an example of how game population data can be used in planning for both conservation and hunting.

ADAPTIVE MANAGEMENT

Adaptive management of non-native ungulates in Hawai'i has never been practiced rigorously; however, some elements of adaptive management employed during the course of eradications have allowed the effectiveness of ungulate management strategies to be evaluated retrospectively (Anderson and Stone 1994, Hess et al. 2007, Stephens et al. 2008). Regardless of whether objectives include eradication or sustained-yield management, long-term monitoring of ungulate abundance, recruitment, and harvests are all necessary to assess the long-term effects of management on ungulate population dynamics. Monitoring sex ratios in harvested animals and populations at-large would also improve the ability to manage female bias in bovids, thereby reducing excessive population growth (Stephens et al. 2008). In most cases, however, population monitoring has not been sufficient to assess management strategies of non-native ungulates in Hawai'i.

Adaptive management approaches would also allow important processes such as herbivory, erosion, and the facilitation of invasive plant species to be studied under specific levels of ungulate abundance. The forms of relationships between these processes may indicate levels of ungulate abundance at which environmental degradation may be minimized. Alternatively, there may be no levels of ungulate abundance which result in acceptable conservation of some resources; however, these questions cannot be addressed in the absence of rigorous data on ungulate abundance. The simultaneous acquisition of rigorous data on ungulate population abundance, plant communities, and ecosystems processes would advance the scientific basis for the management of natural resources and game hunting in Hawai'i.

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