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POSTER PRESENTATIONS

Examining the Risk and Rewards for the Anthropogenic Spread of Wild Hogs

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ABSTRACT: Wild hogs (*Sus scrofa*) are an invasive, exotic species that has spread through much of the US through anthropogenic means. Many states have laws and regulations with the intent of reducing the illegal importation, introduction, and establishment of wild hog populations. However, in many cases, these laws have been ineffectual for stopping the anthropogenic spread of wild hogs. To assess the risk for moving wild hogs, we examined various wild hog-related laws throughout the US and assessed the potential reward for their illegal movement of releasing wild hogs for hunting purposes. Initially, we attempted to use the internet to locate various information regarding laws and penalties regarding illegal activity related to wild hogs; however, we found that laws and penalties were difficult to locate on-line (n=5 states where the necessary information could be located on-line), which may ultimately detract from their ability to serve as a deterrent. Most states (n=21) had to be contacted by phone to collect the appropriate data. We found that among states the definition and names of a feral or wild hog varied, making it difficult for prosecutors unfamiliar with wild hogs to easily locate information. We found that 48% of states base their definition of a feral or wild hog on the amount of time that the animal has spent outside of captivity while 30% of states have no specific definition. We could find no information regarding a definition of wild hogs from 22% of states. We found that minimum fines per hog ranged from \$0 to \$10,000 with a median fine of \$500 ($\bar{x}$ = \$1,085, SE = \$571, n=17) and a mode of \$1000. Maximum fines per hog ranged from \$50 to \$10,000 with a median fine of \$1500 ($\bar{x}$ = \$2708, SE = \$576, n=20) and a mode of \$5000. Years in jail per hog ranged from 0 years to 2 years with a median of 1 year ($\bar{x}$ = .7 years, SE = 0.2 years, n=11). We found that the cost of a single-day wild hog hunting trip prices ranged from \$150 to \$1500 ($\bar{x}$ = \$448.9, SE = 263.6, n=146) with a mode of \$500. By applying an Expected Utility Model E(U) = (1-p) U(y) + p U(y – F) where:

E(U) = the actor's expected utility from a contemplated activity
p = likelihood of being punished in the activity
y = the anticipated returns (material or psychological) from the activity
F = the anticipated penalty resulting if the actor is punished for the activity

We found that it was unlikely that most of the current fine and penalty structures would serve as an effective deterrent for illegally reintroducing wild hogs. In many cases the potential rewards, as demonstrated by the economic utility, for releasing wild hogs far outweighed the monetary risk from getting caught. States with few or no wild hogs and weak laws and/or fines are at a substantial risk for the illegal importation of wild hogs. States, such as Tennessee, which incorporate creative fine structures, such as the loss of hunting privileges, are likely to have a more successful deterrent. To reduce the potential for the spread of wild hogs, agencies should concentrate on increasing monetary fines, increasing the perceptions that this illegal activity will be successfully detected and prosecuted, creative fines and penalties, and actively advertising successful prosecution and application of fines.

Standardizing the name of wild hogs throughout North America in the scientific literature and in legislation would also assist prosecutors for building cases based on scientific evidence and for locating supportive information.

Key Words: fines, illegal movement, reward, risk, wild hogs

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Integrated Wild Pig Control™ Results from the EPD Pennahatchee Creek Project

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ABSTRACT: Feral swine (*Sus scrofa*) have the potential to negatively impact ecosystems in a variety of ways, including contamination of water sources. In response to increasing fecal coliform levels due to feral swine in the Pennahatchee Creek watershed in Dooly County, Georgia, the River Valley Regional Commission submitted a 319(h) Clean Water Act grant application to the Georgia Environmental Protection Division to fund efforts to monitor fecal coliform levels and identify their source. As a result of this investigation, JAGER PRO, LLC was hired to remove feral swine within a 2,000 ha target area.
We began surveillance of sounders using high definition infrared-triggered cameras deployed throughout the area at a density of approximately 10-16/100 ha. Images were used to determine direction and timing of travel from bedding areas to food sources, the number of sounders, and the size and demographics of each sounder. Using this information, we identified target areas for winter (December-March) trapping efforts, a time when alternative food sources are limiting. We then deployed digitally timed automatic feeders filled with whole kernel corn at a density of 1 feeder/100 ha. Each feeder was monitored using a camera. Once animals were conditioned to the feeders, we constructed 11-m diameter corral enclosures with 2.4-m wide gates at each site. Traps were triggered using either onsite user-operated remote control, or user-operated cellular remote control, once the entire sounder was routinely entering the trap. When multiple sounders were using a single enclosure at different times, we captured each sounder in reverse order, with the last sounder to visit each night being captured first. Captured animals were quickly dispatched using a suppressed .22 caliber firearm to minimize the potential for disturbance likely to create avoidance of the trap by remaining sounders. Occasionally, individual animals became trap shy and refused to enter standard, baited corral traps. In these instances we identified natural (e.g., streams) or anthropogenic (e.g., culverts) features that concentrated swine movements along field or food plot edges during the planting/growing seasons and installed a remote operated gate at these points. We then used cameras to determine when the entire sounder was willing to pass through the gate, and erected a large 12-panel enclosure attached to the gate. Observers then monitored the trap and trigger the gate with a handheld transmitter after the sounder crossed the trap threshold into the field. We used a similar technique, with only the remote operated gate and approximately 40 m of fencing or panels on either side, to assist in shooting an entire sounder in a single event by closing the gate and blocking retreat following the sounder's entrance into the field. During spring, summer, and fall, we primarily employed night shooting to remove swine, as this time coincides with greater availability of alternative food sources (e.g., row crops, food plots, and hard mast), making trapping more difficult. Night shooting operations primarily involved two techniques: spot and stalk and shooting over bait. During these operations, we used .308 caliber semi-automatic rifles equipped with infrared optics, which allowed identification and eradication of swine in complete darkness. The spot and stalk technique involved shooters stalking single file, into the wind, to within 60 m of foraging animals. A countdown was used to synchronize the first shot from each shooter. Baiting was typically used to remove individual adult boars or sows who previously avoided traps and feeders. Our baiting technique consisted of digging a 23-cm wide by 45-cm deep hole at a well-used bait site. We filled the hole with soured corn covered in dirt to prevent use by non-target animals and allow shooters ample time for observation and shooting of target animals. Bait sites were monitored with a cellular camera, allowing a shooter stationed in a central location to quietly

approach a site immediately upon receiving an image of a target animal using the site. We observed that targeted removal of adults from a sounder via one or more shooting techniques tended to increase trap susceptibility of remaining animals. During December 2012 to June 2014, 76 combined trapping and shooting events resulted in the removal of 624 swine (353 shot, 271 trapped). We used independent two-group t-tests to test for significant differences in catch-per-unit effort and the proportion of the sounder removed between trapping and shooting. Overall, shooting techniques required greater effort per animal removed than trapping techniques (t = 3.57, P = 0.001). However, the mean proportion of each sounder removed per shooting or trapping event did not differ (t = -1.31, P = 0.20). Despite the additional effort required to remove feral swine via shooting, we believe this technique is a necessary component of a complete feral swine control model due to observed differences in behavior and trap susceptibility among individuals. Furthermore, we believe our use of innovative control methods and technologies (e.g., remote cameras and trap-release mechanisms) increased the cost effectiveness and overall efficacy of feral swine removal.

Key Words: Integrated Wild Pig Control, feral swine, trapping system, hog trap, thermal shooting

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Federal Collaboration in Science for Invasive Mammal Management in U.S. National Parks and Wildlife Refuges of the Pacific islands

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ABSTRACT: Some of the most isolated islands in the Pacific Ocean are home to US National Parks and Wildlife Refuges. These islands are known for flora and fauna that occur nowhere else, but also for invasive species and other factors which have resulted in the disproportionate extinction of native species. The control of invasive mammals is the single most expensive natural resource management activity essential for restoring ecological integrity to parks in the Hawaiian Islands, American Samoa, and the islands of Guam and Saipan. Science-based applications supporting management efforts have been shaped by longstanding collaborative federal research programs over the past four decades. Consequently, feral goats (*Capra hircus*) have been removed from >690 km^2 in National Parks, and feral pigs (*Sus scrofa*) have been removed from >367 km^2 of federal lands of Hawai‘i, bringing about the gradual recovery of forest ecosystems. The exclusion of other non-native ungulates and invasive mammals is now being undertaken with more sophisticated control techniques and fences. New fence designs are now capable of excluding feral cats (*Felis catus*) from large areas to protect endangered native waterfowl and nesting seabirds. Rodenticides which have been tested and registered for hand and aerial broadcast in Hawai‘i have been used to eradicate rats from small offshore islands to protect nesting seabirds and are now being applied to montane environments of larger islands to protect forest birds. Forward-looking infrared radar (FLIR) is also being applied to locate wild ungulates which were more recently introduced to some islands. All invasive mammals have been eradicated from some remote small islands, and it may soon be possible to manage areas on larger islands to be free of invasive mammals at least during seasonally important periods for native species.

Key Words: ecosystem recovery, invasive mammals, island ecosystems, predators, research, rodents, ungulates.

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INTRODUCTION

The remote oceanic islands of Hawaiʻi, often described as the most isolated on Earth, exemplify the transformative effects that introduced mammals can bring to insular terrestrial ecosystems. The founding biota of the Hawaiian Archipelago had to possess extraordinary dispersal capabilities to cross half of the Pacific Ocean, and many groups of organisms with lesser capabilities have never become naturally established (Ziegler 2002). Consequently, the Hawaiian Islands, like many other isolated oceanic islands, developed in the complete absence of all ground-dwelling mammals and their associated ecological processes (Carlquist 1970). The discovery of the Hawaiian Archipelago by ocean-voyaging Polynesians and the introductions of several mammals forever altered the ecosystems of these islands. The Polynesian or Pacific rat (*Rattus exulans*) and domestic swine (*Sus scrofa*) were among the first terrestrial mammals to be introduced to the Hawaiian Islands more than 1,000 years ago (Kirch 1982).

Archaeological evidence documented domestic pigs known as *puaʻa* which originated from Island Southeast Asia (Larson et al. 2005; 2007) at permanent Polynesian settlements on the islands of Oʻahu (Pearson et al. 1971), Molokaʻi (Kirch and Kelly 1975; Kirch 1982), and Kauaʻi (Burney et al. 2001). Both skeletal remains and early historic observers indicated that *puaʻa* were smaller than contemporary Hawaiian feral pigs, weighing only 27–45 kg (Ziegler 2002). Despite the fact that domestic swine have become one of the most widely distributed large feral mammals on most islands throughout the Pacific, there is no evidence that pigs strayed far from commensal situations in Hawaiʻi until the admixture of aggressive European strains (Maly 1998, Ziegler 2002, Larson et al. 2005). The Polynesian rat, also originating in Southeast Asia, accompanied early Polynesian voyagers to virtually every island in the Pacific (Kirch 1982, Matisoo-Smith and Robins 2004). The devastating effects of the third most widely distributed rat on Earth have only recently come to light and may have included the catastrophic disappearance of native lowland forests of Hawaiʻi in as little as 50 years (Athens 2009).

It was not until European explorers discovered the Hawaiian Archipelago and initiated another wave of mammalian introductions that larger European swine interbred with *puaʻa*, the first being a boar and a sow brought to the island of Niʻihau by Captain James Cook in 1778 (Tomich 1986). Swine repeatedly interbred with multiple introduced domestic varieties and escaped European wild boars to become the most abundant large mammal throughout the Hawaiian Islands. Pigs, however, were only one of several introduced mammals that became widespread after Europeans colonized the islands. The discovery of the Hawaiian Islands, like many other islands of the Pacific, marked the beginning of introductions of many beasts of burden, animals for milk and meat on the hoof, an assortment of rodents, and small predators to keep rodents at bay. Notably among these were domestic cattle (*Bos taurus*), goats (*Capra hircus*), and sheep (*Ovis aries*) brought by Cook in 1778–1779 and Vancouver in 1793 and 1794 to establish strategic re-supply outposts for ships on worldwide voyages (Tomich 1986). Livestock became feral and proliferated without any predators or competitors. Sheep were reported at the summit of Mauna Kea, the highest peak in the Pacific, only 32 years after their introduction (Ellis 1917). House mice (*Mus musculus*) were brought unintentionally to the Hawaiian Islands by 1816 and reached the summit of Mauna Kea by 1825 (Tomich 1986). Norway rats (*Rattus norvegicus*) were noted by 1835. Tame cats that had been employed as mousers on sailing ships must have fascinated native islanders, as many were given as gifts, bartered, taken, or otherwise escaped into the wild (Baldwin 1980, King 1984), soon spreading as far as the wilderness of Kīlauea by 1840 (Brackenridge 1841), and becoming notorious predators of native birds (Rothschild 1893, Perkins 1903).

Later arrivals included black rats (*Rattus rattus*), which were not documented until 1899, apparently after the construction of shipping wharfs (Atkinson 1977). The small Indian mongoose (*Herpestes auropunctatus*) was deliberately introduced to the Hawaiian Islands from Jamaica in 1883 and released by sugar planters to reduce rat populations in cane fields on Hawaiʻi Island, Oʻahu, Molokaʻi, and Maui, and later brought to other Pacific islands of Fiji and Japan (Hays and Conant 2007). After tens of millions of years of evolutionary isolation from all terrestrial mammals except bats, islands of the Central Pacific were quite suddenly besieged by a number of alien rodents, carnivores, and both large and small herbivores (Ziegler 2002). Rapid ecological degradation ensued and whole groups of endemic plants and animals suffered extinctions, including virtually all flightless waterfowl (Olson and James 1982, Steadman 1995), and at least nine percent of all Hawaiian flora (Sakai et al. 2002). After a century of settlement by westerners, the concept of eradication came about as a solution to primarily agricultural, public health, or economic problems (Tomich 1986), and only more recently as a solution to ecological problems (Hess et al. 2009). The devastation caused by non-native mammals was slow to be realized and addressed; however, there are now many examples of successful management efforts resulting in the dramatic recovery of native biota.

THE LEOPOLD REPORT AND FERAL GOATS

One of the most influential assessments on the management of mammals on federal lands in Hawaiʻi was the report "Wildlife Management in National Parks" by A. Starker Leopold et al. (1963), who gave national recognition to a notable overabundance of herbivores throughout the entire US national park system. Not only did this spur the removal non-native goats from national parks in Hawaiʻi, but it contributed to the restoration of ecological integrity to parks like Yellowstone where the entire suite of large predators was ultimately restored. Managers of Hawaiʻi's National Parks took action on the recommendation of the Leopold Report, which stated: "*A visitor who climbs a mountain in Hawaii ought to see mamane trees and silverswords, not goats.*" Goats had been removed from Hawaiʻi Volcanoes National Park (HAVO) on Hawaiʻi Island since 1927 but with no lasting effect due to reinvasion from the reservoir of animals in surrounding areas (Baker and Reeser 1972). The re-invasion problem was solved by dividing areas into fenced units of manageable size, a difficult logistical process at the time for large areas and dense tropical forests on volcanic substrates. Managers developed specific techniques necessary to accomplish eradication from the enclosed areas such as the Judas goat method which uses radio-telemetry to take advantage of gregarious behavior in domestic ungulates (Taylor and Katahira 1988). The eradication of goats from 554 km^2 of the park during 1968 to 1984 (Tomich 1986) remained the largest area from which goats have been eradicated on any Pacific island until goats were eradicated from the 585 km^2 Galápagos Island of Santiago, Ecuador, in 2005 (Cruz et al. 2009, Chynoweth et al. 2013). After a century and a half of degradation, a previously unknown endemic plant species, ʻāwikiwiki or *Canavalia kauensis* (now *C. hawaiiensis*), was found growing on the dry lowlands of Kukalauʻula after the removal of goats (St. John 1972).

At Haleakalā National Park (HALE) on Maui, eradication of goats from the 137 km^2 ha park began in 1983 and was completed in 1989 using techniques developed in HAVO (Stone and Holt 1990, L. Loope pers. comm.). Goats and sheep were also eradicated from Kahoʻolawe Island in 1990 by ground shooting, helicopter hunting, and the use of Judas animals (Kahoʻolawe Island Conveyance Commission 1993). Goats and sheep had contributed to the loss of as much as 5 m of soil and interfered with livestock operations before the island became a bombing and shelling range after World War II (Kramer 1971). Goat control in National Parks of Hawaiʻi proved not only the technical feasibility to eradicate ungulates from large areas of multi-tenure islands, but also resulted in the development of specific techniques which became standard operating procedures in other locations. The Judas goat method, which uses radio-telemetry to take advantage of gregarious behavior in ungulates, has been replicated in many other management operations (Taylor and Katahira 1988).

SHEEP

Feral sheep have repeatedly reached excessive densities on Mauna Kea, devastating the watershed and dry subalpine woodland environment. Foresters for the Territory of Hawaiʻi conducted sheep drives starting in 1934 that eliminated tens of thousands. The Mauna Kea Forest Reserve (MKFR) was fenced in 1935-1937 (Bryan 1937a) and nearly 47,000 sheep and over 2200 other ungulates were removed in the following 10 years by foresters and Civilian Conservation Corps workers using drives on foot and horseback (Bryan 1937b, 1947). Populations rebounded when sport hunting became a management goal of wildlife biologists after World War II and by 1960, the dire condition of the Mauna Kea forest was decried but not widely known outside of Hawaiʻi (Warner 1960). Despite this knowledge, European mouflon (*Ovis gmelini musimon*) were hybridized with feral sheep and released between 1962 and 1966 to improve hunting opportunities (Giffin 1982). Scowcroft (1983), Scowcroft and Giffin (1983), and Scowcroft and Sakai (1983) used exclosures, aerial photography and studied tree size classes to demonstrate the effects of browsing and bark-stripping by sheep, cattle, and goats on the subalpine vegetation. U.S. Federal District court orders of 1979 and 1986 mandated the removal of goats and sheep to protect the endangered palila (*Loxioides bailleui*) that feed and raise their nestlings on māmane (*Sophora chrysophylla*) seed pods. More than 87,000 sheep have been removed from the MKFR over a 75-year period, but sheep are still far from being eradicated. Patchy recovery of māmane has occurred after reduction of sheep numbers (Hess et al. 1999). The fence surrounding Mauna Kea has not been maintained and several hundred sheep are removed each year by aerial hunting from helicopters; however, habitat loss compounded by drought has contributed to an ongoing long-term decline of Palila (Banko et al. 2009; 2013).

European mouflon sheep from the Mediterranean Islands have become invasive where they were introduced to the Canary, Kerguelen, and Hawaiian archipelagos (Chapuis et al. 1994, Hess et al. 2006, Nogales et al. 2006). Mouflon were first introduced to the Hawaiian island of Lānaʻi in 1954 as a game species prior to their release on Mauna Kea (Tomich 1986). A third population on Hawaiʻi Island's Mauna Loa was founded by only 11 individuals between 1968 and 1974 at the Kahuku Ranch which was acquire by Hawaiʻi Volcanoes National Park in 2003 (Hess et al. 2006). As the Mauna Kea and Mauna Loa populations grew and started to merge (Ikagawa 2014), a directed volunteer program began to eliminate mouflon to prevent further degradation at Kahuku (Stephens et al 2008).

Control of non-native ungulates is the single most expensive natural resource management activity in many natural areas of Hawaiʻi. It is often difficult to detect small numbers of incipient and relictual ungulates in these areas, especially for cryptic species which have never been domesticated. Aerial surveys are the most common method for assessing ungulate populations on a large spatial scale. However, the effectiveness of aerial surveys diminishes after populations have been reduced to relictual levels. Ground-based surveys, camera trap monitoring, and aerial surveys enhanced with Forward Looking Infrared Radar (FLIR) are now being compared to detect mouflon and other ungulates in a 131 km^2 area at Kahuku. From 2004 to 2014, the number of mouflon observed during aerial surveys decreased from 1,785 to 378, and no mouflon were detected in two intensively managed subunits, despite reports of small numbers (USGS, unpubl. data). During systematic ground-based surveys, fresh sign occurred at 3.6% of plots within one of the managed units. Twenty remote triggered camera traps were positioned in Kahuku; four in the unit where sheep had been detected during ground surveys. Over a 199 day period, 863 images of sheep were collected, including seven detections in a managed unit. Each method has strengths, but is limited by effective detection distance, spatial and temporal coverage, as well as intensity of effort. Systematic survey methods coordinated with continuous camera trap monitoring complemented each other when used for detecting small numbers of ungulates.

FERAL PIGS

Feral pigs differ fundamentally from that of other ungulate species because, in addition to herbivory and trampling, pigs also wallow, dig, and root in soil (Engeman et al. 2006), primarily in wetter forests. The actions of pigs are considered to disperse some alien plants (Diong 1982, Aplet et al. 1991, LaRosa 1992), inhibit regeneration of native plants (Cooray and Mueller-Dombois 1981, Diong 1982), selectively browse and destroy native plants (Ralph and Maxwell 1984, Stone 1985, Stone and Loope 1987, Drake and Pratt 2001, Murphy et al. 2013), spread plant pathogens (Kliejunas and Ko 1976), accelerate soil erosion (Stone and Loope 1987), alter soil microarthropod communities (Vtorov 1993), and alter nutrient cycling (Coblentz and Baber 1987, Singer 1981, Vitousek 1986). Feral pigs in Hawaiʻi also create nutrient-rich wallows and troughs in tree fern (*Cibotium* spp.) trunks (Stone and Loope 1987). Despite the fact that feral pigs have been implicated in altered ecosystem processes in Hawaiʻi and elsewhere, some important aspects of feral pig ecology in Hawaiʻi are still poorly studied because of the inaccessible environments they inhabit, and because their effects cannot be disentangled from those of other sympatric ungulate species.

Several studies have examined the recovery of plant communities after landscape-scale removal of pigs. Loope et al. (1991) found that the removal of feral pigs from a montane bog on Maui reversed damage to vegetation, and the presence of alien plant species was minimal due to inherently low invisibility of native bog communities. Nonetheless, pigs had only a short history (< 20 yr) in this area. Loh and Tunison (1999) monitored vegetation changes following pig removal at 16 plots in pig-disturbed areas of the ʻOlaʻa-koa rainforest unit in Hawaiʻi Volcanoes National Park. Native understory cover increased 48% from 1991 to 1998, largely in the first two years following pig removal. Alien understory vegetation increased 190%. The presence of alien banana poka (*Passiflora mollisima*), however, was reduced from 81% to 40% within plots. Hess et al. (2010) analyzed vegetation monitoring over a 16-year period concurrent with feral pig and cattle removal in a wet montane forest at Hakalau Forest National Wildlife Refuge (HFNWR) on Hawaiʻi Island. Strong increases in understory cover of native ferns and slight decreases in cover of bryophytes and exposed soil occurred. Mean cover of native plants was generally higher in locations that were formerly lightly grazed, while alien grass and herb cover was generally higher in areas that were heavily grazed. In contrast to many other Hawaiian forests, widespread invasion by alien grasses and herbs did not occur after ungulate removal and may be due to dense canopy cover.

Cole et al. (2012) and Cole and Litton (2013) found that stem density and cover of native plants, species richness of groundrooted native woody plants, and abundance of native plants of conservation interest were all significantly higher where feral pigs had been removed from a Hawaiian montane wet forest over 6.5–18.5 years. The area of exposed soil was lower and cover of litter and bryophytes was greater where pigs were absent. Density of groundrooted native woody plants increased sixfold in pig-free sites over 16 years, whereas establishment was almost exclusively restricted to epiphytes at sites inhabited by pigs. Stem density of young tree ferns also increased significantly in pig-free sites, but not at sites inhabited by pigs. Abundance of invasive plants such as strawberry guava (*Psidium cattleianum*) increased fivefold at sites where they had established prior to feral pig removal. While common native understory plants recovered within 6.5 years of feral pig removal, species of conservation interest recovered only on areas that possessed remnant populations at the time of removal. Results indicated that control of nonnative plants and outplanting of rarer species may be necessary after pig removal.

Because pigs are extraordinarily prolific (Hess et al. 2007), reinvasion of from the reservoir of animals in surrounding areas is a perpetual problem, making continuous fence maintenance and population monitoring in managed areas necessary. HFNWR has intensively managed feral cattle and pigs and monitored non-native ungulate presence and distribution during surveys of all managed areas since 1988. Activity indices for feral pigs, consisting of the presence of relatively recent tracks, digging, browse, or scat was recorded at

422 stations along 17 transects, each with roughly 20 sample plots (Leopold et al. 2015). A calibrated model based on the number of pigs removed from one management unit and concurrent activity surveys was applied to estimate pig abundance in other management units (Hess et al. 2007). The resulting time series of pig abundance provides managers with a means to evaluate and refine control efforts in an adaptive management framework. The simultaneous acquisition of rigorous data on ungulate population abundance, plant communities, and ecosystems processes would further advance the scientific basis for the management of natural resources in Hawaiʻi.

RECENT ILLEGAL INTRODUCTIONS: AXIS DEER

Among the wild ungulates introduced to Hawaiʻi 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 Molokaʻi were moved to Lānaʻi in 1920. Axis deer were later released on Maui in 1959 where they have become widespread (Anderson 2003). The introduction of axis deer to Hawaiʻi Island was debated for many years, but opposed by ranchers and environmentalists (Titcomb 1969, Walker 1969). Nonetheless, illegal introductions of deer and mouflon between islands have occurred recently. The U.S. Fish and Wildlife Service launched an investigation after sightings were reported, which revealed that in December 2009, a helicopter pilot and rancher from Maui had covertly transported four deer in exchange for about a dozen European mouflon sheep (Tummons 2011a, b). Because neither species was established in the wild on either of the islands, in June 2012, state lawmakers responded by specifically banning "the intentional possession or interisland transportation or release of wild or feral deer" (Honolulu Star-Advertiser 2012). Two individuals were prosecuted under the Lacey Act for transporting wildlife between islands with the intent to guide hunting for out-of-state residents (Associated Press 2012), while the individual who provided the mouflon was sentenced to community service. Further, the helicopter pilot agreed to provide 500 hours of flight time to locate and eradicate the Hawaiʻi Island deer population in restitution (Hess et al., in press). FLIR has been used to locate and dispatch four individuals to date.

RODENTS

Introduced rodents, particularly black rats, have become superabundant on most of the world's inhabited islands, causing widespread ecological damage and tremendous human health problems. Rodents prey on birds at all life history stages and compete by preying on invertebrates and seeds, often interrupting reproduction in plants (Lindsey et al. 2009). Rodents also carry several diseases that are communicable to humans, domestic mammals, and native wildlife. The bacteriological diseases murine typhus and bubonic plague caused by the organisms *Rickettsia typhi* and *Yersinia pestis* are hosted by many rodent species (Tomich et al. 1984). These diseases have a long history of causing human illness and mortality in Hawaiʻi. Although plague has not occurred in the archipelago since 1957 (Tomich et al. 1984), murine typhus outbreaks still occur periodically, with 47 confirmed human cases in a 2002 outbreak (Manea et al. 2001, Sasaki et al. 2003). Leptospirosis, caused by the spirochete *Leptospira interrogans*, is one of the most widespread, sometimes fatal zoonoses worldwide, having an annual incidence of 1.29 per 100,000 people in Hawaiʻi (Middleton et al. 2001, Katz et al. 2002). Other diseases associated with rodents, such as cryptosporidiosis, giardiasis, and salmonellosis, pose persistent and serious public health problems (Sasaki and Ikeda 2000, Katz et al. 2002).

Recognizing the severe problems rats cause to nesting seabirds, the U.S. Fish & Wildlife Service (USFWS) and the Samoan Department of Wildlife and Marine Resources eradicated Polynesian rats from 6.3 ha Rose Atoll, American Samoa, in 1990 using brodifacoum, a second generation anticoagulant, in bait stations, live- and snap-traps, and subsequent treatment with bromethalin (Morrell et al. 1991, Murphy and Ohashi 1991, Ohashi and Oldenburg 1992). In the Northwestern Hawaiian Islands, Wildlife

Services (WS) of the U.S. Department of Agriculture's Animal and Plant Health Inspection Service and the Hawaiʻi Department of Land and Natural Resources (DLNR) eradicated Poynesian rats in 1993 from 129 ha Green Island, Kure Atoll, using brodifacoum bait stations (J. Murphy pers. comm.). In 1994–1996 the U.S. Navy, USFWS and WS eradicated black rats from three islands of Midway Atoll using brodifacoum, live traps, incidental baiting and rat nest removal (J. Gilardi and J. Murphy, pers. comm.; Murphy 1997a,b). Sand Island of Midway Atoll remains one the largest permanently inhabited islands in the U.S. from which rats have been removed. Growth of the Bonin petrel population from an estimated 32,000 nesting birds (Seto and Conant 1996) to more than 900,000 provides compelling evidence for the enormous benefits of rat eradication. Native vegetation on Midway also became noticeably more dense and abundant (N. Hoffman pers. comm.). Mice on Sand Island are now the only small mammal remaining in the Northwestern Hawaiian Islands.

At Palmyra Atoll in the equatorial Line Islands, rats prevented six seabird species from nesting. The first attempt to eradicate ship rats from the atoll by WS failed in 2001 due to the complexity of the 275 ha area with 54 islets, and dense coconut palms (*Cocos nucifera*), and *Pisonia grandis* trees (Ohashi 2001). Notable factors contributing to the failure included bait taken by land crabs (*Cardisonma* and *Coenobita* spp.). A more intensive second attempt was successful by 2013, benefitting coconut palms and *Pisonia* trees.

The successes of rat eradication on remote islands have also brought about efforts to restore offshore islets of the main Hawaiian Islands. In 2002, the Offshore Islet Restoration Committee was formed to restore selected islets around the Hawaiian Islands. To date, rat eradications have been successful on Mokuʻauia and tiny Mokoliʻi Islet, both near Oʻahu, using traps and diphacinone, a first generation anticoagulant, in bait stations (J. Eijzenga pers. comm.). Wedge-tailed shearwaters (*Puffinus pacificus*) subsequently began fledging from Mokoliʻi (D. Smith pers. comm.). A joint project by the USFWS, Hawaiʻi DLNR and WS to eradicate Pacific rats from 7 ha Mokapu Island off Molokaʻi in February 2008 was the first rat eradication using an aerial application of a rodenticide (diphacinone) which was registered by the Environmental Protection Agency (EPA) in 2007 for conservation purposes in the U.S. (P. Dunlevy pers. comm.). Diphacinone pellets were also broadcast by helicopter for Polynesian rats in January 2009 on 110 ha Lehua Islet, but the eradication proved unsuccessful (VanderWerf et al. 2007; P. Dunlevy pers. comm.).

Larger areas of multi-tenure islands are now under consideration for the use of registered broadcast rodenticides for rodent control. Rodenticide treatment grids are being established in Hawaiʻi Volcanoes National Park where hand and aerial broadcast trials of diphacinone pellets were conducted in support of EPA registration (Spurr et al. 2013). Several native and non-native species will be monitored to examine ecosystem responses. Reinvasion of from the reservoir of animals in surrounding areas is inevitable; however, this type of seasonal management regime may benefit nesting forest birds and other species during important life history stages, thereby providing an important conservation tool.

FERAL CATS

Domestic cats have been introduced to many of the world's islands where they have frequently become the dominant apex predator in the absence of other predatory mammals. The consequences have been particularly devastating for native wildlife, including the decline, extirpation, and extinction of numerous vertebrate populations, particularly ground-nesting and burrowing landbirds and seabirds, as well as many herptile and small mammal species which, in most cases, evolved in the absence of predatory mammals and feline diseases. The depredation of endangered bird species in Hawaiʻi has been frequently documented and attributed to cats based on the characteristic condition of carcasses (Hess et al. 2007, Lindsey et al. 2009, Judge et al. 2012). Remains have also been recovered from stomach contents of feral cat and from cat scats, but dietary studies cannot differentiate between prey killed by feral cats and scavenged food items. Other types of evidence including mortality attributed to pathogens are also often short of conclusive. Photographic or videographic documentation provides direct ‘smoking gun’ evidence that confirms depredation by cats (Judge et al. 2012). The most direct and compelling proof of the effects of feral cats on wildlife populations come from examples where cats have been entirely removed from islands and comparisons of areas with and without cats (Smith et al. 2002). In many cases, several species of extirpated seabirds as well as other wildlife have recovered after the complete removal of cats (Hess, in press and references therein).

In the Central Pacific, five species of seabirds have recolonized the islands of Baker, Howland, Jarvis, and Wake after the removal of feral cats (Rauzon et al. 2011). Worldwide, feral cats have been removed from more than 50 islands, many of which are remote and inaccessible. In cases where follow-up monitoring has been conducted and published, recovery of 22 species of birds on 11 islands has been documented on islands including Ascension, Juan de Nova (Mozambique), Marion, and several Islands of Mexico (Hess, in press and references therein). Where possible, the experimental removal of cats would provide the most conclusive proof of effects on wildlife populations.

MULTI-SPECIES PREDATOR EXCLOSURES

On multi-tenure islands where the eradication of feral cats and other predators may not be possible, predator exclosures provide the best prospects for the recovery of seabirds and other endangered bird species. Four such projects have been planned or undertaken in Hawai‘i. Predator-proof fences have been developed and refined in New Zealand to exclude a wide variety of mammalian predators from vulnerable native bird species. They typically consist of a tall fence mesh fine enough to exclude mice, buried skirt to prevent burrowing, and a curved or floppy top to prevent predators from climbing over (Hess et al. 2009). One of the first predator-proof exclosures in Hawai‘i was a relatively small (~0.7 ha) area in Hawai‘i Volcanoes National Park to protect endangered Nēnē (Hawaiian Goose; *Branta sandvicensis*) goslings from feral cats, feral pigs, and mongooses (Hess 2011). Ka‘ena Point on O‘ahu became the first site in the Hawaiian Islands to get a predator-proof fence to exclude all mammals from mice to dogs (Young et al. 2013). The fence spans 640 meters and encloses an area of approximately 24 ha. Removal of dogs, feral cats, and mongooses has been particularly beneficial to nesting seabirds like Wedge-tailed Shearwaters, but also to Laysan Albatross (*Phoebastria immutabilis*). Dogs frequently do a substantial amount of damage to shearwater populations by killing nesting adults. Another 3.2 ha predator-proof fence was completed at Kīlauea Point National Wildlife Refuge in December of 2015 to protect Nēnē, Laysan Albatross, and Newell’s Shearwater (*Puffinus newelli*). The American Bird Conservancy is currently supporting the construction of a much larger exclosure to protect the largest colony of endangered Hawaiian Petrels on Mauna Loa in Hawai‘i Volcanoes National Park.

PROGNOSIS

Federal agencies have been highly successful by collaborating in scientific research and management of invasive mammals on federal lands, culminating in the removal of several destructive species across large landscapes and many entire islands, and resulting in demonstrated ecosystem recovery (Hess and Jacobi 2011). Additional multi-species eradications of invasive mammals from larger single-tenure islands would benefit numerous species of wildlife. Kaho‘olawe (117 km^2) would not be the largest island in the world from which feral cats have been eradicated, but it would be nonetheless logistically challenging because of unexploded ordnance left after decades of military training, and it would also require coordinated eradication of Polynesian rats and mice. Aerial broadcast of brodifacoum could be highly effective for eradicating rodents and simultaneously reducing feral cats on Kaho‘olawe, but it presents higher risks to non-target animals than diphacinone, which may be less effective, particularly against feral cats and mice (Parkes 2009). While some research may be necessary to develop the best methodological strategy, there is little question that a pest-free Kaho‘olawe would be important for restoration of native seabirds and potentially other native species of plants and animals, including some that do not occur outside the northwest Hawaiian Islands, such as Laysan Teal (*Anas laysanensis*). The future conservation value of Kahoʻolawe may become increasingly important as feral cat colonies continue to become established on other large islands, threatening the viability of native wildlife (Winter 2003).

Multi-tenure islands where rights prevail are substantially more challenging for invasive mammal management, however, and the pace of new introductions is increasing. Better prevention strategies, early detection techniques, and control methodology for incipient invasive species would benefit the environment, agriculture, and economy of the entire Hawaiian archipelago. For example, small Indian mongooses (*Herpestes auropunctatus*), which have infested nearly all of the other Hawaiian Islands, were first discovered and captured on Kauaʻi in 2012, threatening endangered ground-nesting bird populations. Abundant source populations of these and other invasive vertebrates throughout the archipelago present a growing risk for accidental and intentional introductions to cross-contaminate islands. As with deer on Hawaiʻi Island, detection and control is dependent on the trust and cooperation of landowners, who can deny access at any time. Successful eradication cannot be declared yet in many cases because it is virtually impossible to know if the last individual of a population has been removed from such large, populated islands. Therefore, the best chance for stopping additional invasions includes prevention, early detection, and rapid response before newcomers have a chance to reproduce. Vigorous enforcement of existing importation laws would aid in the prevention of additional introductions, while outreach would inform the public of both ecological and legal consequences. Solid engagement from natural resource agencies would improve early detection and rapid response. Once a small population of invaders starts to reproduce and becomes established, long-term commitment to monitoring and removal in partnership with landowners is the best shot for ensuring successful eradication—particularly for cryptic species.

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Are Humane Traps "Humane"? An Animal Welfare Perspective

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ABSTRACT: Wild animal trapping is one of humankind's most ancient occupations having existed as non-controversial for countless millennia as part of subsistence economies worldwide. With the rise of animal welfare and protection interests in the mid-eighteenth century, however, the quiet surrounding the various practices that make up trapping seems to have ended. Not only did critics start to question the pain trapped animals experienced, but they began also to raise concerns for trapping in a broader moral context, as in Darwin's example of the additional suffering a trapped animal might experience when the gamekeeper decides to sleep in on a cold morning (Darwin 1863). Organized opposition to the use of traps in North America can be dated to the formation of the Anti-Steel-Trap League in 1925, which campaigned for legislative bans while raising public visibility about trapping in ongoing awareness campaigns. With the rise of animal rights in the 1970's pro- and anti- trapping interests reached an apparent impasse through their "unreconcilable philosophies" (Proulx and Barrett 1991). That did not prevent, however, movement to seek improvements in "humaneness" through advances in trap design and testing, efforts to rank and standardize injury (Iossa et al. 2007), progress on international agreements focused on best practices (Harrop 1998, Fox and Papouchis 2004) and calls for addressing animal welfare concerns, even for species labeled as "pests" (Littin et al. 2004). It is important such efforts continue and that the concept of humaneness in trapping be broadened beyond concerns for the immediate physical effects of devices to their use within a far wider practical and moral context. Among other reasons for this need is that what have been termed "antiquated systems" remain widely in use today (Proulx et al. 2015). A renewed effort to better understand why animal welfare is not treated as a first order concern in wildlife trapping is necessary. As a part of this effort, we should look beyond the trapping devices themselves and engage the broader circumstances and activities associated with their use. Trapping is a process that involves choices, decisions, actions, and results whose consequences should be amenable to evaluation, all with the objective of improving welfare. Difficulties arise in that any event involving trapping will always be set within a stochastic context where varying conditions or circumstances potentially compromise the "humaneness" of the activity. For example, even a so-called "humane" box or cage trap if left unattended in direct sun on a hot summer day can result in an agonal death for a trapped animal. Poor site selection or lax attendance can subject trapped animals to predation, and trap sets that intentionally submerge and drown animals are not humane (Ludders et al. 1999).
Warburton and Norton (2007) describe trapping as associated with moral, ethical, cultural, economic and wildlife management perspectives, identifying it as multi-dimensional in both technical as well as social respects. Progress on the technical side can be represented by the development of traps that limit the severity of injuries and rejection of traps that exceed thresholds (Iossa et al. 2007). However, because of the many variables inherent to trapping the criteria for the "humaneness" of any device must remain performance-based, so that the state-of-art device might render 70% of trapped animals or more irreversibly unconsciousness within three minutes at a ninety-five percent confidence interval (Proulx and Barrett 1994). Elsewhere, some trap designs allow for selectivity in mostly capturing specific species, leading to claims they are more "humane" because of that (Hubert et al. 1996). In both cases, claims of humaneness are simply relative to what occurs with respect to other practices, and do not mean that either the standards or devices in question are themselves humane. Welfare assessments (Sharp and Saunders 2011) can play an increasingly important role in advancing dialogue about traps as well as the practice of trapping. Matrix models can evaluate the consequence of actions as a function of their duration and begin to account for the magnitude of welfare compromise (Kirkwood et al. 1994). While the "unreconcilable

philosophies" surrounding trapping issues may threaten gridlock, the issues involved are far too significant to allow this to happen.

Key Words: animal protection, animal welfare, humane, traps, trapping, welfare assessment

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Design of a Self-Resetting, Low-Maintenance, Long-Term Bait Station for Rodent Control

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ABSTRACT: A low-maintenance, long-term bait station that resets itself after being triggered would be a very useful tool for controlling Richardson's ground squirrels, or other problem rodent species, in remote locations. With collaborators, we developed and tested two such devices using lab rats in pen settings. The devices can be left *in-situ* for long periods of time without servicing, and requires only occasional bait and/or battery replacement. Squirrels would be unable to cache bait due to the integrated time-out mechanism. The devices use capacitive sensor or strain gauge systems for animal identification, making it very unlikely that smaller non-target species would be able to trigger the systems while the design precludes entry by larger non-target species. Further refinement and testing will be needed before a viable, commercial product can go into production. These refinements include increasing reliability, reducing power requirements, design features and triggering mechanisms tightly linked to the attributes of the targeted pest species, and reduction of production costs. The devices will also need to be tested in field settings for extended periods of time.

Key Words: bait station, remote locations, rodent, rodenticide, wildlife damage

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INTRODUCTION

Approximately 42% of all mammalian species in the world are rodents; this amounts to about 2,277 species rodents (Wilson and Reeder 2005). They occur on all continents with the possible exception of Antarctica. However, even there, commensal rodents may have been accidently introduced to the inhabited research stations. Rodent species have adapted to all life-styles: terrestrial, aquatic, arboreal, and fossorial (underground). Most rodent species are small, secretive, nocturnal, adaptable, and have keen senses of touch, taste, and smell. For most species of rodents, the incisors continually grow throughout their lifespan, requiring constant gnawing to keep the incisors sharp and at an appropriate length. Rodents have ecological, scientific, social, and economic values (Dickman 1999, Witmer et al. 1995). Rodents are important in seed and spore dispersal, pollination, seed predation, energy and nutrient cycling, the modification of plant succession and species composition, and as a food source for many predators. Additionally, some species provide food and fur for human uses, and can provide an ecosystem service for smallholder farmers through consuming pests of their crops.

Rodents cause many types of damage to human resources. The types of agricultural damage inflicted by rodents include the direct feeding on seeds and plants at all stages of the cropping cycle (i.e., planting, vegetative growth, maturation, and pre- and post-harvest). Additionally, rodents cause damage from their burrowing activities which can result in levee failures, flooding of fields, loss of water resources, and the undermining of structures and foundations (Joshi et al. 2000, Stuart et al.

2008). Burrows and burrow openings can result in damage to farm equipment and injury to workers or livestock. Through their gnawing activity, rodents can damage equipment, irrigation tubing, and buildings. For example, house mice cause significant damage to insulation in confined livestock operations (Hygnstrom et al. 1996). Chewing through wiring can result in power failure or devastating fires (Caughley et al. 1994). Rodents also compete with livestock for feed whether in confined operations or open rangeland. They also contaminate stored food with their feces and urine.

Many methods exist to reduce rodent populations and/or damage (Hyngstrom et al. 1994, Buckle and Smith 2015, Witmer and Singleton 2010). However, rodenticides (and to a lesser extent traps) are heavily relied upon (Witmer et al. 2007). While in some situations, rodenticide baits are broadcast by hand or machine over large areas, in or near buildings rodenticides are often placed in bait stations. This reduces the risk of poisoning of children, pets, livestock, and non-target animals. However, current bait stations are passive device which must be checked and refilled periodically. Rodents will often cache or hoard the bait by making repeated trips to take bait to their burrows or nests; thus, requiring frequent refilled of the bait station. This poses issues for widely scattered, remote and unmanned facilities such as power substations and many military sites such as intercontinental ballistic missile (ICBM) silos (e.g., Witmer et al. 2012). In some of these situations, self-resetting, long-term, low-maintenance baits stations would be a valuable addition to the rodent control toolbox.

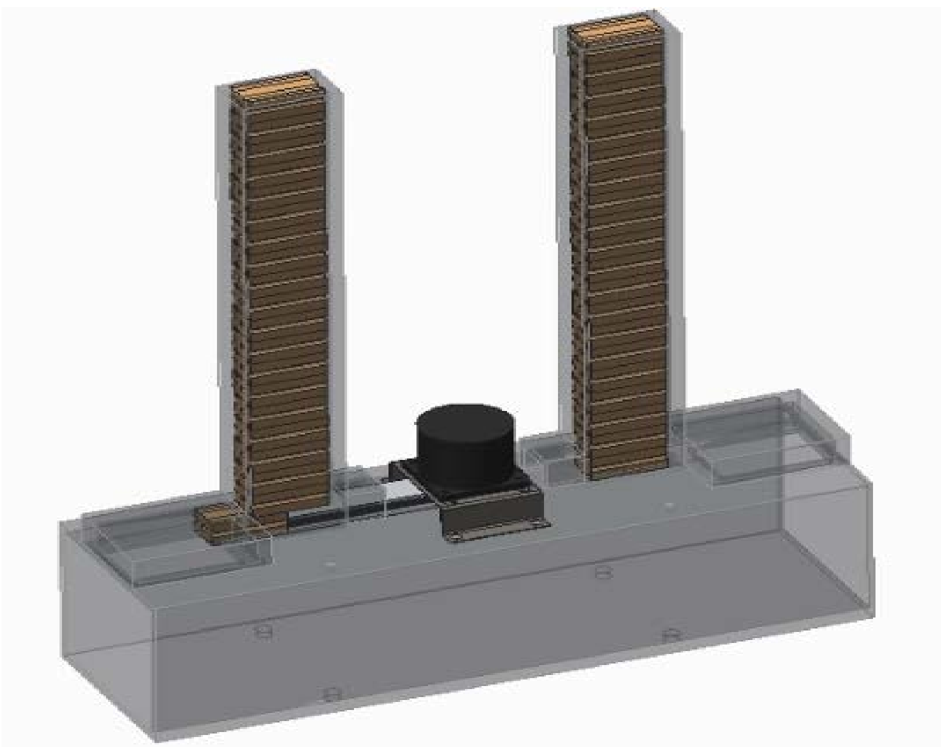
The features and characteristics we sought were:

- High durability
- Low-maintenance
- Capable of storing substantial amounts of bait
- Environmentally robust with bait protected from weathering
- Predetermined lethal dose of bait delivered upon triggering
- Incorporated “time out” (i.e., the bait station would re-set itself after delivering a bait, but will not deliver another dose for a predetermined period of time to prevent bait caching/hoarding)
- Capable of continued operation over long timeframes without staff visits

COLORADO STATE UNIVERSITY PROTOTYPE

Engineering seniors at Colorado State University (CSU), Fort Collins CO, are required to complete a special project in their senior year. We formed a team to design and build a self-resetting bait station to meet that academic requirement. The students designed and built a prototype meeting most of the desired features and we tested it with lab rats, using non-toxic rodent chow blocks. The lower structure was a tunnel-like design that was open at both ends so that rodents could see all the way through the device, thus feeling more at ease in entering the device. The structure was made of hard, clear plastic and had two tall towers to hold rodenticide bait blocks (Figure 1). There was a circuit board to control the 12 volt unipolar stepper motor, timer, strain gauge sensor, and the horizontal rack and pinion track. The linear action of the rack pushed a plunger to drop a bait block from one tower and the next time activated, it would move in the reverse direction to drop a block from the other tower. On the central floor area of the device was the strain gauge sensor which, based on the animal’s weight, would activate the plunger. We had the gauge set to activate if it detected an animal weight of about 400 g (roughly the weight of a ground squirrel) so that mice or small birds would not trigger the device. For the trial with lab rats, the dispense interval was programmed at one hour. Motion sensitive and video cameras were used to record rat use/entries and bait drops of the station. The device performed as designed, dispensing all the bait blocks over the course of 3 days. In a field application, the device would be programmed to only drop a bait every eight hours or so when triggered by an animal. Some redesign was needed to lower the power demand. Additionally, debris tended to accumulate under the strain gauge sensor, affecting its ability to detect the correct animal weight. To remedy that, force sensitive resistors were tried, but they were not suitable substitutes for the strain gauge sensors. The device is powered by a 12 volt battery. Additional efforts were made to reduce the cost of the device. We estimated that if the parts were purchased in bulk, the price of one device would be about $120-130. One of the main upfront costs would be in having the body of the device made through plastic injection molding with a high cost in the production of the mold.

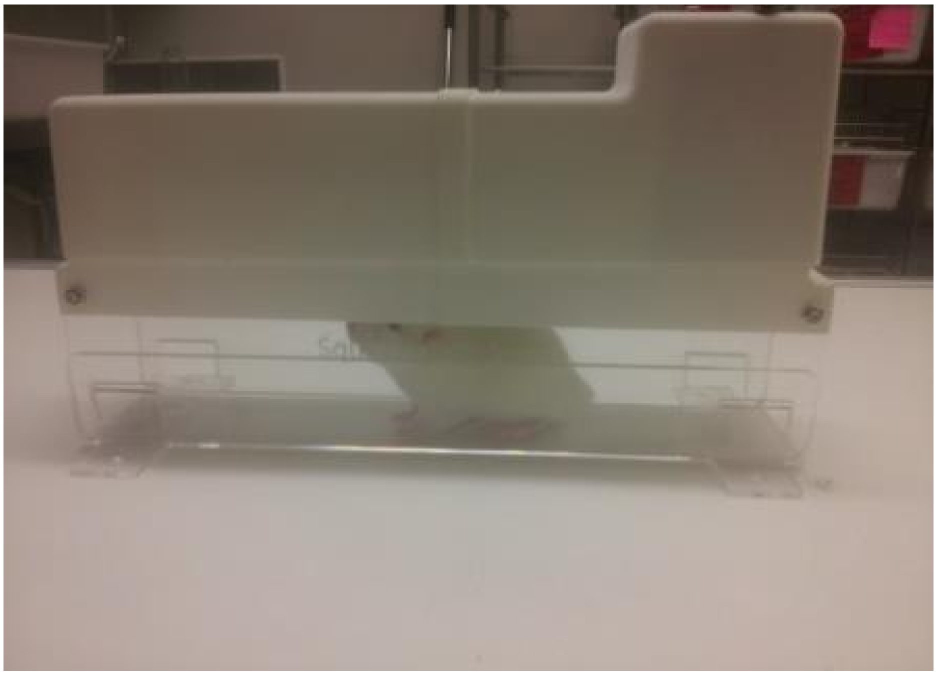
Figure 1. The Colorado State University self-resetting rodenticide bait station.



LINCOLN UNIVERSITY PROTOTYPE

Wildlife and engineering staff at Lincoln University, New Zealand, were subcontracted to design, build and test a prototype self-resetting bait station. They were contacted about the project, in part, because they had been working on similar devices for invasive stoat and weasel control in New Zealand (Blackie et al. 2012). Those devices were designed to detect the invasive animal and spray it with a toxic paste containing para-aminopropiophenone (PAPP). The animal consumes a lethal dose when it grooms the paste off its fur. For our rodent control project, they started out with a vertical device, but then switched to a lower, horizontal device profile that would suit the outdoor terrain better as well as the bait storage area (Figure 2). They used a vacuum-formed rodent-chewing resistant plastic housing which is lightweight, but very robust. Other aspects of the design varied considerably from the CSU prototype. They used a horizontally-oriented bait storage container and bait sachets which could contain, for example, zinc-phosphide coated grain. An acute toxicant would be preferable over an anticoagulant because the animal would be incapacitated or dead before it could take additional baits. While the sachets are housed in a cardboard container, that container resides within the plastic device above the ceiling of the rodent "tunnel". Additionally, instead of using the animal's weight as a triggering mechanism, they used two capacitive sensors an appropriate distance apart for the targeted species. Both sensors have to be triggered at the same time for the device to drop a bait sachet. This approach was found to be simpler and more reliable than a weight-activated platform. Like the CSU prototype, the device has a rodent tunnel that is open at both ends and also uses a "time out" mechanism so that the device will not drop another bait sachet before the programmed time has elapsed. The device has a low power drain, but is equipped with three 9 vole batteries that would last for years in the field. As with the CSU device, the Lincoln University device would be relatively expensive to produce unless they were produced in large numbers with bulk-priced components.

Figure 2. The Lincoln University self-resetting rodenticide bait station.



CONCLUSIONS AND FUTURE NEEDS

The continued development of rodent control technologies is essential to reduce the losses of human resources. This is especially true for remote locations, unmanned sites, and rodent control on distant, uninhabited islands. As stated by Blackie and others (2013): "With the integration of new technological and engineering advances, resetting control systems offer the potential to "set and forget" devices in the field for extended periods, allowing continued population suppression over longer timeframes, and an ultimate decrease in control costs."

We have designed, built, and tested two rodent control prototype devices that appear to meet those goals. The final reports with more details and diagrams than in this summary article are available from the senior author. Further refinement and testing will be needed before a viable, commercial product can go into production. These refinements include increasing reliability, reducing power requirements, design features and triggering mechanisms tightly linked to the attributes of the targeted pest species, and reduction of production costs. The devices will also need to be tested in field settings for extended periods of time.

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Evaluation of a Food Bait Block for Potential Chemical Delivery to Black-tailed Prairie Dogs (*Cynomys ludovicianus*)

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ABSTRACT: Fertility control is a potential method to control prairie dog populations in the urban/suburban environment. However, an effective, oral delivery system is needed. We tested a food bait block delivery system that could make baits available to prairie dogs over a number of days which would make this method more cost-effective than placing food bait by hand near burrows every day. Prairie dogs readily consumed the bait blocks stacked on vertical metal poles during the day. We found, however, that rabbits and mice also consumed the food bait blocks, mainly at night. Over the course of the study, the mean amount removed per site was 81% of the food bait presented. However, to make the food bait blocks primarily available to prairie dogs, a device that would eliminate access to the food bait blocks at night is needed.

Key Words: fertility control, food bait block, prairie dog, cottontail rabbit, wildlife damage

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INTRODUCTION

Prairie dogs (*Cynomys ludovicianus*) are a rodent species of the grass prairies of the USA. They pose many challenges to resource managers in highly disturbed settings, such as suburban areas, where conflicting interests persist regarding the presence of prairie dogs (Witmer et al. 2000). The history, biology, ecology, and status of prairie dogs has been reviewed by Clippinger (1989), Fagerstone and Ramey (1996), Hoogland (1996), Mulhern and Knowles (1996), and U.S. Fish and Wildlife Service (2000). There is a need to better monitor colonies and the changes that they undergo as well as a need to plan for future events. Municipalities have designed management plans to reduce conflicts by using public input, zoned management areas, and a variety of management techniques and tools. Individual populations must often be managed very differently.

The prairie dog management plans of two Colorado cities, Boulder (City of Boulder 1996) and Fort Collins (City of Fort Collins 1998), with sizeable prairie dog populations, illustrate an integrated approach to managing those populations and reducing conflicts. Each city established an advisory committee to address and resolve the management issues. Many elements and techniques are being used in an integrated management strategy, including habitat management, population management, and people management (Witmer et al. 2000). It should be noted, however, that the possible techniques can vary greatly in their effectiveness, cost, and public acceptability (Witmer 2007). For example, barriers are a popular approach to stop colonies from expanding to adjoining landowners' properties where conflicts will occur. However, adequate barriers are expensive to build and maintain and only provide limited containment of the colony (Witmer et al. 2008). Additionally, resource managers are often limited in their management options by budgetary, legal, and socio-political constraints. For example, while several rodenticides are registered for prairie dog control (Witmer and Fagerstone 2003), these are often not socio-politically acceptable, especially in urban/suburban settings.

Fertility control offers another potential solution to control expanding prairie dog colonies. The topic of wildlife fertility control was recently reviewed, including chemicals, delivery systems, advantages, disadvantages, regulatory issues, and challenges (Fagerstone et al. 2010). Previous field studies (Nash et al. 2007; Yoder 2009) indicate that the steroid diazacholesterol can effectively limit prairie dog reproduction if delivered in adequate amounts to the animals over a sufficiently long period of time before the breeding season. The chemical inhibits enzymes required for cholesterol production; hence, production of reproductive hormones from steroid precursors is prevented (Nash et al. 2007). Unfortunately, an efficient way to deliver adequate amounts of the chemical to prairie dogs over an adequate period of time is problematic. If a palatable, long-lasting food bait block system could be developed that prairie dogs would readily feed on, the steroid could potentially be incorporated. This would provide a more cost-effective method of controlling prairie dog fertility and minimizing colony expansion, thus reducing resultant conflicts.

Our objective was to determine the palatability and acceptance of a food bait block by free-ranging prairie dogs. We hypothesized that a commercially-available non-toxic commensal rodent detection food block would be readily accepted by prairie dogs. If that was the case, we will plan to incorporate diazacholesterol into a similar food bait block and test its acceptance in a subsequent field trial.

STUDY AREA AND METHODS

We obtained permission to test a food bait block in a prairie dog colony at the Fort Collins-Loveland Airport, Fort Collins, Colorado. The study was conducted in the winter as this is the time of year that a fertility control material would need to be delivered (i.e., prior to the onset of the prairie dog breeding season). The preliminary food bait block that we tested was DeTex Blox (Bell Laboratories, Inc., Madison, WI). These blocks were developed to detect the presence of commensal rodents. They are rectangular (5 x 2.5 x 2 cm) and have a hole through them so that they can be mounted on wire posts in bait stations. The baits contain ground grains, various flavorings attractive to commensal rodents, and paraffin to increase environmental longevity. The baits also contain 0.2% pyranine, a biomarker that fluoresces when exposed to ultraviolet ("black") light. Thus consumption of the food bait blocks could be confirmed by examining feces or tissues using an ultraviolet lamp.

We placed 10 food blocks in a stack using 1.2 m long, small diameter (0.8 cm) steel rods at each of 6 sites (labeled A-F) that were inserted into the soil in a vertical orientation (see Figure 1). Each block weighed, on average, 20 g so the 10 blocks on the pole weighed about 200 g. By using the poles, as the blocks were fed upon, additional blocks slid down the steel poles and become available to the prairie dogs over time. This was necessary to minimize disturbance of the animals, but also to assure that they have enough material to feed on for at least several days before replacement was needed. Bait availability of at least 10-14 days is the amount of feeding time required for the steroid concentration to build up in the animals' bodies to a level that will inhibit reproduction. Food bait "poles" were placed near burrows in the colony. A group of 4 poles was placed near burrows that were at least 30 m from another group of poles so that each pole group was exposed to different prairie dogs (i.e., different coteries which are extended family groups which defend an area from other prairie dogs). Animal activity near the poles was observed from a distance by study personnel. Additionally, infra-red motion-sensitive cameras were used to monitor animal activity, especially at night so that nocturnal, non-target animal (i.e., rabbits, other rodents) use of the food blocks could be determined. Food block poles were maintained in place for 12 days at 2 sites and 19 days at 4 other sites. The 10 food blocks were maintained over that time period by adding additional food blocks to each pole every 2-3 days as needed. When examined, if half or more (i.e., 5 or more) of the food blocks remained on a pole, that pole was left alone until the next check day. If less than 5 blocks remained, they were removed and placed in a labeled, sealable plastic bag for later weighing. Ten new food blocks were then placed on that pole. This process allowed us to determine the total amount consumed at each pole at the end of the field trial. To provide replication, 6 sites, with 4 food bait block poles each, were randomly assigned to locations in the prairie dog colony.

We also placed food blocks in 8 burrows to test whether or not the prairie dogs would feed on them in the burrows. This was done by attaching 2 food blocks to the end of a 1 m long piece of thin wire. The blocks were dropped into the burrow, but the other end of the wire was staked to the ground a short distance from the burrow opening. This was done so that the blocks could be retrieved to examine for consumption. Wires with blocks were examined every 2-3 days over a 15 day period. Food blocks were replaced as needed.

The mean and standard deviation of the amount (weight) of food bait blocks consumed was determined and compared between sites and days with *t*-tests and ANOVA, using Statistix Version 9 (Analytical Software, Tallahassee, Florida). A *P* value of ≤ 0.05 was considered to indicate a significant difference. Activity of prairie dogs and non-target animals at or near food bait poles was described qualitatively based on remote, motion-sensitive camera pictures, and to a lesser extent, by direct observation.

RESULTS

Food blocks on the metal poles were readily fed upon at all 6 sites to the extent that they had to be replaced every 2-3 days (Table 1; Figure 1). There was no significant difference ($F = 0.55$, $P = 0.6603$) in the amount removed from the poles at the 4 sites (A, C, E and F) that were operated for the same length of time. There was also no significant difference ($t = 1.31$, $P = 0.2394$) in the amount removed from the poles at the other 2 sites (B and D) that were operated for the same length of time, but a shorter period than the previously mentioned 4 sites. The mean amount removed per site was 81% of the food bait presented. There was significantly less ($t = 5.67$, $P = 0.0002$) removed when the food blocks were first put out (i.e., amounts removed on Day 3 versus Day 5), perhaps because of neophobia to the new objects on the landscape. After Day 3, however, food removal from the poles remained high across sites, although significantly more ($F = 6.54$, $P = 0.0029$) was removed on some days rather than others, perhaps because of varying weather conditions. For example, on Day 10 only 24.8 food blocks were removed from the 4 poles, on average, at each site versus all 40 food blocks being removed on Day 8.

It appeared that the food blocks may have been consumed in the burrows, but we cannot definitively conclude that was the case. Most often, both food blocks were gone when the wire holding them was checked. The number of blocks consumed did not differ significantly ($F = 1.97$, $P = 0.0884$) between the 8 burrows used. However, about half of the times that the wires were checked, the wire was found to be outside the burrow with the food blocks missing. It is possible that animals pulled or pushed the blocks out to the surface before feeding on them or they may have consumed them in the burrow and then pushed the wire out. While we used cameras at these burrow sites for a few days, we could not conclude whether prairie dogs or rabbits were mainly consuming the blocks. The pictures often showed the wire extending into the burrow and then the next picture (taken 15 minutes later because we were using a time-delay mechanism), would show the wire out of the burrow. In a few cases, pictures showed prairie dogs feeding on the blocks outside of the burrow, but a few nighttime pictures also showed rabbits and mice feeding on the blocks outside of the burrows.

The remote cameras captured 948 daytime pictures of prairie dogs in the vicinity of the poles, often gnawing at the food blocks (Figure 1). As many as 7 individual prairie dogs were on the surface at a site with poles at one time. No nighttime pictures of prairie dogs were obtained which was expected as the species exhibits diurnal activity patterns. In addition to daytime pictures, the infrared lighting system of the cameras resulted in numerous nighttime pictures of animals, mainly mice and rabbits (Figure 2). A total of 2,422 pictures had rabbits (*Sylvilagus* spp.) in them, while 311 pictures had mice (*Peromyscus* spp.) in them. There were significantly more (F = 10.27, P = 0.0016) pictures of rabbits than prairie dogs or mice. There were significantly more (t = 4.23, P = 0.0018) pictures of rabbits at night (2,388) than during the day (34), showing primarily nocturnal activity patterns. As many as 6 individual rabbits were on the surface at a site with poles at one time. We also obtained a small number of pictures of diurnal birds (mainly larks and sparrows), one picture of a coyote (*Canis latrans*), and one picture of a nocturnal owl swooping near the ground surface.

It was clear from the pictures that prairie dogs were the main species feeding on the food blocks during the day. However, the pictures also made it clear that rabbits (and to a lesser extent mice) were feeding on the food blocks at night. By noting the number of food blocks on the poles at the end of the day and again in the morning, we estimated that the rabbits were consuming significantly more (t = 2.46, P = 0.0335) of the food blocks at night than the targeted species, prairie dogs, during the day (Figure 3).

We collected some pellets from 20 different prairie dog fecal groups. Eight of the 20 samples (40%) fluoresced under ultraviolet light. We also collected one sample of mice fecal droppings and this fluoresced, but neither of the two samples collected of rabbit fecal pellets fluoresced.

Table 1. Amount (g) of food bait consumed at each pole and each site[a].

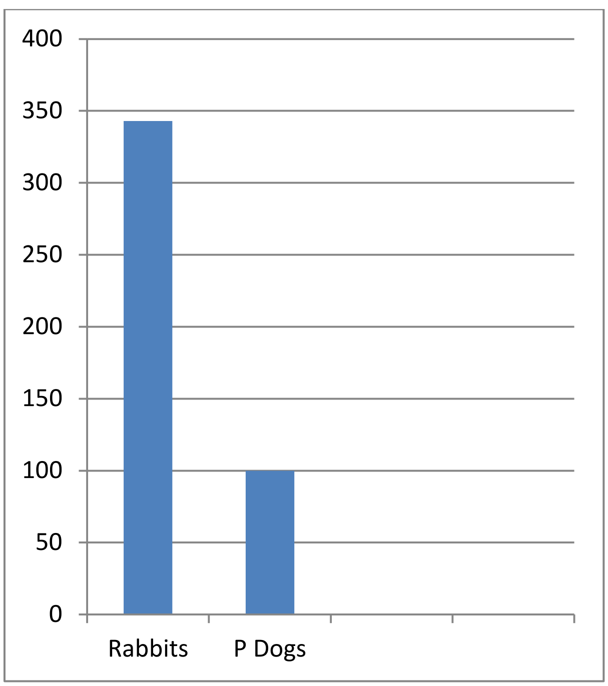
	Site A	Site C	Site E	Site F	Site B	Site D
Pole 1	1154	1204	1012	1003	802	970
Pole 2	1204	1168	1130	1139	802	739
Pole 3	1170	1003	1112	1140	802	571
Pole 4	1404	1300	1244	1361	1003	569
Mean (S.D.)	1233.0	1168.8	1124.5	1160.8	852.3	712.3
	(115.9)	(123.8)	(95.1)	(148.2)	(100.5)	(189.4)
% Removed	87.8	83.2	77.3	80.9	85.0	71.0

[a]Sites A, C, E and F were operated for 19 days with a total of 1404.2 g of food bait was presented, whereas Sites B and D were operated only 12 days with a total of 1003 g of food bait presented.

Figure 1. Photograph of prairie dogs feeding on the food bait blocks.

Figure 2. Photograph of rabbits eating food bait blocks at night.

Figure 3. Estimated total number of food bait blocks consumed by rabbits versus prairie dogs.



DISCUSSION

There are a number of challenges to be overcome before a fertility control material can be used to control rodent populations. First, an oral delivery system must be developed as direct injection of each rodent is not practical, although there is a product registered for injection of white-tailed deer (*Odocoileus virginianus*; Miller et al. 2000). An oral delivery system would be most practical for seasonally breeding rodent species (e.g., prairie dogs) versus continuously breeding species (commensal rats, *Rattus* spp., and house mice, *Mus musculus*).

The second challenge is achieving species specificity in the delivery system so that only the targeted species is rendered infertile. We identified an effective delivery system to get a fertility control material to free-ranging prairie dogs over a period of time, thus reducing labor and travel requirements. However, the lack of pyranine dye in 60% of the prairie dog pellet groups examined suggests that not all prairie dogs are consuming the food bait blocks. This could be due to dominance hierarchies in the coteries. We caution, however, that only a small number of pellet groups were examined for fluorescence and some of the pellet groups may have been older (i.e., excreted by animals before the food bait blocks were available for several days). If this fertility control delivery system is to be pursued further, the next requirement would be to incorporate the diazacholesterol into a palatable food bait block for testing in the field. This might require collaboration with a rodenticide manufacturing company.

As such, it appears that it may be possible to overcome the first challenge of an oral delivery system. Additional effort will be required to overcome the second challenge of species specificity of the fertility control delivery system. We could not determine if placement of the food blocks in the burrows reduced non-target animal consumption. Based on the camera pictures, the main non-target exposure of food bait blocks on poles was to rabbits and this occurred mainly at night. Hence, it might be possible to develop an automated system that will uncover the food bait blocks during the day to allow prairie dogs to feed on them, but then cover the food bait blocks at night to restrict feeding by rabbits and mice. Such a device could be powered by battery or solar panel.

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A Review of Wildlife Hazard Mitigation Techniques on General Aviation Airports

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ABSTRACT: Large commercial airports, also known as Part 139 airports, are required by federal regulation to monitor and control wildlife activity. Due to the regulatory nature of 14 Code of Federal Regulation (CFR) Part 139.337, and the size and scope of these airports, there is sufficient funding to support wildlife management. However, in the United States, there are an additional 19,000 landing facilities, of which 4,600 are known as public use, general aviation airports. These general aviation airports are not bound by any regulation to mitigate wildlife hazards at their facilities; however, at least 33.9% of these airports have known wildlife hazards. Due to their small and often non-commercial nature, general aviation airports have limited operational budgets and often must solve wildlife hazards with existing personnel. Because these personnel are often not trained in wildlife management techniques, they may be unaware of suitable options for controlling wildlife damage. Therefore, we reviewed existing wildlife damage management techniques that are commonly used at Part 139 airports and surveyed airport wildlife damage management professionals to assess the techniques for use at general aviation airports based on the initial costs of implementation; the amount of training required to implement the techniques; perpetual costs; and the amount of man hours per week required to implement the technique. All techniques were scored on a 5-point scale for each category, resulting in a composite score. This review may serve as a guide in the decision making process for general aviation airport managers when considering wildlife management at their airports.

Key Words: airport, bird strike, cost, damage management, GA, general aviation, mitigation, survey, wildlife strike

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INTRODUCTION

Since the first flight of an airplane by Wilbur and Orville Wright in 1903, air transit has become an integral part of the global economy, generating billions of dollars annually. The first bird strike, a red-winged blackbird (*Agelaius phoeniceus*), was recorded by the Wright brothers in 1905. The first human fatality as a result of a bird strike (gull sp. [Laridae]) was recorded in 1912 (DeVault et al. 2013). Over time, the annual number of aircraft operations has increased and aircraft have become faster and quieter (DeVault et al. 2013). The combination of these factors has resulted in an increase in the number of wildlife strikes. Following the implementation of electronic reporting methods, the number of wildlife strike reports has risen. Of the 142,603 strike reports filed between 1990 and 2013 (a 24 year period), 11,315 (8%) were filed in 2013. The number of strikes filed in 2013 is 611% higher than the number filed in 1990 (FAA 2014). These strikes caused damage totaling $103 million in 2013 to commercial aircraft in the United States alone. It is estimated that at least $937 million have been lost since 1990 due to wildlife strikes (FAA 2014). These figures do not take into account monetary losses due to labor costs or flight schedule changes (USDA 2005). Monetary losses aside, wildlife strikes to aircraft can also be deadly, with 255 individuals killed in the United States since 1988 (FAA 2014).

Because of the risk to human life and the potential of negative economic impact, much research has been undertaken in various disciplines to manage wildlife in and around airports, with the primary goal of minimizing the risk posed by wildlife to aircraft and their contents. The existence of the United States Department of Agriculture, Animal and Plant Health Inspection Service (USDA APHIS) Wildlife Services National Wildlife Research Center (NWRC) facility dedicated to researching wildlife hazards to aircraft indicates the importance of this type of research. The studies conducted by the NWRC and others include landscape level planning (Blackwell et al. 2009), habitat manipulation (Blackwell et al. 2008), the deterrence of a particular species of concern or even individual animals (York et al. 2000), and other avenues of research. This research has led to the development of a variety of methods used to mitigate wildlife damage at airports during the past 50 years. To address wildlife strike hazards, each airport must be evaluated separately for wildlife habitat, species present, and the flight operations characteristic of the airport. Because of the unique characteristics of each airport, there is no standard wildlife management plan that can be implemented. Each technique that is to be used must be evaluated by airport wildlife managers for its efficacy, environmental impact, impact on flight safety, and human dimensions (Cleary and Dolbeer 2005). Detailed descriptions of, and instructions on, the proper implementation of these methods are available from many sources including the Federal Aviation Administration (FAA [Cleary and Dolbeer 2005]), the Airport Cooperative Research Program (ACRP [ACRP 2010]), and branches of the Department of Defense (U.S. Air Force 2004, Commander, Naval Installations Command 2010). However, many of these guides are designed for larger airports that can train and employ full-time personnel or contract with wildlife biologists to control wildlife on a regular basis.

In the United States, all airports serving regularly scheduled passenger-carrying operations with aircraft designed with more than 9 passenger seats, or unscheduled passenger-carrying operations of aircraft with 31 or more seats, are governed by 14 Code of Federal Regulation (CFR) Part 139. The regulations in 14 CFR Part 139, among others, set standards for firefighting equipment, airport signage, security procedures, and also require that airports certificated under 14 CFR Part 139 mitigate wildlife hazards to aviation safety as they become known.

14 CFR Part 139.337(a): In accordance with its Airport Certification Manual and the requirements of this section, each certificate holder must take immediate action to alleviate wildlife hazards whenever they are detected.

As of 30 July 2014, there were 542 airports that operated under 14 CFR Part 139 (referred to as Part 139 airports).

With this legal mandate, many of these airports have extensive wildlife management departments consisting of either trained airport personnel or contracted entities. Regardless of who conducts wildlife management on Part 139 airports, if certain wildlife hazard conditions are met, a wildlife damage biologist, having professional training in wildlife hazard management at airports, or their designee must complete a wildlife hazard assessment (14 CFR Part 139.337). Due to the regulatory nature of 14 CFR Part 139, airport managers provide funding to conduct wildlife management and wildlife hazard mitigation. However, in the United States, there are an additional 19,000 landing facilities (e.g. heliports, seaplane bases, and runways), of which 4,610 are public use, general aviation airports, seaplane bases, glider bases, balloon ports, ultralight ports, or heliports (hereafter referred to as general aviation [GA] airports)(FAA 2015). These GA airports are not bound by any regulation to mitigate wildlife hazards at their facilities; however, many of these airports have known wildlife hazards.

Due to their small, often non-commercial nature, GA airports have limited operational budgets, frequently comprised of funds allocated by local municipalities and funding from the United States Department of Transportation (ACRP 2010). These GA airports often have limited staffing (ACRP 2010). It is not uncommon for the airport manager to be the sole employee of the airport. Therefore, that sole employee is often tasked with keeping facilities in working order, maintaining the airport, and conducting traditional managerial activities. Many general aviation airports are often located in rural areas rather than in metropolitan areas, as are many Part 139 airports (ACRP 2010). This factor regularly places airports in close proximity to agriculture, timber production, landfills, and protected natural areas (ACRP 2010). All of these neighboring land uses frequently are associated with wildlife, thereby contributing to wildlife hazards on rural airfields (Cleary and Dolbeer 2005).

In addition to being rural, many GA airports have a low operational tempo. They may only see a few flight operations each day. This low tempo creates a situation where wildlife are not habituated to avoiding areas adjacent to aircraft movement surfaces. General aviation airports are often characterized by the types of aircraft they service: mostly piston-powered light aircraft. Many light aircraft are not hardened against wildlife strikes, like commercial aircraft, since they are not mandated to be so under 14 CFR Part 25. As such, what might be a relatively minor strike to the windscreen, engine, or control surface of a commercial aircraft could be catastrophic to a light aircraft. Though strikes to GA aircraft comprised only 15% of the total number of reported strikes in 2013, the true number of strikes is likely much higher since strike reporting is not mandatory and is not widely practiced in the GA community, likely due to the fact that knowledge of wildlife strike reporting is not required on the FAA recreational pilot or private pilot written tests. (FAA 2014, FAA 2015). There are no data available detailing the prevalence of wildlife hazards, or the species that pose those hazards at GA airports.

Because these GA airports are often lacking in funding, they often attempt to control wildlife using existing personnel. Smaller airports with more limited resources are often not considered when developing manuals or other materials that provide guidance to airport managers. Though there is 1 manual written for GA airport wildlife management (ACRP 2010), there is still a large knowledge gap between GA airport managers and professional airport biologists who are legally required to conduct wildlife hazard assessments and are commonly employed at Part 139 airports. Oftentimes, GA airport managers are frequently left to their own knowledge when examining the feasibility of beginning a wildlife damage management program at their airfield. This may result in inefficient allocation of resources, inefficient wildlife management, and frustration by the airport manager. This may also result in airport managers implementing unsafe, harmful, or even illegal wildlife management methods.

METHODS

To determine how widespread wildlife conflicts were at GA airports, we obtained a spreadsheet of all public use landing facilities in the United States from the FAA website. We removed all Part 139 airports from the list, leaving only GA landing facilities. We also removed balloon ports, glider ports, and ultra-light ports, as they comprised 0.2% of GA landing facilities. We assigned all remaining facilities an identification number from 1 to 4,600. We used a random integer generator to generate 463 random integers between 1 and 4,600. According to Bartlett et al. (2001), for categorical data with a population of approximately 4,000 and a margin of error of 0.05, we would require a sample size of at least 351 airports to have a representative sample of GA airports. We manually searched for each facility corresponding with a generated random integer in the Aircraft Owner's and Pilot's Association (AOPA) Airports online database and assessed whether any remark for wildlife hazards existed. We used the AOPA Airports online database because it compiles aeronautical information from multiple FAA sources and is updated on the FAA update cycles (AOPA We categorized wildlife hazard remarks as warning of waterfowl, birds (not specifying any guild), deer, swine, elk, antelope, gulls, coyotes, cervids (as a guild), or a generic wildlife hazard remark. We then separated landing facilities by type into 3 categories, seaplane base, heliport, and airport), and analyzed the rate of wildlife hazard remarks between types of landing facilities.

To determine what airport biologists would typically choose to use at these non-Part 139 airports, we created a SurveyMonkey® poll that listed wildlife hazard mitigation techniques that were commonly implemented at Part 139 airports which was distributed in the Wildlife Damage Working Group through their quarterly newsletter, *Interactions* (Lewis 2015) and to Wildlife Services biologists who routinely work at airports. We asked respondents to assess each technique for initial procurement costs, training time and costs, amount of time required per week to properly implement the strategy, and the recurring costs of maintenance and expendables using a Likert scale. Respondents were instructed to evaluate only the methods that they were familiar with. Each category was given a score from 0 to 5, representing no costs, nominal costs, low costs, moderate costs, high costs, and prohibitive costs, under normal funding circumstances, respectively. We defined each score as follows, and gave no further guidance on the scores:

0 (None): No cost/time

1 (Nominal): Very low cost/time

2 (Low): Limited cost/time that can be committed with little consideration.

3 (Moderate): Cost/time investment that must be considered. Not insignificant.

4 (High): Cost/time investment that must be carefully weighed.

5 (Prohibitive under normal circumstances): Cost/time investment that is beyond the normal scope of operations for an airport.

The scores for each category were summed, resulting in a composite score.

The Murray State University Institutional Review Board (MSU IRB) was consulted prior to distribution of our survey. They found that this was not human research and thus did not require MSU IRB permission.

RESULTS

Of the GA landing facilities that were searched (n=463), 33.9% (n=157) had a wildlife hazard remark in AOPA Airports. When analyzed by landing facility, 35.4% of airports (153/432), 16.7% of seaplane bases (4/24), and 0% of heliports (0/7) had "wildlife hazard" remarks.

We found that 30% of all sampled airports that reported a wildlife hazard, reported more than 1 species or guild as presenting a hazard at that airport. We also found that deer (51.6%) were the most common animal or guild identified and reported as a hazard at airports, followed by birds (31.9%), and a general wildlife hazard remark (21.7%) (Table 1).

We found that snag removal and manual harassment had the lowest composite scores (5.6 and 6.3, respectively) while trained raptors and avian radar had the highest composite scores (14.3 and 15.5, respectively) (Fig. 1 and Fig. 2)

Figure 1: The results of a 2015 survey of 17 professional airport wildlife biologists asked to evaluate the costs associated with implementing various airport wildlife damage management techniques with 95% confidence interval bars show.

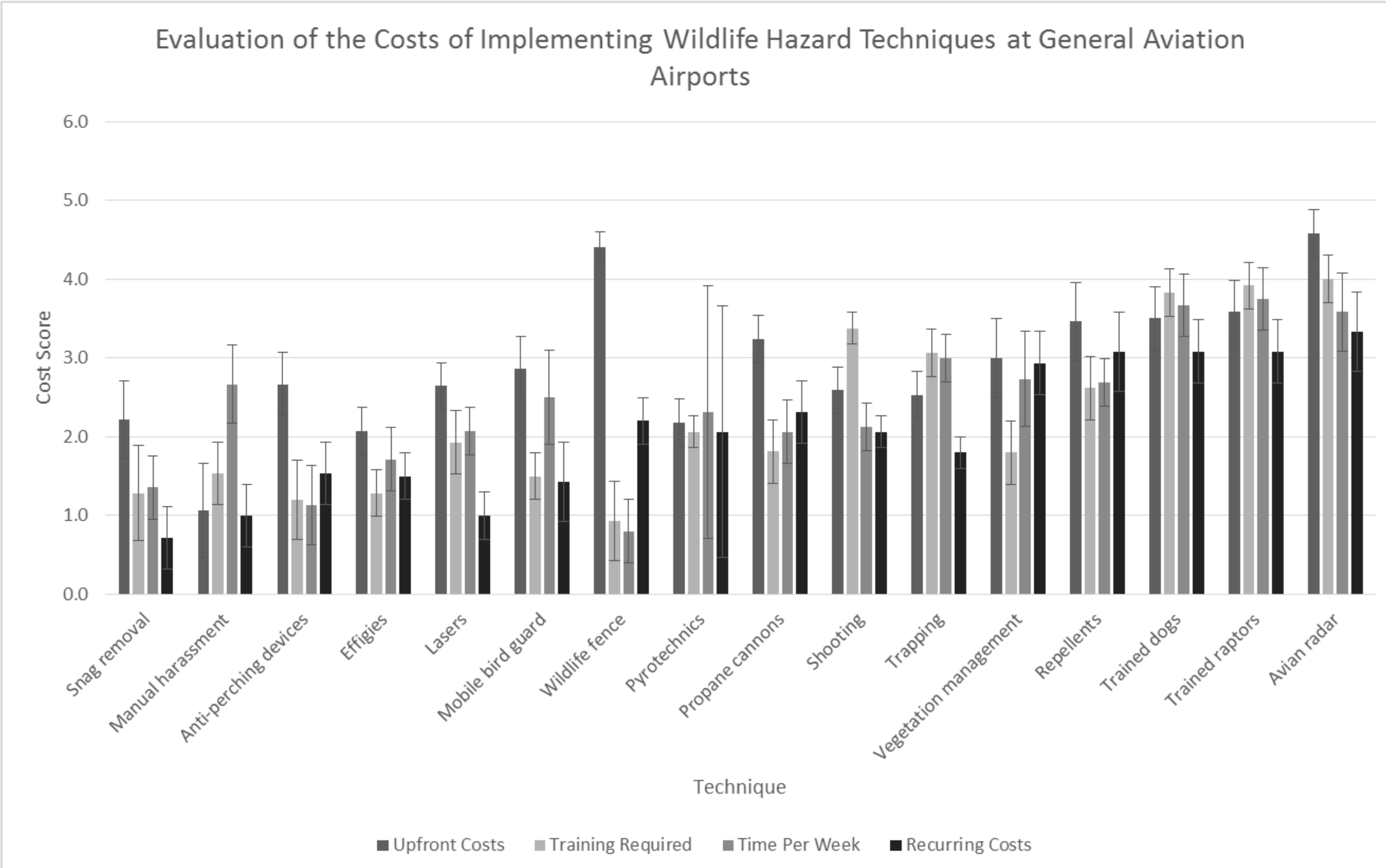

Figure 2: Composite scores of a 2015 survey of 17 professional airport wildlife biologists asked to evaluate the costs associated with implementing various airport wildlife damage management techniques.

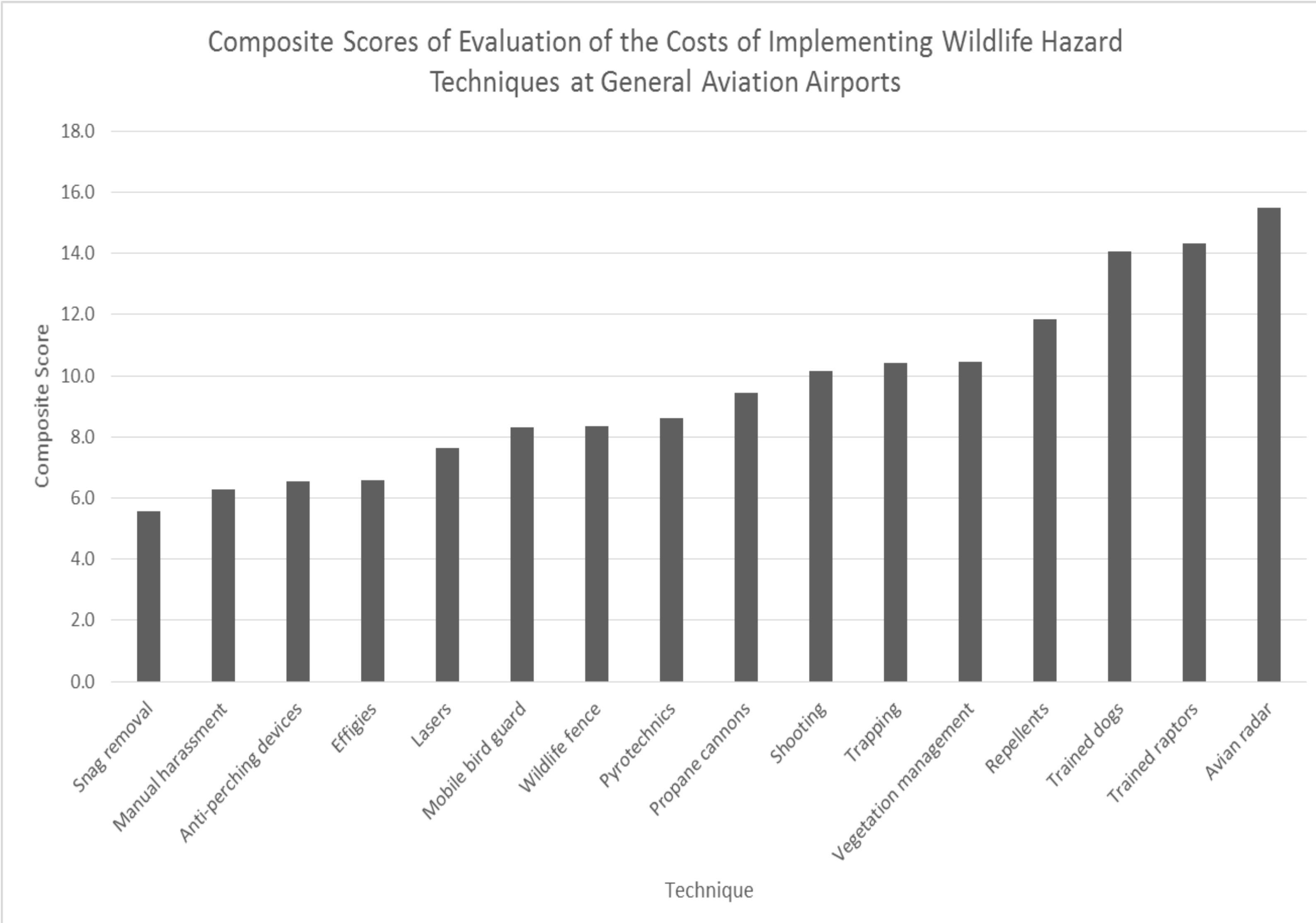

Table 1. Species and guilds identified as hazards to aviation during a February 2015 survey of wildlife hazard remarks at general aviation airports in the Aircraft Owners and Pilots Association Airports database. We surveyed 463/4,600 airports. Of the 463 airports surveyed, 157 had a wildlife hazard remark, with 30% describing more than one species or guild.

Type of Hazard	No. with Remark	% with Remark
Deer	81	51.6%
Birds	50	31.9%
General Remark	34	21.7%
Waterfowl	21	13.4%
Big Game	5	3.2%
Antelope	4	2.6%
Gulls	3	1.9%
Coyotes	3	1.9%
Swine	1	0.6%
Elk	1	0.6%

DISCUSSION

We found that 33.9% of GA airports had reported a wildlife hazard. This value only represents those airports that have recognized a hazard and have chosen to report it. Therefore, a lack of wildlife hazard remark does not necessarily mean that there is not a wildlife hazard present at that airport. Since there is no legal mandate to report wildlife hazards at GA airports, the true percentage of GA airports with wildlife hazards is certainly much higher.

Deer were the species most often identified as a wildlife hazard at airports. Deer are large, easily recognizable, and plentiful across the United States (Conover et al. 1995, McShea 2012). The frequency with which they are identified as a hazard could be due to limited funding at airports, resulting in no perimeter fence and easy access to the airfield for deer. It could also be due to the familiarity that the public has with deer-vehicle collisions. People understand, and often have witnessed, the damage that a deer-vehicle collision can have. Therefore, it is likely that they readily understand deer to be a catastrophic hazard to aircraft and readily remark even on limited numbers of deer as a wildlife hazard.

Birds were the second most often identified group of wildlife that were reported to pose a hazard at general aviation landing facilities. The generic use of the term “bird” masks the species and guilds that pose the largest hazards at general aviation facilities. This could be due to the large number of bird species that frequent airports, belonging to many different guilds, and a lack of skill or effort to identify birds that frequent each airport. A general wildlife hazard remark was the third most reported wildlife hazard remark. Similar to the “birds” remark, this generic term masks the species or guilds that pose the greatest hazards at general aviation facilities. This could also be due to a lack of skill in wildlife identification or a lack of effort to identify individual species or guilds.

We found much similarity among the responses of airport wildlife biologists regarding the costs associated with the implementation of various wildlife hazard management techniques. Responses for each technique generally had low variance (Fig. 1). This could be due to standardization of training.

The responses for shooting, pyrotechnics, and manual harassment were higher in the time per week and recurring costs categories than we had expected. This could be due to the fact that the respondents are full time airport wildlife biologists at large commercial and military airfields. In those situations, the amount of time and resources devoted to each technique may be much higher. For instance, a GA airport manager may only fire 50 pyrotechnics each month, yet a biologist at a large airfield may fire

50 pyrotechnics each day as a part of his daily duties, thereby increasing the time per week and recurring costs of this technique (Biondi et al. 2014).

Biologists reported that techniques such as anti-perch devices, snag removal, and manual harassment, had relatively low costs associated with their implementation. These techniques could likely be implemented on most GA airports without additional funding sources. Techniques such as pyrotechnics, shooting, lasers, and propane cannons had intermediate costs associated with their implementation. Some airports wishing to implement these techniques may need to seek external funding sources. Biologists reported that techniques such as repellents, trained animals, and radar had high costs associated with their implementation. These costs may be high enough that a GA airport wishing to implement these techniques must seek additional funding sources. These funding sources may include FAA Airport Improvement Program Grants, state Departments of Transportation, or local sources (Maryland Aviation Administration 2014). Though wildlife fences had high initial costs, their efficacy in excluding mammals from the airport environment as well as the measure of security they give to the airfield makes them a viable option for an airport that can secure external funding to construct it, but does not have large amounts of time to dedicate to it in the future. Avian radar was rated the most expensive technique overall. These costs, combined with the fact that avian radar does not directly mitigate wildlife hazards, reduces the utility of this technique on a GA airport.

We did not ask our survey respondents to evaluate the efficacy of various wildlife damage management techniques. While there is no ideal damage management technique, there are techniques that are more effective than others in a given situation. While this is a potential weakness of our survey, there are many documents that detail the efficacy of different management techniques (U.S. Air Force 2004, Cleary and Dolbeer 2005, ACRP 2010, Commander, Naval Installations Command 2010). Each airport must be individually evaluated for its specific hazard and mitigation techniques selected to reduce a particular hazard in particular environments.

MANAGEMENT IMPLICATIONS

We have shown that at least 33.9% of surveyed GA airports have reported a known wildlife hazard. Given that GA airports are under no legal obligation to report wildlife hazards, the actual percentage of GA airports with wildlife hazards is likely much higher. In addition, 51.6% of the surveyed airports reporting a hazard reported deer and 13.4% reported waterfowl. These specific guilds pose 2 of the greatest threats to aircraft, largely due to their body size (Wright et al. 1998, FAA 2014). Given that airport wildlife management training is readily available, as it is required for employees of those Part 139 airports that require a wildlife hazard assessment, managers of GA airports should receive training as well. This training will aid in the identification of hazardous species and also aid in the reporting of more wildlife strikes to aircraft. We suggest that GA airport managers and/or their employees contact nearby Part 139 airports to inquire about taking the Part 139 wildlife training.

This amount of risk serves to highlight the need for GA airports to consider the possibility of addressing wildlife hazards at their facilities. Lack of monetary resources often forces GA airports to reject the possibility of managing wildlife to reduce the risk to aviation (ACRP 2010). Our research has evaluated wildlife hazard mitigation techniques that are commonly implemented on Part 139 airports for the costs associated with their implementation. This should give airport managers who are wholly unfamiliar with wildlife management an idea of the relative amount of resources that will have to be devoted to each technique when the manager is considering the unique needs and fiscal situation of the airport. Knowing which techniques are fiscally feasible and which are not, will make the literature review for the implementation of wildlife hazard management techniques more efficient and productive for the airport manager. Before any wildlife hazard mitigation techniques are implemented, airport managers must positively identify the species or guilds that pose risks to aviation safety. If this is not done, airport managers may select techniques that will not properly address the species or guilds causing risks.

Further research on this topic should include surveys among GA airport managers regarding knowledge of, and attitudes towards wildlife hazards and wildlife strike reporting. These surveys should include questions such as: do you consider a wildlife hazard to be present at your airport, if so, what species; has there ever been a wildlife strike at your airport, if yes, was it reported; do you know how to report wildlife strikes; do you actively manage wildlife at your airport; and are you aware of wildlife management resources available to you? Further research should also be conducted examining usage rates, among professional airport wildlife biologists, of the various wildlife mitigation techniques we listed.

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Speed Kills: Effects of Vehicle Speed on Avian Escape Behavior

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ABSTRACT: The avoidance of vehicles is a common challenge for birds in the modern world. Birds generally rely on antipredator behaviors to avoid vehicles, but modern vehicles are faster than predators. We predicted that birds may be unable to accurately estimate the speed of approaching vehicles, which could contribute to miscalculations in avoidance behaviors and cause collisions. We tested our prediction in two studies. In the first (DeVault et al. 2014), we baited turkey vultures (*Cathartes aura*) to roads with animal carcasses and measured flight initiation distance (FID) when driving a truck towards them at 30, 60, or 90 km/h. Despite a wide range of responses, FID of vultures increased by a factor of 1.85 as speed increased from 30 to 90 km/h. At 90 km/h there was no clear trend in FID across replicates; birds were equally likely to initiate escape behavior at 40 m as at 220 m. Seventeen percent of vehicle approaches at 90 km/h resulted in near collisions with vultures, compared to none during 60 km/h approaches and 4% during 30 km/h approaches. In the second experiment (DeVault et al. 2015), we used video playback to investigate escape behaviors of captive brown-headed cowbirds (*Molothrus ater*) in response to virtual vehicles appearing to approach at speeds ranging from 60-360 km/h. Flight initiation distance remained similar across vehicle speeds, indicating that avoidance behaviors in cowbirds were based on distance rather than time available for escape. Cowbirds generally did not initiate flight with enough time to avoid "collision" when virtual vehicle speed exceeded 120 km/h. Although potentially effective for escaping predators, the decision-making processes used by turkey vultures and cowbirds in our experiments appear maladaptive in the context of avoiding vehicles, and may represent important determinants of bird-vehicle collisions.

Key Words: birds, brow-headed cowbirds, escape behavior, speed, turkey vulture vehicle

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Habituation of White-tailed Deer (*Odocoileus virginianus*) in an Urban/Suburban Environment to an Unmanned Aircraft System (UAS) quad copter

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ABSTRACT: The use of remote control Unmanned Aircraft Systems (UAS) with photographic instrumentation has the potential to be a useful tool for various aspects of wildlife management. However, if the presence of an UAS significantly alters normal behavior, use of these devices may be limited. Therefore, the objective of this study was to evaluate behavioral changes of white-tailed deer (*Odocoileus virginianus*) when repeatedly exposed to a commercially available UAS. We hypothesized that white-tailed deer in an urban/suburban environment would rapidly become habituated to the presence of an UAS. Deer in two hay fields on the Berry College campus were subjected to 1 UAS flight per day for 10 consecutive days. Each flight consisted of 2 overhead passes by the UAS at an initial height of 50 m above the ground followed by 2 passes at 40 m, 30 m, and 20 m altitude. Digital camcorder recordings at ground level were obtained during each flight from a minimal distance of 100 m from the deer. Behavior of deer during 12 predefined, 10 sec components of each flight, within the field of view of the digital camcorder, were categorized as Passive (no altered behavior), Alert (actively observing and/or listening toward the UAS), Active (slow to moderate movement away from area), or Flight (running away from area). The average number of deer observed during each flight was similar ($P \geq 0.05$) at each respective location (12.1 ± 3.9; 12.8 ± 5.6). There was an increase in Passive Behavior ($P \leq 0.05$) and a corresponding decrease in Alert Behavior ($P \leq 0.05$) of deer as the number of flights and subsequent exposure to the UAS increased. Too few observations of Active or Flight Behavior were recorded to provide meaningful interpretation. The results of this study indicate white-tailed deer in an urban/suburban environment can readily become habituated to the presence of an UAS with repeated exposure.

Key Words: behavior, habituation, UAS, white-tailed deer

INTRODUCTION

The rapid advancement and availability of various platforms of Unmanned Aircraft Systems (UAS) have resulted in a proliferation of potential uses for these devices. Classification of the different types of vehicles available for civilian use has primarily been a result of application of military descriptions based on size, endurance, capabilities, and physical conformations of the vehicles (Watts et al. 2012). Terminology used to describe different platforms also continues to evolve, including Remotely Piloted Vehicle (RPV), Unmanned Aircraft (UA), Unmanned Aerial Vehicle (UAV), and the more recent term of Unmanned Aircraft System (UAS) (Watts et al. 2012, Gupta et al. 2013). An UAS is described as an air vehicle and associated equipment that does not carry a human operator and flies by autonomous control or remote piloting (Gupta et al. 2013). Regardless of classification, the primary civilian use at this time is for surveillance.

Over the past decade, there has been a proliferation of proposed and documented use of various UAS platforms for environmental monitoring. Unmanned aircraft system imaging has been used for monitoring vegetation, including rangeland (Quilter and Anderson 2001, Rango et al. 2006, Laliberte et al. 2011) and various types of forests (Tomlins and Lee 1983, Paneque-Galvez et al. 2014). Agricultural applications documented suggest that UAS have been useful for evaluating soil erosion (d'Oleire-Oltmanns et al. 2012), vineyard status (Baluja et al. 2012), and detection of diseases of citrus trees (Garcia-Ruiz et al. 2013). Monitoring the status of fires (Ambrosia et al. 2003), avalanche zones (Watts et al. 2012) and oil spills (Allen and Walsh 2008) has also been reported as a use of these devices. It should also be noted that UAS have significant use and potential for human surveillance such as law enforcement and border patrol efforts (Gupta et al. 2013).

The potential of UAS applications for wildlife management objectives, particularly those typically involving low-altitude aerial surveys using conventional aircraft, are evident. According to Wiegman and Taneja, (2003) crashes of light aircraft while conducting aerial surveys are the leading cause of death for wildlife researchers. Manned aerial surveys also tend to have a high cost/hour flight for the aircraft operation, and significant additional expenses related to personnel and logistic considerations such as working within airport constraints (Watts et al. 2010). Watts et al. (2010) further reported problems with survey repeatability, restrictions due to climatic conditions, and challenges with small special scales or area access when conducting surveys with conventional aircraft.

Application of capturing aerial images of wildlife in the 1990s through the early 2000s primarily involved modification of recreational remote control aircraft (Thome and Thome 2000, Abd-Elgrahman et al. 2005, Jones et al. 2006). As various UAS platforms became available from commercial sources, classifications and availability of these vehicles as well as considerations for particular use also expanded (Watts et al. 2012).

Surveillance of wildlife species using UAS technology is becoming more widespread. There are reports of using various UAS platforms to survey wading birds (Abd-Elrahman et al. 2005, Jones et al. 2006), black-headed gulls (*Chroicocephalus ridibundus*) (Sarda-Palomera et al. 2011), Canada geese (*Branta canadensis*) and Snow geese (*Chen caerulescens*) (Chabot and Bird 2012), and assessing bird risk hazards in power lines (Mulero-Pazmany et al. 2013).

Use of unmanned aircraft to survey marine mammals has been considered successful (Koski et al. 2009, Hodgson et al. 2013). Unmanned aircraft systems have also been used for detection of Roe deer (*Capreolus capreolus*) (Israel 2011), and monitoring disease transmission in Red deer (*Cervus elaphus*) and Fallow deer (*Dama dama*) (Barasona et al. 2014). Vermeulen et al. (2013) examined the use of UAS to survey populations of African elephants (*Loxodonta africana*).

While the use of UAS platforms for wildlife surveillance is evident, the influence on animal behavior while being subjected to the presence of the vehicles is unclear. Vermeulen et al. (2013) reported no observable reaction by African elephants when the UAS utilized for survey purposes was maintained at an altitude of 100 m. Various wetland bird species reacted more to vertical approaches from a UAS compared to approaches at other angles (Vas et al. 2015). While the use of remote control UAS platforms with photographic instrumentation has the potential to be a useful tool for various aspects of wildlife management, if the presence of the vehicle significantly alters normal behavior, the use of these devices may be limited. Therefore, the objective of this study was to evaluate behavioral influence of white-tailed deer (*Odocoileus virginianus*) in an urban/suburban environment when repeatedly exposed to a commercially available UAS.

STUDY AREA

We conducted our study on the 1,215 ha Berry College Wildlife Refuge (BCWR) within the 11,340 ha Berry College campus in northwestern Georgia, USA. The BCWR was within the Ridge and Valley physiographic province with elevations ranging from 172 m to 518 m (Hodler and Schretter 1986). The BCWR was characterized by campus-related buildings and facilities for the 2,100 student body, interspersed with expansive lawns, hay fields, pastures, woodlots, and larger forested tracts. Forested areas were dominated by pines (*Pinus* spp.), oaks (*Quercus* spp.) and hickories (*Carya* spp.). The two test areas used for this study were characterized as a transition zone from campus lawn to agricultural hayfields. Lawn areas consisting of orchard grass (*Dactylis glomerata*), fescue (*Schedonorus phoenix*), and white clover (*Trifolium repens*) extended from buildings used for housing, approximately 100 m into hayfields predominantly composed of Bermuda grass (*Cynodon dactylon*). Each hayfield immediately adjacent to the campus buildings used as test sites were approximately 8 ha (Deer Field Hall (DF)) and 13 ha (Rollins Hay Field (RF)). Unmanned aircraft system flights initiated were within 100 m of a campus building and typically within 100 m of the same location at each site for each flight.

The BCWR had a deer population estimated at 25 deer/km^2 (D. Booke, Georgia Department of Natural Resources, personal communication). Due to significant contact with humans and lack of hunting pressure, deer on the college campus are highly habituated to the presence of humans. Approaching some animals to within a 10 m distance is common.

METHODS

We used a commercially available UAS (Phantom 2 Vision, DJI North America, Los Angeles, CA, USA). This platform was classified as a small UAS quad copter, capable of vertical take-off and landing. The UAS is operated by a portable remote control unit, with a range of 300 m and a typical flight time of up to 25 min per battery charge. This UAS is reported to have the ability for ascent at 6 m/s, descent at 2 m/s and a maximum flight speed of 15 m/s. The vehicle, operated by four electric propeller driving motors, weighs 1.2 kg, including battery and a factory-included camera. The camera is capable of still photos (14-megapixels) and high definition video recording (HD 1080/p30 or 1080/60i) with a panoramic (120^o) field of view. Live video feed of the camera view, camera angle, and flight information data is displayed by use of a smart phone application that connects to the UAS via a unique WI-FI signal generated by the flight control unit (Phantom 2 Vision – Specifications. DJI North America, Los Angeles, USA. http://www.dji.com/product/phantom-2-vision/spec.). To minimize potential variation in the designated flight sequence, there was a single operator of the UAS for all flights.

Groups of deer located within the two hay fields, Deer Field Hall (DF) and Rollins Hay Field (RF) on the Berry College campus, were subjected to 1 UAS flight per day (with multiple passes per flight; see below) for 10 consecutive days, typically between 0700 hr – 1000 hr, from 8 July – 17 July 2014. Criterion for a flight to occur required at least five mature deer within the field of view of the digital camcorder used for recording behavior. A flight of the UAS was initiated at a minimum of 100 m from the group of deer that were within the operating range of the UAS (300 m), as determined by use of a range finder (Rangemaster 900, Leica Camera Inc. Allendale, NJ, USA). Climatic conditions including temperature, relative humidity and wind speed were recorded prior to each flight (Skymaster SM-28, Speedtech Instruments, Great Falls VA, USA). At the initiation of each flight, the UAS ascended vertically to an altitude of 50 m directly over the operator. Each flight consisted of two overhead passes by the UAS, between the operator to the approximate center of the group of deer at the initial height of 50 m, followed by the same number of passes at 40 m, 30 m, and 20 m altitude. The UAS then completed a vertical landing within 3 m of the operator/take-off location.

Digital camcorder (Handycam DCR-SX63, SONY Corp. of America, New York, NY, USA) video recordings at ground level for each flight were obtained for at least 5 min prior to UAS take-off and continued for at least 5 min post-landing. Twelve, 10-second periods for each phase of each flight were examined using video playback software (VLC Media Player for Windows, VideoLAN, Paris, France). Time periods for behavioral evaluation were determined by identifying specific digital recording periods, based upon audio descriptions provided by the UAS operator and recorded by the digital camcorder during each flight. These time periods were determined by the UAS operator without input or disclosure to the video reviewing personnel. Specific time stamps for designated periods to be evaluated were identified and provided as reference points to the two individuals evaluating behavior. The 12 periods within each flight evaluated included 1-min before take-off (Pre-Flight); initiated at take-off (Take-Off); when the UAS was directly overhead of the deer for each of the two overhead passes made at altitudes of 50 m, 40 m, 30 m and 20 m; during the UAS landing (Landing); and 1 min post-landing (Post-Flight). Reviewers categorized behavior as number of seconds, within the 10 sec observation period, that deer exhibited passive (no altered behavior), alert (the animals ears and face pointing toward the UAS), active (slow to moderate movement away from area), or flight (running away from area). Each deer within the field of view during each 10 sec behavioral observation period received an individual behavioral analysis. Deer entering or leaving the field of view during the prescribed 10 sec period were included by observation for the appropriate number of seconds prior to entering or after leaving the field of view to reach a total of 10 sec evaluation.

Sound recording of decibel (dB) level was obtained using a hand-held sound meter (Extech Model 407732, Extech Instruments Corp., Nashua, NH, USA). Sound intensity levels (dB) were recorded in one of the test areas (RF) approximately 14-days following collection of behavioral data. Three sound intensity levels (dB) were initially recorded during a 5 sec period, without the operation of the UAS to obtain background sound levels. Three sound intensity levels were recorded in a similar manner when the UAS was being operated at altitudes of 1 m, 10 m, 20 m, 30 m, 40 m and 50 m directly over the operator utilizing the hand-held sound meter.

A spectrum frequency profile software (Spectrum View, Oxford Wave Research Ltd., UK) operated on an iPad (Model A1395, Apple, Cupertino CA, USA) was utilized to record sound produced by the UAS. A 1 min recording was obtained using the iPad, at a distance of 50 cm from the UAS, while hovering over a concrete surface at an altitude of 1.3 m.

Animal use procedures were approved by the Berry College Institution Animal Care and Use Committee (IACUC No - 2013-14-013).

Data Analysis

The linear model for the passive behavior or alert behavior data, y_{ijklm}, is given by: $y_{ijklm} = \mu + \alpha_i + \beta_j + \tau_k + \delta_l + e_{ijklm}$ where μ denotes the overall mean, α_i denotes the effect of location *i* (*i*=Morgan, Hayfield), β_j denotes the effect of technician *j* (*j*=1, 2), τ_k denotes the effect of flight *k* (*k*=1,..,10), δ_l denotes the effect of period *l* (*l*=pre, takeoff, pass1_50m, pass2_50m, pass1_40m, pass2_40m, pass1_30m, pass2_30m, pass1_20m, pass2_20m, landing, post) and e_{ijklm} denotes the error term, assumed to be normally distributed with mean 0 and with variance-covariance matrix Λ. The variance-covariance matrix Λ is assumed the same for all subjects. Individual observations at each period interval from all data sets were treated as repeated measurements of the corresponding experimental unit. In R-project, the function *gls* (generalized least squares) within the *nlme* library (R Development Core Team 2014) was used to fit a linear model with several different structures for the correlations among measurements. The optimal covariance structure for the variance-covariance matrix was determined using Schwarz's Bayesian Criterion (Littell et al. 1997). The passive behavior and alert behavior data sets were analyzed using the first-order autoregressive covariance structure where correlations increase as the time interval decreases (Littell et at. 1997). After significant effects were identified, differences between least squares means were considered significant at 0.05 based on the Tukey adjustment Type I error rate.

Analysis of decibel intensity was conducted using one-way ANOVA analysis procedures of IBM SPSS 22.0 (SPSS 22.0 2013) and Duncan Multiple Range Analysis to determine differences among different altitudes as treatments at the 0.05 significance level.

RESULTS

There were no differences ($P \geq 0.10$) in behavioral analysis parameters observed between the two independent reviewers of the digitally recording data. The number of deer observed in digital recordings observed during each flight were similar ($P \geq 0.05$) at the DF (12.1 ± 3.9) and RF (12.8 ± 5.6) location, ranging from 5 – 23 animals per flight. However, there was an overall difference in behavioral response of white-tailed deer exposed to the UAS treatment between the two locations. Deer exposed to the UAS platform exhibited less ($P \leq 0.001$) Passive Behavior (7.45 sec ± 0.08) in DF compared to RF (7.99 sec ± 0.08) across all 10-sec observation periods and flights. Conversely, more ($P \leq 0.004$) time exhibited as Alert Behavior was observed in deer in the DF (2.41 sec ± 0.08) versus the RF (2.08 sec ± 0.08) location. The average flight time required to complete a flight were 13.53 min ± 0.59 in the DF field and 11.63 min ± 0.32 in the RF area.

The average number of seconds white-tailed deer exhibited Passive and Alert Behavior occurring with the 10 sec observation sequences, across the 12 defined periods of each flight, indicated a progressive pattern of increasing acceptability of the presence of the UAS upon repeated exposure (Table 1). During the first flight white-tailed deer exhibited the least ($P \leq 0.05$) Passive Behavior (5.65 sec ± 0.17) and the most Alert Behavior (4.18 sec ± 0.17). There was a general progression of increasing ($P \leq 0.05$) amount of time observed as Passive Behavior and a decrease in Alert Behavior as more exposure to the UAS occurred during the 10 consecutive flights. The exception to this progression occurred during the 9^{th} of the 10 flights. During this flight, Passive Behavior and Alert Behavior was characterized as being more similar to flights 1-2 as compared to later flights. Temperature (22.19 C ± 0.42), humidity (60.0% RH ± 3.40) and wind velocity (0.80 m/s ± 0.60) were relatively consistent across most treatment days. However, during the morning of the 9^{th} flight, temperature dropped to 18.33 C with wind velocity gusting to 7.6 m/s as an impending thunderstorm approached. This storm resulted in 9.4 mm^3 precipitation. It is likely that the impending weather condition had significant impact on the deer behavior as opposed to the presence of the UAS.

Table 1. Mean time (sec) white-tailed deer exhibited Passive and Alert Behavior during the 10-sec observation time frames recorded during the 12 predefined distinct periods within each UAS flight.

Flight	Mean Passive Behavior ± SE	Mean Alert Behavior ± SE
1	5.65 ± 0.17^{a}	4.18 ± 0.17^{a}
2	7.00 ± 0.18^{b}	2.54 ± 0.18^{c}
3	7.69 ± 0.20^{c}	2.28 ± 0.19^{c}
4	7.47 ± 0.16^{c}	2.46 ± 0.16^{c}
5	8.33 ± 0.16^{d}	1.64 ± 0.16^{d}
6	8.28 ± 0.20^{d}	1.74 ± 0.20^{d}
7	8.77 ± 0.15^{e}	1.25 ± 0.15^{e}
8	8.56 ± 0.23^{de}	1.82 ± 0.22^{d}
9	6.64 ± 0.19^{b}	3.34 ± 0.19^{b}
10	8.83 ± 0.19^{e}	1.21 ± 0.19^{e}

Mean ± SE within same column with different superscripts differ ($P \leq 0.05$)

White-tailed deer exhibited a consistent pattern of Passive and Alert Behavior during the 10 sec observation time frames, within the 12 predefined flight periods, occurring during the 10 consecutive flights (Table 2). As expected, deer exhibited the most Passive and least Alert Behavior during the pre-flight period, prior to initiation of a flight. Deer on the campus are habituated to the presence of humans. Filming and preparation of each UAS flight, at a minimum distance of 100 m from the animals, induced virtually no visible response.

Table 2. Mean time (sec) white-tailed deer exhibited passive and alter behavior during the 10-sec observation time frames recorded during the 12 predefined distinct periods across all UAS flights.

Flight Period	Mean Passive Behavior ± SE	Mean Alert Behavior ± SE
Pre-Flight	9.57 ± 0.17^{a}	0.47 ± 0.17^{a}
Take-Off	7.37 ± 0.17^{e}	2.62 ± 0.17^{ef}
1st Pass 50 m	6.65 ± 0.18^{f}	3.42 ± 0.17^{g}
2nd Pass 50 m	7.16 ± 0.17^{e}	2.74 ± 0.16^{f}
1st Pass 40 m	7.14 ± 0.17^{e}	2.91 ± 0.17^{f}
2nd Pass 40 m	7.79 ± 0.18^{cd}	2.19 ± 0.17^{cd}
1st Pass 30 m	7.48 ± 0.18^{de}	2.34 ± 0.18^{de}
2nd Pass 30 m	8.27 ± 0.17^{b}	1.72 ± 0.17^{b}
1st Pass 20 m	7.35 ± 0.18^{e}	2.58 ± 0.19^{d}
2nd Pass 20 m	7.83 ± 0.19^{c}	2.08 ± 0.19^{bc}
Landing	8.01 ± 0.19^{bc}	1.97 ± 0.19^{bc}
Post-Flight	8.07 ± 0.18^{bc}	1.93 ± 0.18^{bc}

Mean ± SE within same column with different superscripts differ ($P \leq 0.05$)

Take-off of the UAS decreased ($P \leq 0.05$) Passive Behavior and increased ($P \leq 0.05$) Alert Behavior compared to the pre-flight period. Typically, the take-off and filming location was between 100 m – 150 m away from the deer. However, it was during the initial pass at 50 m altitude, culminating when directly overhead of the animals, that elicited the greatest decrease in Passive Behavior and increase Alert Behavior ($P \leq 0.05$) compared to the pre-flight activity.

Sound intensity in decibels (dB) indicated that the amplitude produced by the UAS from altitudes of 1 m to 50 m directly overhead was greater ($P \leq 0.05$) than background noise levels

(Table 3). The sound spectrum frequency profile obtained while the UAS was hovering at a height of 1.3 m produced predominant peaks ranging from 200 Hz – 4,000 Hz. In addition to the behavioral observation of deer suggesting auditory response, these frequencies (Hz) and intensities (dB) are within the range of hearing reported for white-tailed deer (D'Angelo et al. 2007). It should be noted that during any form of rapid acceleration, in any direction, there is a distinct increase in frequency (Hz) and intensity (dB) of sound produced by the UAS.

Table 3. Mean intensity of sound (dB) produced by the UAS operated at different altitudes (m).

Altitude	Mean Decibel Level (dB) ± SE
1	73.10 ± 1.50^{a}
10	58.13 ± 1.34^{b}
20	54.17 ± 0.93^{c}
30	50.43 ± 0.73^{d}
40	52.70 ± 0.50^{c}
50	48.70 ± 0.23^{e}
Background Level	44.87 ± 0.92^{f}

Mean ± SE within same column with different superscripts differ ($P \leq 0.05$)

DISCUSSION

The flight protocol utilized in this study was intended to provide a progressively increasing source of stimulus and exposure by decreasing the altitude of the UAS during the two-pass process from 50 m to 20 m, in 10 m increments. Because of the presence of power poles and transmission lines reaching a maximum height of 11 m in the RF area, it was not considered safe to fly at an altitude below 20 m. Regardless, it was during the initial pass at 50 m altitude, culminating when directly overhead of the animals, that elicited the greatest decrease in Passive Behavior and increase Alert Behavior compared to the pre-flight activity. This response is likely due to the initial approach of the UAS toward the deer creating a brief period of threat assessment. Subsequent passes resulted in a consistent trend of increasing Passive Behavior with the corresponding decrease in Alert Behavior. This suggests deer did not consider the UAS a substantial threat after initial exposure even though altitude during subsequent passes continued to decrease from 50 m to 40 m, 30 m and finally 20 m, before landing. Based upon the behavioral responses elicited by white-tailed deer when subjected to the flight protocol, habituation to the presence of an UAS appeared to be evident over the 10 day treatment period.

Research utilizing UAS platforms to quantify animal abundance continues to expand. However, behavioral influence as a result of the presence of the UAS in operation is only beginning to emerge. Various wetland bird species exhibited minimal reactions when approached by different colored UAS platforms from an initial altitude of 30 m, when approach angles were from 20^{o} – 60^{o} (Vas et al. 2015). However, birds reacted more to the UAS when a vertical approach (90^{o}) was initiated. Vermeulen et al. (2013) reported no observable reaction in elephants was recorded when a UAS was operated at 100 m altitude. However, no information of the potential amplitude or frequency of sound from the UAS was presented. Additionally, it was reported that medium and small mammals could not be observed at that height (100 m). Thus, utility of the UAS-camera combination used as the height of 100 m was effective to count elephants, but yielded little other information. The UAS-camera combination used in our study has a relatively wide field of view (120^{o}) that is useful for panoramic viewing of the environment and providing ease of orientation since environmental landscapes are clearly visible. However, this camera configuration might limit visual information of a target individual without flying the UAS in close proximity, which in turn could alter the animals' behavior. Conversely, utilization of a camera with higher focal power tends to decrease the field of view, potentially resulting in difficulty finding specific target animals or identifying environmental features and locations.

There are a number of potential applications of the UAS for wildlife related issues. However, significant consideration in selection of the type of UAS and camera configuration must be considered to be effective for any given objective. It should also be recognized that the UAS may not be an ideal tool or necessarily more effective than other options. Vermeulen et al. (2013) reported that while the UAS was effective and accurate for counting elephants, it cost approximately 10x more to operate compared to conventional aircraft due to limited amounts of land that could be observed over a given period of time. A study comparing the use of images produced by a UAS to conventional ground counts of flocks of geese produced varying results. The number of Canada geese was lower based on UAS information compared to humans counting from the ground. However, counts of snow geese by UAS images were 60% higher compared to ground counts (Chabot and Bird 2012). It was suggested that contrast in feather color between the birds and the environment contributed to the different results. The proliferation of commercial and private operation of UAS vehicles may enhance human-wildlife conflicts by increasing collisions with birds as airspace becomes more crowded (Lambertucci et al. 2015).

White-tailed deer observed in the current study were habituated to the presence of humans on the college campus. Deer under other conditions, particularly those receiving hunting pressure by humans, may not habituate as readily. Currently, there are also significant challenges related with operation of UAS as the Federal Aviation Administration (FAA) continues to develop regulatory policies for recreational, research and commercial applications. With careful consideration of research objectives, environmental and regulatory limitations, the UAS will likely continue to evolve and provide another tool for wildlife related objectives.

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An Efficient Method of Capture and Field Euthanasia of Flightless Mute Swans

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ABSTRACT: Concerns surrounding the ecological impacts from increasing numbers of feral mute swans (*Cygnus olor*) have led some management agencies in the United States to implement control efforts directed at reducing populations of this invasive species. To remove large numbers of flightless mute swans from the Maryland portion of Chesapeake Bay, we developed a field live-capture technique using a modified design of the British swan pole. During the summers of 2005–2008, we captured and euthanized 1,396 mute swans from molting flocks in 24 operations. Swans culled per operation ranged from 6 to 199 with an average cull rate of 32 swans per hour. Our capture method frequently resulted in removal of all flightless mute swans in the area. Cost was $40,259 for the 24 field operations. Mean cost per swan culled (including disposal) was $28.84. We also describe an effective, humane method of field euthanasia for large birds, such as mute swans, using mechanical cervical dislocation with an emasculatome. We used these methods as part of an integrated control program that also included egg oiling to reduce swan recruitment and the humane shooting of adult swans (2002–2014) that resulted in a reduction of the State's mute swan population from 3,995 in 1999 to 41 in 2014. These techniques will benefit other state and provincial wildlife agencies in North America that are undertaking or considering implementation of mute swan control programs.

Key Words: capture, cull, *Cygnus olor*, emasculatome, field euthanasia, mute swan, swan pole.

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INTRODUCTION

Populations of local breeding mute swans (*Cygnus olor*) are widespread and increasing in certain regions of the United States and southern Ontario (Ciaranca et al. 1997, Petrie and Francis 2003, Baldassarre 2014). As these populations have grown, so have concerns about their ecological impact on native bird populations and their habitats. Maryland's feral mute swan population originated from the escape of five captive birds in 1962 (Reese 1975). The population grew slowly through the 1960s and 1970s but then underwent rapid growth from 264 swans in 1986 to 3,955 in 1999 (Hindman and Harvey 2004). In Chesapeake Bay, mute swans have caused the abandonment of nesting areas by State-threatened waterbirds like the least tern (*Sternula antillarum*) and black skimmer (*Rynchops niger*) (Therres and Brinker 2004). Large flocks of nonbreeding swans have also reduced submerged aquatic vegetation (SAV) at the local level (Tatu et al. 2007).

The growth in mute swan numbers has also increased conflicts between people and swans, particularly swans defending their nest territory and young. Examples of conflicts with territorial swans include threat displays and direct attacks toward swimmers and people in small watercraft. The aggressive behavior of breeding swans can prevent people from using riparian shorelines (Hindman and Harvey 2004). Although no serious injuries to people have been reported in Maryland, there have been two recorded drownings caused by mute swans elsewhere (Indiana and Illinois) in the U.S. (Williams 1997, Golab 2012, Steckling 2012).

Because mute swans are considered invasive species by state and federal wildlife management agencies, some limited population control efforts have been aimed at slowing population growth (Ciaranca et al. 1997, Atlantic Flyway Council 2003). In 2003, the Maryland Department of Natural Resources (MDNR) adopted a mute swan management plan aimed at reducing the State's mute swan population to protect critical Chesapeake Bay living resources (e.g., native waterfowl, colonial waterbirds, and SAV). However, population control actions were delayed by negotiations with the Human Society of the United States and legal challenges from animal rights organizations (Tatu 2006). In 2004, the U.S. Congress provided clarification of the intent of the Migratory Bird Treaty Act (MBTA) by passage of the Migratory Bird Treaty Reform Act 2005 (Tatu 2006). The Reform Act stipulated that the MBTA only applies to migratory bird species that are native to the U.S. Congress also directed the U.S. Fish and Wildlife Service to prepare a list of nonnative species to which the act does not apply. The list was finalized on 15 March 2005 and mute swans were included, thereby returning management authority to the states. Thus, in July 2005, the MDNR initiated an integrated control strategy aimed at eliminating all mute swans from areas designated as "swan free areas" (e.g., colonial waterbird and black duck nesting habitats, SAV beds) and initially reducing the State's mute swan population to <500 by 2008 (MDNR 2003). The strategy used a combination of nest and egg destruction (Hindman et al. 2014) and the culling of adult swans using shooting and live capture with euthanasia. In 2011, the MDNR revised its mute swan management plan to include a population objective of reducing the swan population to as few as possible (MDNR 2011).

Because mute swans molt all their flight feathers simultaneously and are flightless for 4–7 weeks, they can be captured during the annual mid-summer molt (Ciaranca et al. 1997). In Britain, family groups of wild mute swans have been captured for centuries during a ceremonial activity known as swan-upping (Birkhead and Perrins 1986); swans are surrounded with several small boats or herded or driven towards shore and are captured either by hand, landing net, catch pole, or herded into temporary pens erected near the water's edge (Scott 1972, Birkhead and Perrins 1986). One of the largest single captures of mute swans occurred in 2011, when about 750 mute swans were captured for banding in The Fleet Lagoon near Abbotsbury, England, using about 90 canoeists and >150 people to form a human net to herd swans into onshore capture pens (The Independent 2011). Mute swans have also been captured in Britain for ringing (banding) studies by baiting them and then catching them by hand or with a capture pole known as a swan pole (Minton 1968, North West Swan Study 2007).

In the U.S., mute swan capture has been limited to small numbers of birds for marking studies (Reese 1975, Sousa 2005, New York Department of Environmental Conservation 2013), nuisance or escaped individuals, and removing birds to aid in reestablishing trumpeter swan (*Cygnus buccinator*) populations (Ciaranca et al. 1997). In the U.S., flightless mute swans are normally captured by pursuing them with a boat and capturing them with a large fish-landing net (Gelston and Wood 1972, Sousa 2005). In 1995, we attempted (unsuccessfully) to capture 150–200 flightless mute swans by herding them with boats towards shore and into onshore capture pens. This method has been used to capture large numbers of flightless Canada geese (*Branta canadensis*) for banding studies (Costanzo et al. 1995). However, the escape behavior of flightless mute swans differs from geese in that swan flocks do not remain intact when being herded by 3–4 small boats. Rather, they avoid capture by dispersing as individuals or as small groups (3–10 birds).

Herein we describe an efficient capture technique using a modification of the British swan pole (Minton 1968) that was used in the large-scale control of mute swans in the Maryland portion of Chesapeake Bay. We also describe a rapid, effective, and humane field method of euthanasia for mute swans.

STUDY AREA

We conducted this work in the tidal estuarine waters of the Potomac River in St. Mary's County (centered at 38°12'09"N, 76°35'55"W) and along the Eastern Shore of Chesapeake Bay in Kent, Queen Annes, Talbot, Dorchester, and Somerset counties, Maryland (between 38° 55' 17"N, 76°15'11"W and 37°57' 16"N, 76°02'50"W). These areas supported concentrations (e.g., 25 – 250 birds per flock) of flightless, nonbreeding mute swans and smaller numbers of failed breeding pairs. These portions of the Potomac River and Chesapeake Bay contained an interspersion of SAV beds, open water, tidal estuarine wetlands, and irregular shorelines.

METHODS

Molting swans in Chesapeake Bay congregated in large tidal creeks and bays or narrow (1.5–2.4-m wide) tidal creeks lined with high tide bush (*Iva frutescens*) and Phragmites (*Phragmites sp*.). Molting sites typically had abundant SAV nearby and shallow waters that limited boat traffic. We observed as many as 75–200 swans hiding within the cover provided by these creeks.

Aerial surveys using fixed-wing aircraft were used to locate 10 swan molting sites along the Eastern Shore and 1 site in the lower Potomac River. We used live capture and euthanasia to remove molting swans at 6 of the 11 molting sites where culling by shooting using 12-gauge shotguns was inappropriate because of the proximity to waterfront residential homes.

We began capture operations between 1000 to 1300 hours when boating activity was lowest and about 1–2 hours prior to high tide to ensure adequate water for capture boat maneuverability. It was difficult to operate small boats powered with conventional outboard motors where swans congregated in shallow waters and creeks. We used a 4.2-m jon boat powered by a long-tail mud motor (Mud Buddy ®, West Jordan, UT) to drive flightless swans from the protective cover of these creeks. Once in the open, swans were slowly herded by 2–3 additional capture teams in jon boats to deeper offshore waters (1.2–3.7 m) where they were easier to capture and where the operation was less visible from waterfront homes.

Once swans were positioned offshore, we captured individuals with a swan pole after pursuit by boat. The swan pole was a modified aluminum, telescopic pole (approximately 2.4 m fully extended) that had a smooth, rounded hook or shepherd's crook at one end (Figure 1). The pole's crook was placed quickly around a swan's neck so that the bird could be pulled toward the person making the capture. We captured most swans on the first attempt, but some required 2–3 capture attempts. A handler lifted each swan into the boat and restrained the bird on the boat floor below the gunwale where it was immediately euthanized by mechanical cervical dislocation) and the carcass placed in a plastic bag for transport and disposal.

We recorded staff hours, vehicle and boat costs, equipment purchases, and miscellaneous expenses for each of the live-capture culling operations. The duration of each culling operation was also recorded and began when capture teams arrived at a capture location and ended when each capture team had transferred bagged carcasses to onshore trucks for transport to disposal locations and began their return to nearby boat launch ramps. We determined the mean number of swans culled per hour of an operation by dividing the total number of swans culled by the number of field operation hours required to complete the 24 culling operations (for example, 1,396 swans/44-culling hours = 31.88 swans culled per hour).

Figure 1. Distal end of telescopic, aluminum swan pole (3.2-cm crook gap) made of marine- grade aluminum rod (0.6-cm) used to capture flightless mute swans in the lower Potomac River and upper Chesapeake Bay, Maryland, USA, 2005–2008.

Field Euthanasia

Cervical dislocation can be applied manually, which involves stretching and separating the vertebrae by hand, or mechanically, which involves the use of a tool such as bovine castration forceps (emasculatome) to sever or crush the vertebrae (Galvin et al. 2005). For mute swans we used a 48-cm emasculatome (Jeffers, Dothan, AL) to mechanically perform the cervical dislocation. Mechanical cervical dislocation using this tool has been recommended as a field method of euthanasia and farm culling for large birds (U.S. Department of Interior and U.S. Geological Survey 1999, Canadian Council on Animal Care 2009). We used the American Veterinary Medical Association (AVMA) guidelines for the euthanasia of wildlife and consulted with veterinarians to ensure that the field techniques used for culling swans was humane (AVMA 2000).

We used mechanical cervical dislocation to humanely euthanize all captured mute swans. Each member of our capture teams received training in the proper use of the emasculatome to perform the cervical dislocation. We restrained each captured swan by laying the bird on its sternum with its neck outstretched on the boat floor while holding the base of the wings next to the body. We found mechanical cervical dislocation could be performed rapidly and humanely by placing the open emasculatome forceps about 3-cm below the base of the skull and clamping the forceps tips shut firmly for 2–5 seconds. Following luxation of the cervical vertebrae and coincident severing of the spinal cord, and cessation of reflex muscle spasms, we immediately placed each swan carcass in a plastic 3-mil 182–227 liter contractor bag. The entire process from time of capture until a single bird was humanely killed and then stored for transport averaged about 30 seconds.

Swan Pole Construction

We constructed swan poles patterned from the Abbotsbury Swannery in the Britain (Birkhead and Perrins 1987). To construct our swan poles we modified a 1.47- to 2.43-m telescopic aluminum boat hook (West Marine, Watsonville, CA) by removing the hook portion of the tool and welding a 1.5-cm diameter, marine-grade, aircraft aluminum rod to the distal end of the pole. The aluminum rod was heated and bent into the shape of a hook or shepherd's crook (Fig. 1). The rod extended 43.2 cm from the end of the pole and was bent and extended 27.3 cm in the opposite direction and parallel to the portion of the rod extended from pole. The inside dimension of the gap between the rods that formed the crook was 5.1 cm.

In the spring and summer of 2002 and 2003, we tested the swan pole design in capturing and marking about 100 mute swans including incubating swans, adult swans with cygnets that were either flightless or reluctant to fly, and flightless swans associated with a swan research project (see Sousa 2005). Although successful, we noted that the original swan pole design enabled some swans to escape from the pole's crook. We modified the original pole design by first bending the outward tip (8.25 cm) about 45º to help guide a swan's neck into the crook, and second, reducing the gap of the crook from the original 5.1 cm to 3.2 cm. The weight of the distal end of the swan pole was also reduced by using a smaller gauge marine-grade aluminum rod (1.27-cm diameter) to form the crook. These modifications, especially the smaller gap distance, resulted in an improved capture rate with reduced effort (i.e., fewer capture attempts).

Project Costs

For each of the 25 live-capture culling operations we recorded the manpower (person hours and salary), vehicle- and boat-use expenses, and cost of field equipment and supplies. We included the cost required for disposal (i.e., burial). However, some carcasses were incinerated at Maryland Department of Agriculture (MDA) Animal Health Diagnostic Laboratories. Incineration costs were not included in the operation costs as our swan carcasses were added to MDA's weekly incineration of commercial poultry carcasses as an integral part of their poultry health surveillance program. We used the total operation costs to calculate the mean cost required to cull an individual swan.

RESULTS

Between 11 August 2005 and 21 September 2008, we culled 1,396 flightless mute swans on public waters during 24 live-capture culling operations (Table 1). Most flightless (n = 1,020; 74%) swans were culled during the last 2 weeks in August (Fig. 2). The number of swans removed was greatest in 2005 (n = 721) when molting flocks were largest and declined each successive summer thereafter as swan population size declined (Table 1). Mean cull size was 58 swans per operation (range 6–199) for the 4-year period (2005–2008). Mean cull size per operation was also highest (120 swans) the first year (2005) and declined steadily each year thereafter (Table 1). Culling operations lasted between 1.0–3.5 hrs for all 4 years combined (44 hours total) and cull success averaged 32 swans per hour. This culling method frequently resulted in removal of all flightless mute swans in the area.

Other flightless, molting flocks of mute swans on public waters in remote locations were culled by shooting during this same 4-year period. After 2008, molting flocks of swans were rare and only flightless individual and paired swans were live-captured in subsequent years (2009–2014). Live capture was used in combination with the culling of adult swans by shooting throughout the spring, summer, and fall and egg oiling of nests during the spring (Hindman et al. 2014) to reduce the State's mute swan population.

Interactions with the public occurred during only 1 of the 24 live-capture operations. No press or media coverage resulted from any of the culling operations (live capture or shooting). Public reaction to the control of mute swans was mixed, but opposition was less than expected. Results of a random telephone survey of Maryland citizens in 2005 indicated that nearly all respondents (n = 539; 86%) would support mute swan population control after they were provided evidence that this species was harmful to the Chesapeake Bay ecosystem; they felt the health of Chesapeake Bay was more important than sustaining a non-native swan population (Hindman and Tjaden 2014). Of the respondents that supported aggressive control measures, 62% supported the use of lethal methods of control, including hunting.

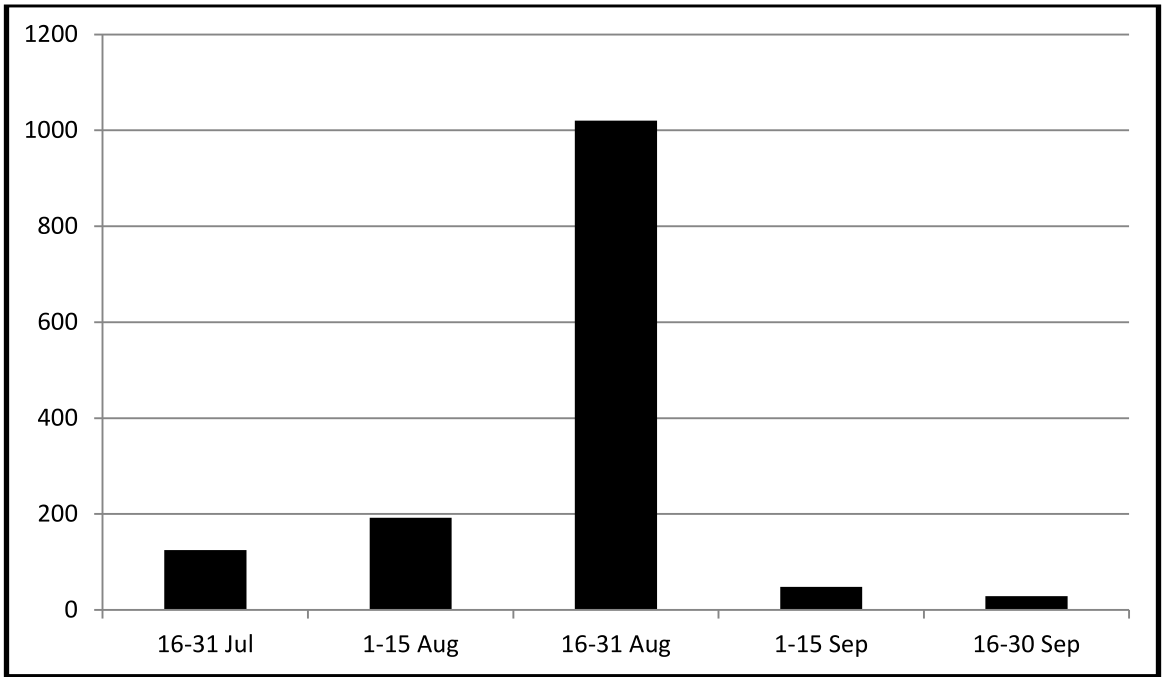

Figure 2. Temporal distribution of flightless mute swans captured during 24 live-capture operations in the lower Potomac River and upper Chesapeake Bay, Maryland, USA, 2005–2008.

Table 1. Population size and number of flightless mute swans live captured and euthanized, with number and dates of cull operations, mean and range of swans culled per operation in the lower Potomac River and upper Chesapeake Bay, Maryland, USA, 2005–2008.

Year	Population size[a]	No. swans culled	No. of operations	Mean no. swans culled per operation	Range of swans culled per operation	Cull dates
2005	3,624	721	6	120	58–199	11–30 Aug
2006	2,174	453	11	41	6–139	26 Jul–28 Sep
2007	1,455	158	5	32	9–60	8 Aug–6 Sep
2008	927	64	2	32	17–47	11 Aug–21 Sep
Total		1,396	24	58	6–199	26 Jul–28 Sep

[a] Population size from annual September survey prior to implementation of swan cull operations the following summer. Population size used for 2005 was count from 2002; no surveys were available for 2003 and 2004.

Project Costs

Total cost incurred during the 24 live-capture cull operations was $40,259.74 (Table 2). As expected, staff hours was the most expensive part of cull operations. Salaries of MDNR staff ($29,699) composed 74% of the total project costs. Operation costs declined over the 4-year period as fewer molting swans were encountered. Costs were highest the first year (2005; $25,541) when 721 birds were culled during 6 field operations, and lowest the fourth year (2008; $2,319) when only 64 birds were culled during 2 operations. Mean cost per swan culled was $28.84 for the 24 operations and ranged from $25.92 in 2006 to $36.24 in 2008.

Table 2. Estimated cost of culling flightless mute swans by live capture and euthanasia in the lower Potomac River and upper Chesapeake Bay, Maryland, USA, 2005–2008.

Year	No. culled	Staff Hours	Salaries	Vehicle costs	Boat costs	Misc. costs	Total costs	Mean cost per swan culled
2005	721	623	$15,928	$2,088	$2,467	$1,057	$21,541	$29.88
2006	453	354	$8,756	$1,367	$1,620	$229	$11,743	$25.92
2007	158	144	$3,382	$833	$320	$120	$4,655	$29.65
2008	64	60	$1,633	$341	$280	$75	$2,319	$36.24
Total	1,396	1,181	$29,699	$4,629	$4,687	$1,481	$40,259	$28.84

DISCUSSION

We captured mute swans by herding flightless birds offshore to deeper waters which increased capture effectiveness and efficiency. This technique reduced capture time by maximizing boat maneuverability, resulting in fewer attempts to catch individual swans. Capture in shallow waters compromises boat maneuverability and increases capture time unnecessarily by having to adjust outboard motor propeller position and clear the propeller fouled by SAV. Herding of flightless swans offshore for culling also minimized potential conflicts with onshore property owners. Our control method also allowed us to conduct swan control when fewer people were engaged in commercial and recreational fishing and boating. This technique allowed us to remove swans in highly developed areas where shooting would not have been appropriate.

Our method was also more efficient than the methods used in Britain where large numbers of canoeists and volunteers forming a human pen are used to herd flightless swans into onshore capture pens (The Independent 2011). Further, our method did not require us to secure property owner permission to herd swans onto a private beachhead near locations where flightless swans congregated to molt. However, in some instances we obtained landowner permission to offload bagged carcasses at a private beachhead for transport.

The use of the modified swan pole was more effective and efficient than using a fish-landing net. The swan pole was far more maneuverable than a bulky landing net. Also, it is more difficult to get a landing net around a swan's body on the water. We found that a swan captured in a landing net took longer to remove because its wings and feet often became entangled in the netting. The use of the swan pole also enabled us to capture swans without causing physical injury (e.g., broken wing).

Captured swans were killed quickly and humanely using mechanical cervical dislocation, consistent with the guidelines for euthanasia of free-ranging wildlife (AVMA 2000). Cervical dislocation humanely kills waterfowl and poultry by causing instant loss of central nervous system activity, resulting in simultaneous anesthesia and death. Cervical dislocation can be applied manually in the field and is typically used on small to medium-sized birds, such as ducks (New York Department of Environmental Conservation 2004). However, manual cervical dislocation of large birds, like mute swans, is physically difficult to conduct and may not result in a rapid and painless death (U.S. Department of Interior and U.S. Geological Survey 1999).

Mechanical cervical dislocation is sometimes recommended for the euthanasia of large birds when manual means are difficult to apply (Canadian Food Inspection Agency 2007, Saif 2008, CCAC 2009). Both manual and mechanical cervical dislocation are listed as killing methods for poultry by the World Organization for Animal Health for the purposes of disease control (Galvin et al. 2005). Cervical dislocation and blunt trauma are the methods most commonly used on commercial turkey farms and are thought to be humane (Erasmus et al. 2010). However, there is little scientific evidence to confirm this observation (AVMA 2007).

We chose to use mechanical cervical dislocation as the preferred method of field euthanasia for captured mute swans because it (1) was considered efficient and humane by consulting veterinarians given the field conditions; (2) was consistent with the guidelines for euthanasia of free-ranging wildlife (AVMA 2000); (3) minimized distress to captured swans associated with alternative methods of euthanasia; (4) was practical under field conditions (marine habitat from boats), (5) reduced worker safety risks; and (6) allowed for burial of tissues free of chemical contamination.

MANAGEMENT IMPLICATIONS

In 2011, the MDNR updated its 2003 mute swan management plan by revising the primary management objective to reducing the mute swan population to as few birds as possible (MDNR 2011). Our live capture and field euthanasia techniques were part of an integrated population control strategy aimed at reducing Maryland's mute swan population (MDNR 2003, 2011). We reduced the State's mute swan population from 3,995 in 1999 to 41 in 2014. Our work demonstrates that the use of these control methods can be used to reduce a jurisdiction's mute swan population. These techniques can be especially effective in eliminating flightless swans during the annual feather molt in areas where culling by shooting is not appropriate. Our work will benefit other state and provincial wildlife agencies in North America that are considering or undertaking the implementation of mute swan control programs.

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A Description of Lethal and Nonlethal Predator Management at Two Piping Plover (*Charadrius melodus*) Nesting Colonies in Michigan

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ABSTRACT: Predator management in the Great Lakes region of Michigan has played an important role in the recovery program of the federally endangered Great Lakes piping plover (*Charadrius melodus*). We describe 2 long-term piping plover breeding sites located on Lake Michigan with different management strategies. We review data (2003-2014) from Dimmick's Point on North Manitou Island (NMI), part of Sleeping Bear Dunes National Lakeshore, and Ludington State Park (LSP). These sites were chosen because both have had multiple breeding pairs of piping plovers during the entire period we considered, and are in the same region of Michigan, approximately 75 miles apart. The 2 sites are likely impacted by similar environmental conditions and influenced by the same predators. Predator species common to both locations include American crow (*Corvus brachyrhynchos*), common raven (*Corvus corax*), ring-billed gull (*Larus delawarensis*), herring gull (Larus argentatus) and merlin (*Falco columbarius*). On Dimmick's Point, combinations of lethal and non-lethal predator-management methods were used including shooting with shotgun, suppressed rifle and pyrotechnics. On LSP the only control measures included predator nest exclosures and plover monitoring. Dimmick's Point had a 62% fledge rate of chicks that were hatched after lethal predator management was implemented compared with only 49% of chicks fledged of those hatched at LSP during the same time period. During this time period Dimmick's Point fledged 2.07 chicks per pair compared with 1.77 chicks per pair at LSP, this despite LSP averaging 3.52 chicks hatched per pair compared with 3.28 hatched per pair at Dimmick's. These results suggest that the use of lethal predator management can be useful to increase plover fledging rates at locations where predation continues to be a limiting factor. Without effective predator management, some long-term piping plover nesting sites on the Great Lakes could experience significant losses to predation.

Key Words: American crow, common raven, *Charadrius melodus*, endangered species, herring gull, merlin, piping plover, predator management, ring-billed gull.

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INTRODUCTION

The Great Lakes population of piping plovers (*Charadrius melodus*) was listed as endangered under provisions of the U.S. Endangered Species Act in 1986. The Great Lakes population had declined from a historic size of several hundred breeding pairs to 17 at the time of listing. From 1986-2002, the population fluctuated between 12 and 51 breeding pairs, with breeding areas remaining largely confined to Michigan (USFWS 2003).

The plovers are imperiled chiefly by significant loss and degradation to the wide sandy beaches they require for nesting, beaches where they often face a wide range of predators. Predation was identified as the cause of nest failure of approximately 14.5% of clutches in Michigan from 1981 to 1999 (Wemmer 2000), and predators are an important source of mortality for piping plover chicks (Roche et al. 2008). The Recovery Plan for the Great Lakes piping plover identifies predator management as a high priority (USFWS 2003).

In 2003, a pilot project of predator management was initiated by Wildlife Services at the plover nesting colony at Dimmick's Point (Dimmick's) on North Manitou Island (NMI) in Sleeping Bear Dunes National Lakeshore, which appeared to achieve considerable success (Struthers and Ryan 2005). That project has continued at varying levels through 2014.

In this paper, we review the results of 12 years of that effort and compare those results with a similar plover colony 75 miles to the south at Ludington State Park (LSP) in Michigan (Figure 1). Both sites have had multiple breeding pairs during the period of 2003-2014. Additionally, these locations are influenced by similar weather conditions and have similar types of predators including American crow (*Corvus brachyrhynchos*), common raven (*Corvus corax*), ring-billed gull (*Larus delawarensis*), herring gull (*Larus argentatus*) and merlin (*Falco columbarius*). During this period a combination of lethal and nonlethal predator management was utilized on Dimmick's, while at LSP only nonlethal management was used. The predator management team responsible for making program decisions included biologists from the National Park Service (NPS), Michigan Department of Natural Resources (DNR), U.S. Fish and Wildlife Service (USFWS), and USDA – Wildlife Services (WS).

STUDY AREA

Dimmick's Point – This is one of the most important nesting locations for Great Lakes piping plovers. It contains 109 acres (44 hectares) and 2.1 miles (3.3 km) of designated piping plover critical habitat shoreline. Located on the southeastern end of NMI (Fig. 1), it is approximately 9.9 miles (16 km) from the mainland and is managed as a wilderness island, allowing foot travel only.

Conducting predator management on NMI required a considerable logistical effort. Wildlife Services employees were stationed in Gaylord, MI approximately 90 miles (145 km) from the NPS shuttle boat that is used to transport employees and gear to Dimmick's. There are no facilities such as shelter or potable water available at Dimmick's, requiring employees to move food, water, tents and other equipment for periods ≤ 5 days (Table 1).

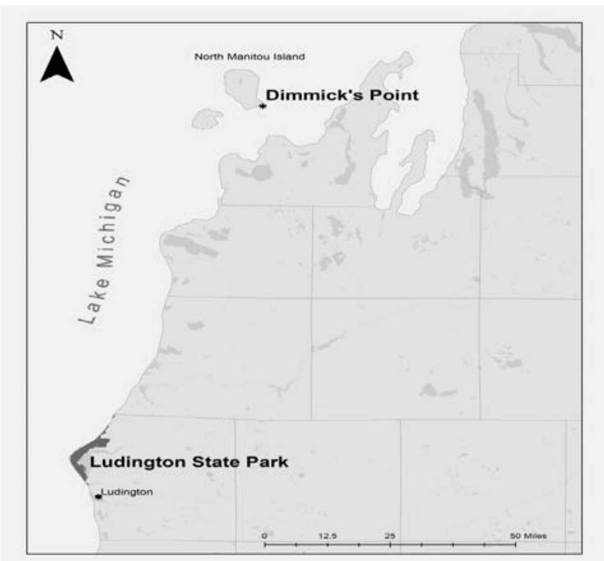
Ludington State Park – This site is approximately 5,300 acres (2,144 hectares) in size and consists of a vast dune complex situated between Lake Michigan and the inland Lake Hamlin (Fig. 1). Piping plovers at LSP typically concentrate nests in 2 areas, from near the Big Sable Point Lighthouse north to the northern boundaries of the park, and an area just north and south of the Hamlin Lake outlet. Piping plovers nest near the beaches and fore dunes but also in extensive cobble pans located in the back dunes farther from Lake Michigan.

METHODS

Dimmick's Point

Crows, ravens, and merlins were removed using a shotgun or suppressed rifle. In some cases, crows were lured within shooting range with an electronic call. Gulls were dispersed using pyrotechnics reinforced by shotgun shooting. These methods were evaluated and no disruption or disturbance effect on nesting or foraging plovers was observed (Struthers and Ryan 2005). Fenced exclosures were installed around all plover nests shortly after nesting activity began.

Figure 1. Relative locations of Dimmick's Point and Ludington State Park.



Funding limitations did not allow for WS employees to be on-site continuously. The timing and duration of visits was a joint decision between NPS and WS personnel. The first visit was generally scheduled in advance and aimed to coincide with the anticipated peak of hatching. To the degree that funding allowed, additional visits were requested by NPS when predator threats became significant. Typically, 2 or 3 employees were deployed together and worked from dawn to dusk, e.g. 0600 hours to 2100 hours.

Phases of the management plan. - As predator abundance and behavior changed, predator management activities had to adapt to be effective. This was of critical importance. The implementation of predator management was also influenced by the funding provided by either NPS or FWS. The evolution of the management plan as collectively decided by the management team was divided into a succession of 7 phases.

Phase I: Before lethal predator management (1993 to 2002). - During this time period the only predator management activities were nest exclosures to protect the nesting plovers and eggs.

Phase II: Getting started (2003– 2006). - During this phase, crows and gulls were the primary predators. There was sufficient funding to conduct multiple trips each season. Crows were removed by shooting, but crows became increasingly wary, requiring adaptations such as electronic crow calls, owl effigies, crow decoys, blinds, and various stalking techniques. Once gulls were dispersed, plovers moved into the unoccupied habitat and nested in areas not observed during the previous 10 years.

Phase III: Complications (2007 – 2008).-.During both years funding was limited, which restricted management activities to only 1 trip per year. This resulted in reduced survivorship of adults and chicks. Crow and gull management activities were implemented and merlins started to visit regularly. In 2008, 4 plover nests/adults were lost to suspected merlin predation (SBDNL 2008).

Phase IV: Restricted merlin management (2009 – 2010). - Funding was restored to allow multiple trips, providing adequate plover chick protection from crows and gulls. However, 5 plover nests/adults were lost to suspected merlin predation (SBDNL 2009, SBDNL 2010). Because merlins were a state-listed threatened species, the management team limited the lethal take of merlins. For example, there were occasions that the management team would only allow the take of 1 or 2 merlins per trip, even though additional merlins were actively hunting in the plover habitat.

Phase V: Tipping point (2011). - Funding was available for multiple trips, providing adequate plover chick protection from crows and gulls. Early in season, before plovers started nesting, NPS monitors found leg bands of an adult plover in a raptor pellet under a popular merlin perch tree. This discovery suggested that merlins might focus on adult plovers and prompted an early emergency trip to conduct merlin management. The perception of a high level of merlin activity put tremendous strain on the decision-making process of how many merlins should be removed. Two plover nests/adults were lost to suspected merlin predation (SBDNL 2011).

Phase VI: Predators are relentless (2012). - Funding was available for 5 trips providing adequate chick protection from gulls and crows. This was the first year the predator management team agreed that all merlins using the plover breeding area should be removed. Four of the 5 trips occurred by request in reaction to multiple merlin sightings, requiring constant redeployment. Ten plover chicks were predated during a 5 day period, and multiple crows and merlins were reported using the plover breeding area. One plover nest/adult was lost to suspected merlin predation (SBDNL 2012).

Phase VII: Intensive program (2013 – 2014). - This began a new era in proactive predator management. This was the first year the predator management team reorganized the structure of predator management to include 3 planned trips. In the past, one predetermined trip was planned during mean plover hatch dates, and redeployment occurred only after predation occurred. This new strategy, coupled with effectual merlin management, resulted in an increase in plover abundance; 9 pairs in 2013 and 10 pairs in 2014. In both years, no plover adults/nests were lost to suspected merlin predation (SBDNL 2013, SBDNL 2014).

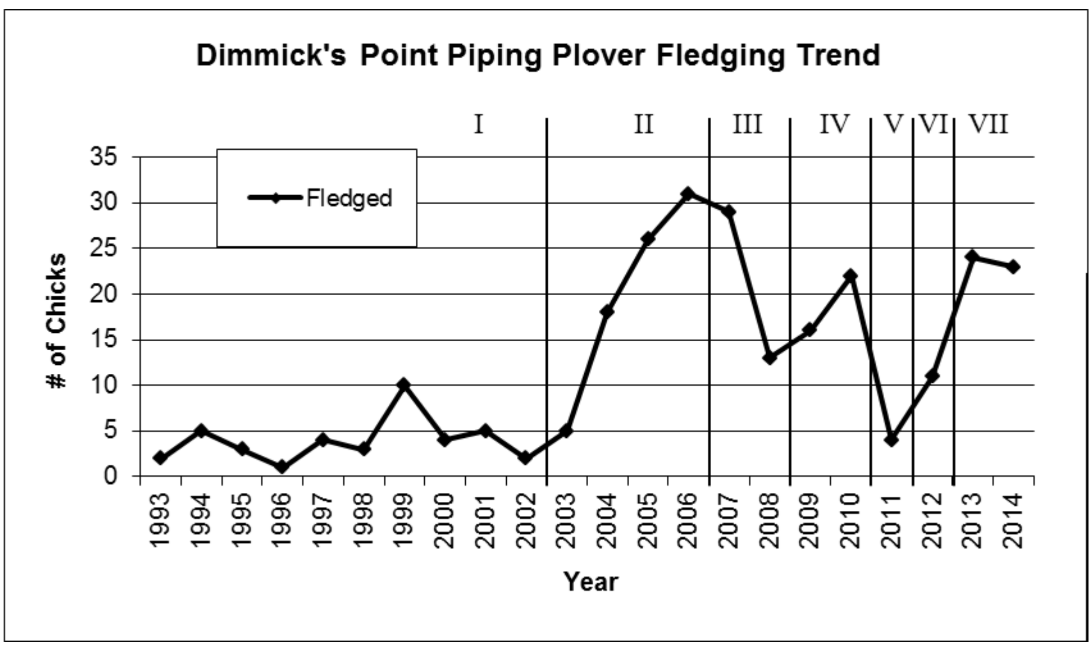


Figure 2. Piping plover chick fledging success at Dimmick's Point, Sleeping Bear Dunes National Lakeshore. Periods marked I through VII refer to different phases of predator management (see main text).

Ludington State Park

The only predator management activities implemented were nest exclosures. A nest exclosure is a welded wire fence that completely encloses the nesting plovers. It is buried 4 to 6 inches into the beach-sand to prevent access to egg predators, and has netting over the top to prevent predation from avian predators. The adult plovers gain access by walking through the spaces between the welded wires. The exclosure is very effective at protecting the nesting plovers and eggs (Larson et al, 2002, Stringham and Robinson 2015). However, piping plover chicks are precocial and leave the nest 4 hours after hatching. They remain flightless for approximately 27 days, which makes them very vulnerable to predation (Roche et al. 2008).

At LSP, plover monitors are responsible for locating individual plovers along a 6-mile section of Lake Michigan shoreline. Plover monitors were also responsible for documenting predators and predator tracks observed during their daily duties. Each location was only observed a few hours a day, making the task of witnessing a predation event very difficult.

Predators observed at LSP worth noting were crow, raven, ring-billed gull, herring gull and merlin. These are the same species found at Dimmick's Point.

RESULTS

Dimmick's Point

Predator abundance and behavior varied somewhat over the years and, consequently, so have predator management and results (Tables 1, 2). In the early years, crow and gull numbers were noticeably higher and required the majority of the management activities. Eventually, the crow and gull numbers were reduced, requiring less management effort. Conversely, the merlin numbers rose steadily and became the primary management concern.

Between 2003 through 2006, gull harassment played an important role in protecting plover chicks. At that time, it was common to see approximately 2,000 gulls using the plover habitat. Once the gulls were dispersed, the plovers nested in areas not observed by NSP monitors in the previous 10 years before management. Between 2007 through 2014 gull numbers declined and the harassment efforts were conducted with less frequency.

During the first 3 years of the project, crows were abundant, vocal, and predictable. Early on, the local crow population was high with multiple nesting pairs within a mile of the plover colony. Each crow nest had multiple subadults assisting with chick rearing duties. However, by the end of the 2005 season we noticed a significant change in crow behavior. The local surviving crow population was reduced, more wary, less vocal, and non-responsive to the electronic call.

Before predator management (1993-2002) the percentage of hatched plovers that fledged averaged 45%. When active predator management was applied from 2003-2014, the average number of hatched plovers that fledged rose to 62%.

Table 1. Summary of predator management effort (# of trips, # of days) by USDA-WS and predator management results to protect piping plovers at Dimmick's Point, Sleeping Bear Dunes National Lakeshore, MI. (AMCR = American crow, CORA= common raven, RBGU = ring-billed gull, HEGU= herring gull, MERL= merlin)

Year	Trips	Days	AMCR removed	CORA removed	MERL removed	RBGL removed	HEGU removed	Gulls dispersed
2003	2	13	23	0	0	50	6	750
2004	3	14	23	7	0	60	15	1200
2005	2	10	26	0	3	75	12	900
2006	3	10	14	0	0	200	0	3650
2007	1	6	17	0	0	15	0	400
2008	1	4	0	0	1	0	0	0
2009	2	6	6	0	1	57	0	400
2010	5	8	5	3	4	0	0	0
2011	2	8	2	0	1	8	0	700
2012	5	12	8	0	9	0	0	0
2013	4	13	7	0	10	6	0	800
2014	3	9	3	2	4	0	0	0
TOTALS	33	113	134	12	33	471	33	8800

Table 2. Piping plover nesting results at North Manitou Island, Sleeping Bear Dunes National Lakeshore, MI.

Year	PIPL Pairs	Eggs Hatched	PIPL chicks fledged	PIPL chicks/ pair
2003	2	7	5	2.5
2004	7	26	18	2.57
2005	10	39	26	2.6
2006	12	40	31	2.58
2007	13	47	29	2.23
2008	12	32	13	1.08
2009	10	25	17	1.7
2010	10	27	22	2.2
2011	3	8	3	1
2012	8	28	11	1.375
2013	9	34	23	2.56
2014	10	35	25	2.5
TOTALS	106	348	223	2.07

Ludington State Park

From 2003 to 2009 nesting success varied with several years of relatively good productivity followed by unproductive years, much like results at Dimmick's prior to predator management (1993-2002) (Figure 3; Table 3). In 2013 and 2014 Ludington had extremely low productivity, possibly due to increased merlin predation.

Figure 3. Piping plover fledging success at Ludington State Park - without lethal predator management

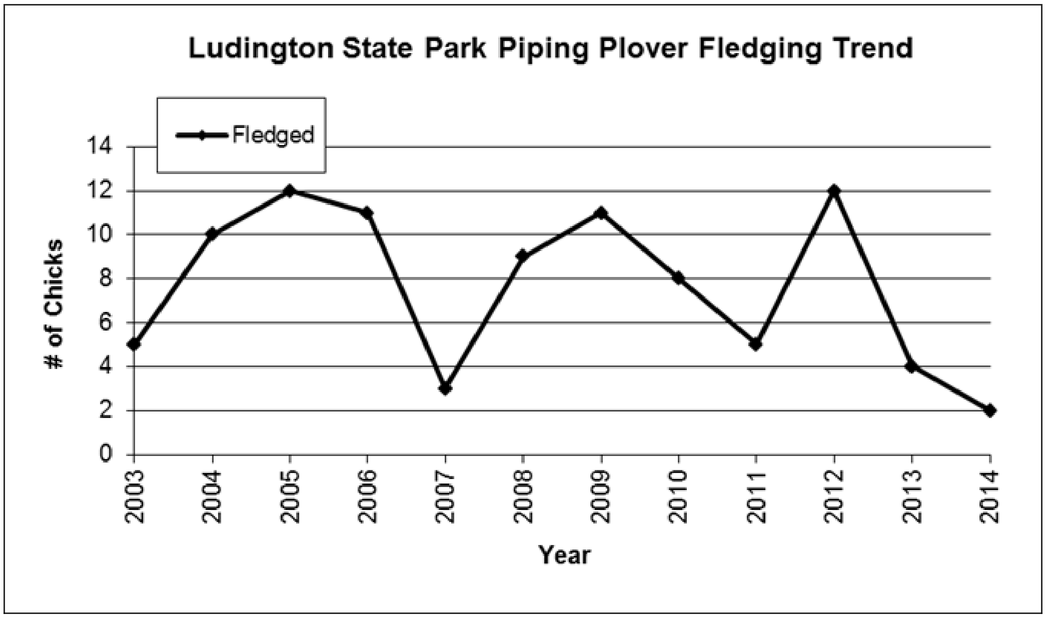


Table 3. Piping plover nesting success at Ludington State Park, MI from 2003 to 2014.

Year	PIPL Pairs	Eggs Hatched	PIPL chicks fledged	PIPL # chicks/ pair
2003	3	12	5	1.67
2004	3	11	10	3.33
2005	4	15	12	3
2006	7	25	11	1.57
2007	3	10	3	1
2008	4	15	9	2.25
2009	4	12	11	2.75
2010	5	20	8	1.6
2011	4	13	5	1.25
2012	7	26	12	1.71
2013	5	19	4	0.8
2014	6	15	2	0.33
TOTALS	55	193	92	1.77

DISCUSSION

Predators have been implicated in the decline of piping plovers and other similar beach nesting shorebirds (Ivan and Murphy 2005, Dinsmore et al. 2014). Efforts to increase plover survival have included the use of predator exclosures and lethal predator control. A modeling study by Stringham and Robinson (2015) found that a combination of using both predator exclosures and lethal predator control was the best option for increasing piping plover abundance on the Atlantic Coast. While our sample size is small, we have some evidence that this may be the case for Great Lakes piping plovers as well. Before predator management at Dimmick's (1993-2002), only 45% of chicks hatched survived to fledge. When active predator management was applied from 2003-2014, 62% of hatched chicks survived until fledging.

Although each site has somewhat different conditions and there is still a limited sample size to compare these locations, there is suggestive evidence that a combination of predator exclosures and lethal predator management at Dimmick's has been more successful than using predator exclosures alone at LSP. During this time period Dimmick's fledged 2.07 chicks per pair compared with 1.77 chicks per pair at LSP, this despite LSP averaging 3.6 chicks hatched per pair compared with 3.2 hatched per pair at Dimmick's. While monitoring and nest exclosures are key to hatching success, it may be that lethal predator management is one of the few ways to protect chicks during the vulnerable period between hatching and fledging. Dimmick's had a 62% fledge rate of chicks that were hatched after lethal predator control was instituted compared with only 49% of chicks fledged of those hatched at LSP during the same time period.

Merlins are very serious threats because they will take adult plovers and adults, especially experienced breeders that are critical to the long-term rebuilding of this population (LeDee et al. 2010). Merlins have been implicated in loss of many adult Great Lakes piping plovers (Roche et al. 2010). Lethal predator management is one of the few ways to reduce the presence of Merlins at Piping Plover breeding sites. This could lead to a long-term increase in plover abundance if this helps increase survival of breeding adult plovers.

Predators remain an important source of mortality in the Great Lakes piping plover population and present a barrier to recovery of this federally endangered population. A combination of predator exclosures and lethal predator management may be an important strategy to increase long-term plover survival and lead to population recovery.

Lessons Learned

It was valuable that experienced employees were available for predator management. Experience paid off in not only in skill with management techniques but also in understanding predator behavior and patterns. Often the predators used the same habitats and were found at the same locations. These locations often took years to identify.

The NPS stationed plover monitors at Dimmick's and their astute observations of predators were extremely useful since Wildlife Services could not be on Dimmick's continuously.

Crows/ravens become wary and elusive when exposed to management and thus required elaborate measures to be successful. The following are our observations and recommendations for a successful crow/raven management program for the long term.

1. Initially crows/ravens were easy to call and would immediately start flying towards the sounds of the electronic call.
2. Eventually crows/ravens become call-shy and had to be coaxed in by diversifying calls.
3. Do not remove the crow/raven nest or chicks until all the adults have been collected.
4. Often crows/ravens can be found in the same locations year after year.
5. It may take several days to remove the last few educated crows/ravens from the targeted group.
6. Crows/ravens begin foraging at daybreak in the plover nesting area.

Gulls can be incredibly difficult to disperse because they seem to be slow to associate danger with humans, shooting and pyrotechnics. It may require several days of repetitive harassment to alter their habits. Initial harassment efforts may take all of the daylight hours and hundreds of pyrotechnics.

Merlins may be difficult to observe and thus require diligent surveillance. Most encounters with merlins in the plover nesting area occurred at dawn or dusk but can occur at any time. Merlins have a unique call which can be useful for locating a nest.

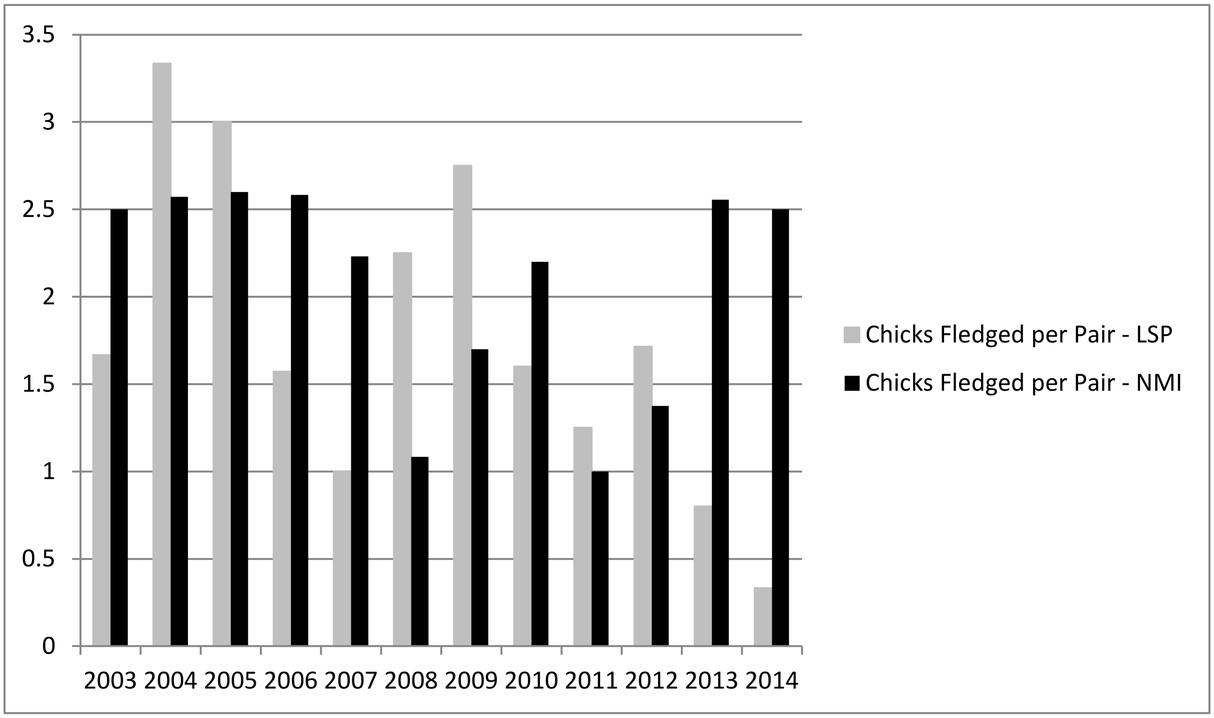


Figure 4. A comparison of piping plover chicks fledged per pair at Dimmick's and LSP. The Recovery Goal is to maintain 1.5 chicks fledged per pair each season.

MANAGEMENT IMPLICATIONS

Predation can be a serious obstacle to the recovery of piping plovers. An effort to intervene on behalf of plovers needs to take into account the sudden and relentless nature of predation. Twelve years of experience at Dimmick's which combined non-lethal measures with lethal removal appears to provide benefits for plovers. Keys to a successful strategy include the timely and prompt application of a full range of methods by skillful and experienced personnel.

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Blackbird Damage is an Important Agronomic Factor Influencing Sunflower Production

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ABSTRACT: From 2001 to 2013 (except 2004), the National Sunflower Association sponsored a comprehensive production survey of physiologically mature sunflower (*Helianthus annuus*) fields in the Canadian province of Manitoba and eight states in the United States. Trained teams of surveyors randomly stopped at one sunflower field for every 4,047 – 6,070 ha (10,000-15,000 acres). Each team evaluated plant stand, yield potential, disease, insect, weed, and bird damage for each field. We pooled data gathered during the most recent 5-years (2009 to 2013) of the survey and found that sunflower damage caused by blackbirds and plant lodging ranked fifth (behind plant spacing, disease, drought and weeds) as the most limiting factors on production. We found that overall annual economic losses from blackbird damage averaged $US13.5 million and $US4.9 million for oilseed hybrids and confectionery hybrids, respectively. We suggest elements of a multi-faceted bird management plan that might help reduce damage.

Key Words: blackbirds, crop damage, Icteridae, Integrated Pest Management, nonlethal management, Prairie Pothole Region, sunflower

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INTRODUCTION

Blackbird (Icteridae) damage is the most common reason that sunflower (*Helianthus annuus*) producers in North Dakota stop planting sunflower (Linz and Homan 1998, Linz et al. 2011, Hulke and Kleingartner 2014). Blackbirds form large flocks in late summer that feed on ripening crops, including sunflower, corn (*Zea mays*), and small grains (Peer et al. 2003). Klosterman et al. (2013) estimated that annual blackbird damage to oilseed sunflower and corn in the Prairie Pothole Region (PPR) of North Dakota averaged $US3.5 and $US1.3 million, respectively. These are direct costs of damage and do not include costs of damage management.

From 2001 to 2013 (except 2004), National Sunflower Association sponsored national surveys of blackbird damage in physiologically mature sunflower fields throughout the main sunflower growing states of the United States and the Canadian province of Manitoba. Annual reports are available on factors that limit national sunflower production (Berglund 2005, 2006, 2007, 2008, 2009; Kandel 2010, 2011, 2012, 2013; Lamey et al. 2002, 2003). In this paper, we analyze and summarize the magnitude of blackbird damage in 8 states in the U.S. and Manitoba over the most recent five years (2009-2013) of the survey.

METHODS

From mid-September to early October 2009-2013, 32 to 60 trained teams, including agronomists, entomologists, pathologists, crop consultants and producers, randomly selected one physiologically mature sunflower production field, for every 4,047 to 6,070 ha (10,000 – 15,000 acres). Planted hectares were determined by the USDA-Farm Service Agency and other state estimates. The exception is Vermont where most of the fields in an extension bio-diesel project were surveyed. Each team evaluated plant stand, yield potential, disease, insect, and weed issues for each field. They also assessed bird damage and agronomic practices used in the field. A sunflower seed sample was taken from each field to detect insect damage in the laboratory.

Yield was estimated in two random locations within the field. Surveyors entered the field in a random location and walked ≥ 25 m from the field edge, stopping in a representative area of the field. The second representative location was selected at ≥ 100 m further into the field. Yield was based on plant stand, head size, seed size, percent filled seeds, center seed set, and percent loss due to bird feeding. Plant stand was estimated based on counting all consecutive yield-contributing plants in 7.6 m within the row. Head diameter was measured for 5 consecutive heads in the row. Five wedges, one from each head, were cut out of the head and seeds were hand shelled. Average seed size was determined comparing seed sample with a chart (Anonymous 2008). One hundred seeds were evaluated for seed fill and percent filled seed determined. The center area of the head without seeds was measured and subtracted from the production estimate. Loss due to bird damage was estimated based on sample charts with examples of various levels of bird damage (Anonymous 2008).

We used arithmetic means and standard errors to describe central tendency and accuracy of the damage estimates. We used analysis of variance to assess statistical differences in damage between confectionery and oilseed hybrids and among study years.

RESULTS

From 2009 to 2013, sunflower damage caused by blackbirds and plant lodging (i.e., plants that fall on the ground and are unharvestable) ranked fifth (behind plant spacing, disease, drought and weeds) as the most limiting factors on production (Table 1). Among biological issues, blackbird damage to sunflower ranked 3rd behind disease and weeds.

Percentage of sunflower damaged did not differ across the five study years ($F_{4, 65} = 0.95$, $P = 0.440$). Thus, we combined study years for further analyses. Percentages of oilseed and confectionery sunflower hybrids damage also did not differ ($F_{1, 68} = 0.64$, $P = 0.427$). However, confectionery and oilseed hybrids produce achenes which are fundamentally different in oil content, size and hull thickness. Confectionery achenes are also sold at a ~35% premium over oilseeds (NASS 2015). Thus, we present damage data for both variety types.

We pooled the data over years and found mean percent blackbird damage was 2.5% in oilseed fields and 1.9% in confectionery fields (Tables 2, 3). Average annual blackbird damage was valued at $US13.4 million and $US4.9 million for oilseed and confectionery sunflower, respectively. Of the 8 states and Manitoba, North Dakota growers suffered the highest economic damage, with average annual losses of $US8.7 million for oilseed and $US2.0 million for confectionery hybrids (Table 4). South Dakota ranked 2nd and 3rd in total annual damage to oilseed sunflower ($US3.4 million) and confectionary fields ($US905), respectively. Nebraska ranked 2nd ($US1.2 million) in damage to confectionery hybrids.

Of the 951 oilseed and confectionery sunflower fields surveyed, 72% had ≤ 1% damage, 16% were >1 and ≤5%, 8 % >5 and ≤ 15% and 4% >15%. Across all years, 122 fields (12%) had damage >5%. This level is often considered significant economic damage and thus might warrant damage management actions (Linz et al. 2011).

Table 1. In late summer 2009 to 2013, trained teams assessed 951 physiologically mature oilseed and confectionery hybrid sunflower fields for yield and production limiting factors, in 8 states and the Canadian province of Manitoba. Percentage of fields with production limited by each listed agronomic factor were calculated for data pooled across years (n=5).

	First Limiting Factor		Second Limiting Factor	
	Mean	SE	Mean	SE
Plant spacing	20	1.6	13	1.5
Disease	18	3.6	9	1.0
Weeds	8	1.0	9	1.0
Lodging	7	1.9	6	1.6
Birds	7	0.5	4	0.6
Drought	11	4.9	4	1.0
Other	7	0.9	8	1.6
Insects	4	0.8	6	1.1
Uneven plant growth	3	0.3	3	1.2
Drown out	1	0.7	1	0.6
Hail	1	0.4	1	0.5
No problem	13	0.7	36	2.0

Table 2. During late summer 2009 to 2013, trained teams assessed physiologically mature oilseed hybrid sunflower fields for agronomic characteristics, including blackbird damage, in 8 states and Manitoba. Mean value of damage (@$US0.49/kg) and SE were calculated for data pooled across years (n=5).

	Harvested (10^3 ha) [1]		Sampled Fields	Yield (kg ha) [1]		Percent Bird Damage		Damage Value ($US/ha)	
	Mean	SE	N	Mean	SE	Mean	SE	Mean	SE
North Dakota	251	29	378	1774	81	4.2	0.9	36.4	7.8
South Dakota	195	14	164	1820	112	1.7	0.8	17.0	9.0
Kansas	40	8	26	1514	145	1.0	1.0	9.9	9.9
Colorado	29	4	31	1156	156	0.3	0.3	2.5	2.5
Minnesota	15	2	37	1744	120	0.8	0.3	7.4	3.0
Texas	16	3	23	1241	86	0.3	0.2	1.5	0.9
Manitoba	10	2	6	1784	86	2.0	1.0	18.9	22.7
Nebraska	11	<1	15	1342	169	5.2	2.3	36.2	19.4
Vermont	2	<1	41	1694	198	7.2	0.6	58.9	5.3

[1]Estimated production prior to bird damage based on NASS (2015) reported production.

Table 3. During late summer 2009 to 2013, trained teams assessed mature confectionary hybrid sunflower fields for agronomic characteristics, including blackbird damage in 8 states and Manitoba. Mean value of damage (@$US0.66/kg) and SE were calculated for data pooled across years (n=5).

	Harvested (10^3 ha) [1]		Sampled Fields	Yield (kg ha)[1]		Percent Bird Damage		Damage Value ($US/ha)	
	Mean	SE	N	Mean	SE	Mean	SE	Mean	SE
North Dakota	41	8	79	1764	98	4.5	1.2	53.6	15.3
South Dakota	32	6	31	1905	86	1.7	1.6	23.6	21.6
Texas	17	2	21	1333	261	0.0		-	
Nebraska	8	2	11	1734	368	5.8	4.6	93.8	79.9
Manitoba	27	9	43	1837	239	1.9	0.3	24.0	6.3
Minnesota	7	2	25	1762	241	0.8	0.6	8.8	5.8
Colorado	8	2	14	1585	147	0.5	0.5	5.3	4.7
Kansas	8	1	6	1716	38	0.0		-	
Vermont	0		-	-		-		-	

[1]Estimated production prior to bird damage based on NASS (2015) reported production.

Table 4. During late summer 2009 to 2013, oilseed and confectionery sunflower fields were assessed for blackbird damage in 8 states. Oilseed was valued @$US0.49/kg and confectionery was valued @$US0.66/kg[1]. Data were pooled across years.

	Oilseed		Confectionery	
	Mean Damage $US 10^3$	SE	Mean Damage $US 10^3$	SE
North Dakota	8708	1498	2019	510
South Dakota	3395	1874	905	834
Texas	26	17	0	0
Nebraska	360	186	1234	1113
Manitoba	218	145	637	267
Minnesota	134	62	71	48
Colorado	70	70	72	68
Kansas	559	559	0	0
Vermont	11	1	-	-

[1]NASS (2015)

DISCUSSION

Our data show that sunflower damage caused by blackbirds and plant lodging ranked fifth (behind plant spacing, disease, drought and weeds) as the most limiting factors on production. The amount of precipitation falling on fields is an uncontrollable environmental factor. Plant spacing can be addressed with changes in planting depth and seed density, and plant lodging might be reduced with selection of an appropriate hybrid.

Blackbird damage ranked 3rd behind disease and weeds among biological issues that limited production. Improved pesticides are now available for controlling disease, weeds (and insects). On the other hand, sunflower growers have limited cost-effective options for addressing blackbird damage (Linz et al. 2011).

From 2009 to 2013, bird damage in North Dakota averaged 4.2% in oilseed sunflower compared to an average loss of 2.7% in 2009 and 2010 reported by Klosterman et al. (2013). We inspected the data and found that percentage of bird damage was similar in both studies during 2009 and 2010. The higher percentage damage in our study might be related to 30% fewer hectares harvested from 2010 to 2013 compared to 2009 and 2010 (NASS 2013).

Bird damage was highly variable within and among the sampled states and Manitoba. This is not surprising as blackbirds tend to be clustered around certain landscape features, such as wetlands and trees that are favored roosting sites and the availability of food, particularly sunflower. The availability of preferred roosting sites and food also can vary among years as a result of extreme environmental events (e.g., drought, flooding).

Our data showed that the birds ate 35% more oilseed achenes ($\bar{x}$ = 2.5%) than confectionery achenes ($\bar{x}$ = 1.9%). These percentages are not statistically different due to high variance; nevertheless, the arithmetic difference might be biologically important.

Oilseed hybrids produce achenes that have a higher oil content, smaller size and thinner hull than do confectionery hybrids. These factors can affect the birds' food selection when given a choice between oilseed and confectionery hybrids (Mason et al. 1991). That is, as the confectionery achenes mature, it becomes more difficult for the birds to obtain the kernel, forcing the birds to search for more easily acquired food (Linz et al. 1984). The second author (GML) has observed that when confectionery and oilseed fields are planted in juxtaposition, invariably the oilseed field will suffer a greater percentage of damage. We hasten to add, however, when a confectionery field is the only source of food, the birds will cause significant economic losses.

There is no doubt that the potential for significant economic losses due to blackbird feeding is real. Additionally, feeding flocks are highly visible in ripening fields, further adding to the perception of huge losses. Despite the overall economic losses from other sources (e.g., disease and weeds), producer surveys show blackbirds are a major cause of declining sunflower hectares in the PPR (Kleingartner 2003, Klosterman et al. 2013, Hulke and Kleingartner 2014). The lack of management techniques that are consistently effective for reducing damage and the availability of alternative profitable crops that suffer less bird damage likely contribute to a decline in planted hectares (Linz and Hanzel 2015).

MANAGEMENT IMPLICATIONS

Bird damage to sunflower is an especially difficult problem for producers because damage occurs from early seed-set until harvest; however, most of the damage occurs before the achenes reach physiological maturity (Cummings et al. 1989). Thus, resources dedicated to management efforts should be focused when birds are first noticed in the fields. Linz et al. (2011) and Linz and Hanzel (2015) suggested that producers develop a bird management plan that might include modifying roost habitat; using a plant desiccant to accelerate fall harvest; using propane cannons; planting decoy crops; synchronizing planting time of sunflower with neighbors; leaving stubble, especially sunflower, unplowed to provide alternative feeding sites; and planting short-stature sunflower to facilitate bird-hazing strategies.

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The Consumption of Metallic Lead and Its Effects on Tissue Lead Levels of Urban Eastern Gray Squirrels (*Sciurus carolinensis*)

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ABSTRACT: Eastern gray squirrels (*Sciurus carolinensis*) are known to routinely consume or be exposed to lead from many anthropogenic sources, including ingesting bullet fragments, and gnawing on flashing. However, there is little research on consumption of metallic lead in squirrels. To determine if squirrels purposefully consume and metabolize lead, we supplied lead in the form of ingots to determine if squirrels are primarily gnawing lead, but not ingesting any, or incidentally ingesting relatively small amounts and compared that to lead levels from untreated squirrels from the same area. We found that squirrels readily consumed the provided lead ingots. The pooled mean liver and muscle lead levels of treated squirrels was 2.790 ppm (n = 6; CI $\pm$ 3.478) and 0.524 ppm (n = 5; CI $\pm$ .159), respectively, compared with the pooled mean liver 0.374 ppm (n = 6; CI $\pm$ 0.079) and muscle 0.252 ppm (n = 6; CI $\pm$ 0.094) lead levels from untreated squirrels. Even though this was a relatively large effect size between the liver of the squirrels fed lead (Cohen's d = 1.00) and a smaller effect size between muscle tissue (Cohen's d = 0.28), the 2 groups were not statistically different, likely due to the small sample size. Because squirrels will readily consume anthropogenic lead, raptors and other predators may bioaccumulate this lead through their foraging behaviors.

Key Words: lead consumption, eastern gray squirrel, *Sciurus carolinensis*, lead toxicosis, environmental contamination

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INTRODUCTION

Eastern gray squirrels (*Sciurus carolinensis*) have been known to routinely gnaw lead and damage items constructed from lead (McKinnon et al. 1976, Lewis et al. 2001, Pokras and Kneeland 2008). Though this phenomena is well documented, the cause for this behavior is not known (Pokras and Kneeland 2008) and there is little research on deliberate consumption of metallic lead in squirrels. Medvedev (1999) examined lead levels in several species of wildlife in Russia, including a native species of squirrel (*Sciurus vulgaris*), and found elevated liver lead levels. The authors speculated this liver lead elevation to be due to the squirrel's preference for mushrooms. In addition, liver lead levels were higher than muscle lead levels. We know that eastern gray squirrels will metabolize large quantities of lead (McKinnon et al. 1976, Lewis et al. 2001); however, the source of lead exposure in eastern gray squirrels seems to be variable. McKinnon et al. (1976) speculated the source of lead in their study to be both inhaled from leaded gasoline and ingested while foraging in the urban environment. While aerosolized lead is no longer a significant problem (EPA 2012), metallic lead is still commonly found in association with human activity. Eastern gray squirrels were found to have elevated tissue lead levels in a study conducted at the Federal Law Enforcement Training Facility in Glynn County, Georgia. This study determined that lead was being ingested in the form of lead bullet fragments as the animals foraged at a firing range, but it was unclear if this was deliberate or a consequence of foraging in areas with large amount of lead fragments. The authors also speculate that some species may have been attracted to the lead bullets because of the taste of the oxidized lead salts that formed on the fragments over time (Lewis et al. 2001), but no evidence was provided that squirrels purposefully seek out and consume lead. We have also observed squirrels gnawing lead from buildings in Murray, Kentucky and Hollywood, Maryland, but it is unclear if they are gnawing or consuming lead.

While we know that squirrels commonly gnaw metallic lead, we do not know if they are gnawing it for behavioral objectives and inadvertently ingesting it, if they metabolize it when they ingest lead, it, or if they are ingesting lead but it is passing it through the digestive system without being metabolized. In order to address these two questions we supplied anthropogenic lead in the form of ingots to determine if squirrels are primarily gnawing lead, but not ingesting any or incidentally ingesting relatively small amounts; and, if squirrels are ingesting lead, are they metabolizing it or is the lead being passed rapidly through the digestive tract without significant absorption.

STUDY AREA AND METHODS

Our study area was Murray, Kentucky, a small city of approximately 18,000 residents, and is also located in the Jackson Purchase region of western Kentucky. We collected by trapping and shooting with a pellet rifle using Gamo© PBA Raptor non-lead pellets. In addition, road-killed squirrels were collected when available. Live captured squirrels were euthanized with inhaled carbon dioxide (American Veterinary Medical Association 2013). Treated squirrels were collected from locations where they were actively removing metallic lead from soft lead ingots that we placed at sites where squirrels were known to have damaged lead components on homes in the past.

We collected liver and muscle samples from each squirrel. We combined muscle samples into ~6g pools consisting of ~1g of tissue from 6 squirrels and combined liver samples into ~5g pools consisting of ~1g of tissue from 5 squirrels for analysis. Tissue samples were analyzed by the Breathitt Veterinary Center Toxicology Laboratory (BVCTL) with atomic absorption spectroscopy. A Cohen's *d* effect size test was performed on tissue lead levels in addition to a one-tailed Welch's T-test to test for statistical significance between the squirrels in Murray and the squirrels collected from LBL. We also calculated a 95% confidence intervals (CI) to examine differences between the treated populations. Institutional Animal Care and Use Committee (IACUC) approval was obtained prior to the research (IACUC Number: 2012-016).

RESULTS

We collected squirrels from 30 untreated squirrels and 30 treated squirrels from the city of Murray. Squirrels in untreated and treated areas were pooled in to groups of 5-6 squirrels for testing. In a 3-month period (May-July 2013) approximately 184 g of metallic lead were removed from one site. Over a 2.25-year period, approximately 1,360 g of placed metallic lead was consumed. We recovered only a few small fragments of lead from the ground underneath locations where we placed the ingots.

Mean liver lead levels of treated squirrels was 2.790 ppm (n = 6; SE $\pm$ 1.353; CI $\pm$ 3.478) and 0.524 ppm (n = 5; SE $\pm$ 0.057; CI $\pm$ 0.159) from untreated squirrels. Pooled liver samples ranged from 0.25-9.24 ppm in treated squirrels and 0.42-0.73 ppm in untreated squirrels. Mean muscle lead levels from treated squirrels was 0.288 ppm (n = 6; SE $\pm$ 0.045; CI $\pm$ .1282) and 0.252 ppm (n = 6; SE $\pm$ 0.037; CI $\pm$ 0.094) in untreated squirrels. Pooled muscle samples ranged from 0.15-0.41 ppm in treated squirrels and 0.14-0.35 ppm in untreated squirrels. There was no statistically significant difference in the liver ($t_{1,5}$ = -1.67, P = 0.155) or the muscle ($t_{1,5}$ = -0.59; P = 0.284) between the treated and untreated squirrels; however, we did see a large effect size between liver (Cohen's d = 1.00) and a small effect size between muscle tissue (Cohen's d = 0.28).

DISCUSSION

Our research indicates that squirrels will actively seek out and consume lead, and that it is ingested and readily stored in the liver. While we could not positively demonstrate a difference in liver lead levels between the treated and untreated squirrels in Murray, the large difference between the means and the large effect size provide some evidence that this is the case. This supports previous evidence from Lewis et al. (2001) that squirrels will consume lead if it is readily available in the environment.

While we did see a large difference between the means, small sample size and large variance tempers these results. The large range in the pooled liver samples fed lead in Murray (0.25-9.24 ppm) indicate that it is likely that some squirrels were shot without having consumed lead. Squirrels were shot if they were in close proximity to our lead ingots; however, we had no way to determine if they had previously consumed lead. Ideally, individual squirrels would have been tested to account for this; however, it was necessary to pool the samples to provide enough tissue for sampling using the standard protocols at BVCTL. But even with the lack of statistical significance, the effect size, lack of lead fragments and shavings under limbs where lead was provided, and large range in liver levels indicate that squirrels purposefully consumed lead.

There was little difference in lead concentrations within muscle tissues between the two groups, suggesting that although squirrels readily metabolize and store lead in the liver, less lead is stored in muscle tissue. This is consistent with studies in swine (*Sus domesticus*) and cattle that found metabolized lead was stored more readily in liver tissue than in muscle tissue (Neimi et al. 1991). Medvedev (1997) also found higher concentrations of lead in squirrels (*Sciurus vulgaris*) than other tissues examined and relatively low lead levels in muscle tissue; however, concentrations of lead levels among other species and tissues were variable.

MANAGEMENT IMPLICATIONS

Squirrels have been known to metabolize lead in the vicinity of firearms ranges (Lewis et al. 2001). The precedent to issue consumption advisories due to possibly high lead concentrations in squirrel meat (EPA 2007, Division of Epidemiology 2009) could have consequence for sport hunting and consumption of squirrels near the thousands firing ranges and other areas where large quantities of available anthropogenic lead exist. While squirrel liver is not typically consumed by hunters , raptors and other carnivores may commonly consume squirrel liver and other organs when eating these prey item. Lead fragments from bullets are often cited as the primary source of lead for lead toxicosis in raptors (Kendall et al. 1995); however, if increased lead levels in squirrels and other rodents is wide spread, it may be that consumption of lead from anthropogenic sources are also a significant contributor. Sources of anthropogenic lead, including flashing, are still sold in home improvement stores, though acceptable non-lead alternatives area available. If increased lead concentrations in squirrels and other rodents is a concern for biomagnification, legislation may need to be enacted reducing the availability of commonly consumed anthropogenic lead sources.

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Poster Presentation

Sterilization, Hunting and Culling: Combining Management Approaches for Mitigating Suburban Deer Impacts

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ABSTRACT: Based on decades of growing deer impacts on local biodiversity, agricultural damage, and deer-vehicle collisions, in 2007 we implemented an increasingly aggressive suburban deer research and management program on Cornell University lands in Tompkins County, New York. We initially divided Cornell lands into a suburban core campus area (1,100 acres [4.5 km2]) and adjacent outlying areas that contain lands where deer hunting was permitted (~4,000 acres [16.2 km2]). We attempted to reduce deer numbers by surgically sterilizing deer in the core campus zone and increasing harvest of female deer in the hunting zone through an Earn-a-Buck program. During the first 6 years of this study, project staff spayed 96 female deer (>90% of all deer on campus); 69 adult does were marked with radio transmitters to monitor movements and survival. From 2008 to 2013, hunters harvested >600 deer (69–165 each hunting season). By winter 2013, we stabilized the campus deer herd to approximately 100 animals (57 deer/mi2 [22 deer/km2]), a density much higher than project goals (14 deer/mi2). Although we reduced doe and fawn numbers by approximately 38% and 79%, respectfully, this decrease was offset by an increase in bucks that appeared on camera during our population study. In 2014, we supplemented efforts using deer damage permits (DDP) with archery sharpshooting over bait, and collapsible Clover traps with euthanasia by penetrating captive bolt. In concert with sterilization and hunting, the use of DDPs and deer capture resulted in a herd reduction of approximately 45% in just one year on core campus. Based on our experiences, we discontinued use of surgical sterilization, and modified hunting on Cornell University lands in 2014. Future impact mitigation efforts will focus on lethal deer control in huntable areas, and DDPs in areas closed to hunting.

Key Words: clover traps, culling, hunting, sterilization, suburban, white-tailed deer

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Poster Presentation

Distinguishing between Eurasian Wild boar Hybrids and Feral Swine Using Molecular Analyses

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ABSTRACT: Wild hogs (*Sus scrofa*) are a serious threat that impact natural areas, farmland, and even urban landscapes. They destroy personal property, predate on wildlife, displace native species, and destroy the diversity of native wetlands. Previous research has shown that examining the differences in the gene MC1R using molecular methods and the examination of the hair coat of wild hogs has the potential to identify wild hogs and hybrids from domestic species; however, this technique has also not been evaluated in such a manner that would make it useful for conservation officers and prosecutors in a court of law. Therefore, we propose to evaluate both the morphological and genetic methods as a tool for identifying wild hogs using the model of disease testing where the morphological methods are applied by field personnel as a screening test and the genetic methods are used in a confirmatory manner. The objective is to determine the accuracy and precision of each of these methods for identifying wild hogs in the US. We will compare the MC1R gene between samples of DNA from known Eurasian wild boar, domestic hogs, wild hogs exhibiting the white-tipped guard hair phenotype, and feral swine that do not exhibit the white-tipped guard hair. We will use gel electrophoresis will be used to differentiate between the various wild and domestic hogs breeds. We will also enlist biologists, students, and other wildlife professionals assess photos and patches of hair from each type of hog to determine the accuracy of morphological assessment for identifying wild hybrids and recently released feral hogs. We believe these methods will be instrumental for law enforcement to identify and prosecute individuals involved in the anthropogenic spread of wild hogs in Kentucky and throughout the US.

Key Words: DNA, Eurasian wild boar, feral swine, genetics, molecular, wild hogs

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Poster Presentation

Efficacy of Milorganite® as a Repellent for Domestic Mice

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ABSTRACT: The objective of this study was to determine the potential of Milorganite® as a repellent for the domestic house mouse (*Mus musculus*). Milorganite® is the biosolids by-product left from the activated sludge process from the Milwaukee Metropolitan Sewer District. Within a climate controlled building, two triangular enclosures consisting of panels (2.4m x 1.2m x .064m) resulting in 2.6m 2 floor surface area were secured to a concrete floor and provided with pine shavings and a container of water. Round metal containers (8.3cm x 3.0cm) were each secured to a 10cm x 20cm plastic lid and placed within the three corners of each enclosure. Two, 6-day treatment periods, consisting of three, 48-hour trials were conducted. During each trial, 100g of a pelleted feed was placed within each metal container. Treatments were applied to the plastic tray surrounding each feed container at a rate of 1g Milorganite®, 500mg Milorganite® or 0mg Milorganite®. Ten mature mice were placed within each enclosure for each 6-day treatment period. Consumption of the 100g pelleted feed in each container during each 48-hour trial was utilized to determine repellent potential. Consumption of feed across all trials were similar (p=.87) among mice for the control (49.6g ± 3.2), 500mg Milorganite® (49.7g ± 2.8) or 1g Milorganite® (50.7g± 2.7) treatments. It was also observed that mice would consume Milorganite®. Results of this study indicate Milorganite® was not effective as a repellent for mice.

Key Words: feed, house mouse, Milorganite, repellent

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Poster Presentation

Nonlethal Approaches to Wildlife Damage Management

Charlotte Conley
Defenders of Wildlife

ABSTRACT: Lethal wildlife management, especially that of large predators, is particularly controversial in the public eye. By contrast, proactive nonlethal approaches, including different livestock husbandry strategies, strategic grazing, guard animals, electric fencing and temporary fencing, can reduce, if not avoid, negative attention generated by lethal control. We have worked with producers on the ground for over 20 years to prevent and mitigate wildlife-livestock conflicts. Wildlife damage management is often viewed and conducted remedially to damage that has already occurred. However, we encourage a different paradigm, where conflict is prevented. Working with producers, communities, state, federal and tribal agencies, and local governments we have pioneered the use of a range of nonlethal tools and strategies for preventing wildlife-livestock conflict. Defenders' programs include polar bears, prairie dogs, bison, wolves, grizzly bears, and Florida panthers. Over the past 7 years, Defenders has managed a program using only nonlethal tools to protect over 25,000 sheep grazing annually in the "sheep super-highway" in the Sawtooth Mountains of Idaho, with losses of less than 30 sheep, and no wolves. We present preliminary findings of this community-based project, The Wood River Wolf Project, as evidence that nonlethal approaches to wolf-sheep conflict can be used to significantly reduce depredation and loss. Another of our non-lethal programs assists landowners and producers prevent conflicts with grizzly bears, through bear-resistant electric fencing incentives. This program, active since 2010, reimburses the landowner 50% of the cost of the bear-resistant electric fence around bear "attractants", such as chicken coops, beehives, fruit trees, livestock and compost piles. The program has resulted in over 150 fences installed in high priority conflict zones within Montana, Idaho, and Wyoming. Using a proactive nonlethal approach can mitigate or prevent wildlife-livestock conflict, circumventing the public response to remedial lethal control. Proactive solutions may also be applied to a greater number of livestock operations, not just those experiencing conflict, but those that may. Here we feature two significant conflict prevention programs that use different approaches, with the potential for application elsewhere or on a broader level.

Key Words: community-based, nonlethal tools, wildlife-livestock conflicts

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Poster Presentation

GSM---Based Telemetry to Define Turkey Vulture Movements at Key West Naval Air Station

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ABSTRACT: Throughout its North American range, the turkey vulture (*Cathartes aura*) appears to be thriving. Turkey vulture populations wintering at Key West Naval Air Station (KWNAS), Florida are no exception to this trend. As vulture numbers continue to increase, so do potential conflicts with human activities. Abundant feeding opportunities and ample roost sites create ideal circumstances for wintering vultures. The increasing TUVU winter population is of particular concern because of the potential interaction with aircraft using the landing facility at KWNAS. Prior to developing vulture management recommendations at KWNAS, we needed to learn more about TUVU activity and movement patterns. Here we present results from 2013-2014 on trapping and marking efforts, with emphasis on vulture movement data acquired using GSM transmitters.

Key Words: GSM transmitters, Key West Naval Air Station, turkey vulture, wintering

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