Ecology of Mouflon Sheep Hybrids in Hawaii and Estimation of Population Size Using Imagery of Unmarked Animals

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CHAPTER 1

Space-Use and Survival of Feral Sheep (Ovis gmelini musimon and O. aries) in Hawaii

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INTRODUCTION

Taxa that are introduced to areas outside of their native range are classified as exotic species. The rate of exotic colonization has increased in tandem with rates of travel and trade due to human globalization (Buenavista & Palomares, 2018; Crowley et al., 2017; Hulme, 2009; Perrings et al., 2005). The introduction of exotic species can be unintentional (e.g. unknowingly carrying seeds or pests in cargo or on people) or intentional (e.g. importing a species to provide hunting opportunity). Regardless of their medium of introduction, exotic species have been identified as a major management concern for plant and wildlife biologists worldwide (Burton et al., 1992; Myers et al., 2000).

Managing exotic wildlife is critical for the preservation of endemic biodiversity (Butchart et al., 2010). This need is especially true for island ecosystems, where native species have evolved in relative isolation and may have reduced resistance to competition, predation, or herbivory (Bode et al., 2015; Myers et al., 2000; Weller et al., 2011). Ungulates, more specifically, may have high impacts on vegetation and soil structures (Hannon & Bradshaw,

2013; Long et al., 2017). For example, several species of endemic reptiles reliant on understory vegetation were found to be negatively impacted by the herbivorous pressure of unmanaged populations of spotted deer (Axis axis) in the Andaman Islands (Mohanty et al., 2016). Bird abundance and diversity was negatively influenced by unmanaged populations of Sitka black-tailed deer (Odocoileus hemionus sitkensis) utilizing significant levels of understory vegetation which bird populations used for forage and shelter (Chollet et al., 2013; Chollet et al., 2015).

The Hawaiian archipelago provides many more instances of the fast-growing need for management of exotic wildlife, especially considering its high concentration of both alien species and threatened endemic species (Eldredge, 2006; Kraus & Duffy, 2010). Hawaii's remote location and historic isolation contributed to the establishment of a unique ecosystem, as the only species that colonized the islands were those with capabilities for dispersal sufficient to traverse the Pacific Ocean (Ziegler, 2002). An increased demand for hunting over the past hundred years has created a niche for managing game species as a potentially lucrative resource, with hunting activities often doubling as a form of population control (Lohr et al., 2014). With human assistance, several species of game mammals have become established throughout the main islands, including pigs (Sus scrofa), goats (Capra hircus), axis deer (Axis axis), sheep (Ovis aries), and mouflon (O. gmelini musimon). Our study focuses on the ecology of the two lattermost species.

Mouflon-feral hybrid sheep (hereafter, sheep) have been identified as a considerable source of habitat pressure (Scowcroft & Sakai, 1983). Sheep graze, browse, and strip bark of endemic plants that have not evolved alongside large herbivores (Scowcroft & Giffin, 1983; Scowcroft & Sakai, 1983). Sheep have also been theorized as the primary source of habitat damage affecting the palila bird (Loxioides bailleui), an endemic species of Hawaiian honeycreeper. Studies conducted on Mauna Kea have indicated a correlation between mouflon

presence in critical palila habitat and subsequent habitat loss. The mamane tree (Sophora chrysophylla), an important source of forage and shelter for palila, has demonstrated particular vulnerability to browsing (Paul C. Banko et al., 2014; Hess & Banko, 2011) though drought events and climate change have also played a considerable role in the already steady decline of optimal mamane climate (P. C. Banko et al., 2013; Hess & Banko, 2011). Court-ordered eradications have since been conducted to control sheep populations on Hawai'i, including Mauna Kea (Paul C. Banko et al., 2014; Duffy, 2010). However, there has been little follow-up on how these removals affected critical habitats. With the majority of biological and public opinions favoring eradication (Hess, 2016), there is limited information on space-use and survival rates of sheep in Hawaii, and how these data can be utilized to better manage the remaining populations.

Our objective is to determine the seasonal space-use and survival of mouflon hybrids in a dryland forest reserve on the island of Hawai'i. Due to limited heterogeneity in seasons, we predict a lack of migratory behavior for both males and females and consequently high overlap of spatial use between seasons. Additionally, we predict that high range productivity and minimal predatory risk (sole predators including wild dogs and hunters for adults, and pigs for lambs) will contribute to a reduced need for movement and consequently smaller home-range sizes. Furthermore, we predict the abundance of resources will mitigate for forage density-dependence and, in addition to low predation risk, will result in relatively high survival rates for all sexes and age classes.

METHODS & MATERIALS

Study Area

The selected study area encompasses the Pu'u Wa'awa'a and Pu'u Anahulu (hereafter PWW and PAH, respectively) game management areas south of the Mamalahoa highway on the island of Hawai'i (Figure 1). The area north of the highway is not excluded, but is often managed separately as multiple fence boundaries minimize sheep egress. The area south of the highway is approximately 25 km wide and 17 km long. Topography is characterized by a steady incline in elevation from 640 m to 2000 m. An inactive lava flow loosely follows the western limit of PAH, and the Pohakuloa Training Area to the southeast provides a fenced boundary. Mount Hualālai is located at the southern end of PWW. Although Hualālai is an active volcano, no lava flows down the northern slope into PWW. A prominent volcanic cinder cone provides a recognizable landmark and observation point just south of the highway (Giffin, 2003). It is a common source of tourism and foot traffic.

Vegetation varies between the PWW and PAH areas. The dryer of the two is PAH, classified as montane dryland forest and grassland dominated by fountain grass (Pennisetum setaceum). Lower elevations at PWW are classified as montane dryland forest, and higher elevations on Hualālai classified as subalpine (Giffin, 2003). With roughly 90 percent of the flowering plant species native to Hawaii, PWW is endemically diverse (Wagner et al., 1990) and many of those species are endangered. Subalpine forests in PWW are characterized by 'ohi'a trees (Metrosideros polymorpha) with shrub understories of pukiawe (Styphelia tameiameiae) and 'ōhelo 'ai (Vaccinium spp.) all native to Hawaii. Among the many endemic and endangered plant species in the lower montane dryland forests are notably po'e (Portulaca sclerocarpa), a'e (Zanthoxylum hawaiiense), and narrowleaf stenogyne (Stenogyne angustifolia). Dry forests are dominated by 'ohi'a, naio (Myoporum sandwicense), and a'ali'I (Dodonaea viscosa). In addition to 'ohi'a, there are also small distributions of mamane and koa (Acacia koa), all of which have

significant history as building materials or jewelry in Hawaiian culture (Giffin, 2003; Wagner et al., 1990).

Climate in PWW and PAH is relatively dry in comparison to the mesic forests found throughout Hawai'i, despite updrafts caused by the warming Hualālai slopes generating routine cloud cover during the afternoon. There is limited seasonal heterogeneity, but a wet (Apr-Oct) and dry (Nov-Mar) season are commonly agreed on (Giffin, 2003), with monthly rainfalls ranging from 25mm to 45mm. Mean monthly temperatures are highest in September (86° F) and lowest in February (81° F).

Field Work

In September 2016 with the assistance of the Hawaii Division of Forestry and Wildlife (DOFAW; via a private capture company) we will capture sheep via helicopter net-gunning and ground-herding. Upon capture, sheep will be fitted with GPS or VHF collars to remotely track their movements and will be released at the point of capture. There will be 24 total GPS collars available for deployment and will be programmed to record locations in lat/long format every 6 hours at 2:00, 8:00, 14:00, and 20:00. Every two days all fixes will be transmitted to a database via satellite where they will be available for download. In October 2017, an additional 23 collars will be available for deployment. Additionally, 16 VHF collars are available for deployment.

Following release, we will regularly download and archive locations for visualization in GIS software via ArcGIS Pro (ESRI, 2017). Collars are programmed to send alerts when movements indicate possible mortality. Technicians provided by DOFAW will retrieve collars that are in a state of mortality, and will determine cause and time of death as soon as possible through postmortem examination of carcass and surrounding area. Collars will be routinely tracked via radio telemetry with the assistance of DOFAW technicians whenever possible.

Data Analysis

We will model individual home-ranges of sheep for different time periods. GPS locations will be divided into 3, 6, and 12 month periods starting with October 2016. We will assess accuracy of GPS locations by evaluating horizontal dilution of precision (HDOP). To estimate home-range size, we will compute utilization distributions (UD) through the Brownian Bridge Movement Model (Horne et al., 2007; Nielson et al., 2013). Brownian Bridge (BB) utilizes coordinates and associated timestamps to estimate variance around GPS locations and is particularly useful when time intervals are relatively short (Anile et al., 2018). We will utilize the package "BBMM" (Nielson et al., 2013) within the statistical software R (Team, 2016) to produce BB UDs for every individual and export them for visualization and analysis in ArcGIS Pro (ESRI, 2017). Explanatory variables (gender, age, season, year, study area, etc.) will be used to model home-range size and will be assessed through a corrected Akaike's Information Criterion (AICc) to determine top models (Akaike, 1973; P. Burnham & R. Anderson, 2002). We will then evaluate the strength of covariates by determining the impact of their beta coefficients on home-range size.

To estimate seasonal and annual rates of survival in relation to covariates of interest (gender, body condition, age, average daily movement, etc.), we will use known-fate models in Program MARK (White & Burnham, 1999). We will also assess relative model support using AICc (Akaike, 1973; P. Burnham & R. Anderson, 2002). In the event of model uncertainty, we will use model averaging to produce estimates of survival. Significance in covariates will be evaluated by examining confidence intervals around β values.

CONCLUSIONS

Home-range and survival estimates are essential components in developing wildlife management goals. Our aim is to provide managers with the information they need to implement cost-effective and ecologically sound plans. With an understanding of space-use of sheep, managers will be able to identify areas of concern, especially those with endemic plants and wildlife. Additionally, survival estimates will aid in setting harvest limits or increasing tags available to hunters.

CHAPTER 2

Estimating Population Size Utilizing Imagery of Unmarked Animals

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INTRODUCTION

Accurate and unbiased estimates of population size are crucial in the decision-making process of wildlife management (Keever et al., 2017; Le Moullec et al., 2017; Murray et al., 2009). Knowledge of population size can be used to determine harvest limits and set management goals, especially regarding large game animals such as ungulates. However, confidently determining population size for ungulates is often limited by non-uniform distribution, elusiveness of species, and low detection due to dense vegetation (DeCesare et al., 2012; Duquette et al., 2014). In addition, traditional approaches to estimating population size can be expensive, risky, and invasive.

Common approaches to population monitoring include aerial surveys. While widely used due to their ability to cover large areas in a short amount of time (Terletzky & Koons, 2016), they also have several statistical and logistic limitations. For instance, sightability of animals can be reduced by vegetation, topography, and behavioral conditioning (e.g. fleeing at the sight or sound of aircraft) which often result in population counts that are biased low (Terletzky & Koons, 2016; Zabransky et al., 2016). Variable detection rates caused by low sightability are demonstrated by Reilly (2017) to reduce the statistical power of aerial surveys. Double counting has also been noted as a source of observation error (Terletzky & Koons, 2016). The cost of conducting aerial surveys also reduces cost-effectiveness.

Remote cameras have become increasingly popular as a flexible and relatively inexpensive tool for estimating population size through models such as capture-recapture (CR) (Rovero & Marshall, 2009). The CR model relies on individuals being uniquely identifiable. This is possible for species with unique pelage patterns, especially felids (Karanth & Nichols, 1998; Silver et al., 2004). A species of antelope in South Africa, the nyala (Tragelaphus angasii), provide an example of a cryptic ungulate with unique stripes and spots (Marshal, 2017),), however many ungulates like the white-tailed deer (Odocoileus virginianus) and bighorn sheep (Ovis canadensis) have less distinct markings between individuals. Estimates derived from CR for these species often rely on anthropogenic marks (e.g., ear tags and radio collars) to identify individuals upon recapture (Moeller, 2017). Furthermore, error associated with misidentification of marked animals can restrict the accuracy and reliability of these models. For example, remote cameras cannot ensure an animal is postured properly in an image where an ear tag is clearly visible. Equipment quality, adverse weather conditions, and user error can also increase the probability of misidentification (Keever et al., 2017).

When treating camera data as a random sample of density, otherwise known as Instantaneous Sampling (IS), animals do not need to be uniquely identifiable. Each remote camera treats the area in front of it as a local sample of density whenever an image is taken. When deployed randomly, camera data represents a proportionate sample when the size of the study area is known, according to sampling theory. Cameras deployed over time continuously sample density, providing temporal replication (Moeller, 2017).

Our objective is to estimate population size utilizing imagery of unmarked feral sheep (Ovis gmelini musimon and Ovis aries). To test the viability of IS, we will also estimate population size of mouflon with CR and compare the results of both methods. We predict that

the superior spatial and temporal replication of counts through IS will provide unbiased estimates of population size. We also predict that greater amounts of replication will return smaller confidence intervals. Due to unknown rates of redetection frequency, we anticipate it will take fewer camera trapping days to return a suitable dataset for IS than for CR.

METHODS & MATERIALS

Study Area

The selected study area encompasses the Pu'u Wa'awa'a and Pu'uanahulu (hereafter PWW and PAH, respectively) game management areas south of the Mamalahoa highway on the island of Hawai'i. (Figure 1). The area north of the highway is not excluded, but is often managed separately with multiple fence boundaries minimizing sheep egress. The area south of the highway is approximately 25 km wide and 17 km long. Topography is characterized by a steady incline in elevation from 640 m to 2000 m. An inactive lava flow loosely follows the western limit of PAH, and the Pohakuloa Training Area to the southeast provides a fenced boundary. Mount Hualālai is located at the southern end of PWW. Although Hualālai is an active volcano, no lava flows down the northern slope into PWW. A prominent volcanic cinder cone provides a recognizable landmark and observation point just south of the highway (Giffin, 2003). It is a common source of tourism and foot traffic.

Vegetation varies between the PWW and PAH areas. PAH is generally the drier of the two, classified as montane dryland forest and grassland dominated by fountain grass (*Pennisetum setaceum*). Lower elevations at PWW are classified as montane dryland forest, and higher elevations on Hualālai classified as subalpine (Giffin, 2003). With roughly 90 percent of the flowering plant species native to Hawaii, PWW is endemically diverse (Wagner et al., 1990) and

many of those species are endangered. Subalpine forests in PWW are characterized by ‘ohi’a trees (*Metrosideros polymorpha*) with shrub understories of pukiawe (*Styphelia tameiameiae*) and ‘ōhelo ‘ai (*Vaccinium spp.*) all native to Hawaii. Among the many endemic and endangered plant species in the lower montane dryland forests are notably po’e (*Portulaca sclerocarpa*), a’e (*Zanthoxylum hawaiiense*), and narrowleaf stenogyne (*Stenogyne angustifolia*). Dry forests are dominated by ‘ohi’a, naio (*Myoporum sandwicense*), and a’ali’I (*Dodonaea viscosa*). In addition to ‘ohi’a, there are also small distributions of mamane and koa (*Acacia koa*), all of which have significant history as building materials or jewelry in Hawaiian culture (Giffin, 2003; Wagner et al., 1990).

Climate in PWW and PAH is relatively dry in comparison to the mesic forests found throughout Hawai’i, despite updrafts caused by the warming Hualālai slopes generating cloud cover most afternoons. There is limited climatic heterogeneity, but a wet (Apr-Oct) and dry (Nov-Mar) season are often referenced throughout the state (Giffin, 2003), with monthly rainfalls ranging from 25mm to 45mm. Mean monthly temperatures are highest in September (86° F) and lowest in February (81° F).

In September 2016 and October 2017 with the assistance of the Hawaii Division of Forestry and Wildlife (DOFAW; via a private capture company) we will capture sheep via helicopter net-gunning throughout the study area and herding where accessible by ground. Sheep will be marked with ear tags to enable re-sighting and in some cases were fitted with radio collars (GPS and VHF) which in combination with ear tags provide a higher probability for re-sighting.

Camera sites will be determined before deployment by randomly generating points throughout PWW and PAH in ArcGIS Pro (ESRI, 2017). Due to predicted logistical constraints from terrain ruggedness and visibility, we will develop a buffer derived from average daily

movement to ensure cameras are not spaced too close. This buffer will be given to each point to allow for flexible camera placement at the discretion of the user.

During late June through October 2017 we will deploy 95 Reconyx (PC900 HyperFire) cameras. Cameras will be anchored 2-6 feet above ground to whatever is available at the site, preferably trees for convenience or if they are not available, t-posts. Each camera will face in a general SW direction to minimize sun glare. The actual camera location will be recorded on deployment to facilitate retrieval and calculate the sample's average nearest neighbor (Morrison, 2008). This way we can determine if the distribution of cameras is statistically random.

Each camera will be programmed to take pictures when triggered through motion-detection and infrared sensors. Sheep detected after triggering a camera will represent a CR dataset. Additionally, each camera will be programmed to take a photo every 15 minutes regardless of triggering. Sheep detected in this manner will represent an IS dataset. In this way we will be able to acquire two separate datasets over the same period of data collection.

Data Analysis

We will use mark-recapture models in Program MARK (White & Burnham, 1999) to estimate population size from the CR dataset. We will use the statistical software R (Team, 2016) to estimate population size from the IS dataset. The IS method relies on zero-inflated Poisson statistics to estimate population size. Whenever a camera takes a photo on the designated 15-minute time interval, that photo acts as an instantaneous sample of local density. We predict a majority of non-detections from these photos. As developed by Moeller (2017), IS predicts local density ($\bar{D}$) in a camera's viewable area (a_{ij}) by taking the mean count n_{ij} at location i = 1, 2, …, M and occasion j = 1, 2, …, J as illustrated by the following:

$$\overline{D} = \frac{1}{J} \cdot \frac{1}{M} \sum_{j=1}^{J} \sum_{i=1}^{M} \frac{n_{ij}}{a_{ij}}$$

We define viewable area (a_{ij}) as a sector with a lens angle (θ_{ij}) of 42.5 degrees (according to Reconyx PC900 Hyperfire specifications) and a maximum view distance of (r_{ij}). We determined a constant maximum view distance of 35m by taking several images with sheep at multiple distances and determining at which point it becomes too difficult to safely identify an individual by species with a range-finder. View area is calculated as follows:

$$a_{ij} = \pi r_{ij}^2 \frac{\theta_{ij}}{360}$$

Assuming that each site has equal detectability of individuals and that the population is evenly distributed across the study area with a known size (A), we calculate population size (N) by:

$$N = A * \overline{D}$$

To account for probable difference in site detectability, we will implement bootstrapping with many repetitions (eg. 10,000) and estimate variance by calculating the standard deviation of population estimations.

If IS and CR are comparable, our results can provide a more cost-effective and less invasive method of population estimation. Managers will have access to a tool that can be applied broadly across species and regions, and will have better access to information that is vital to decision-making.

FIGURES

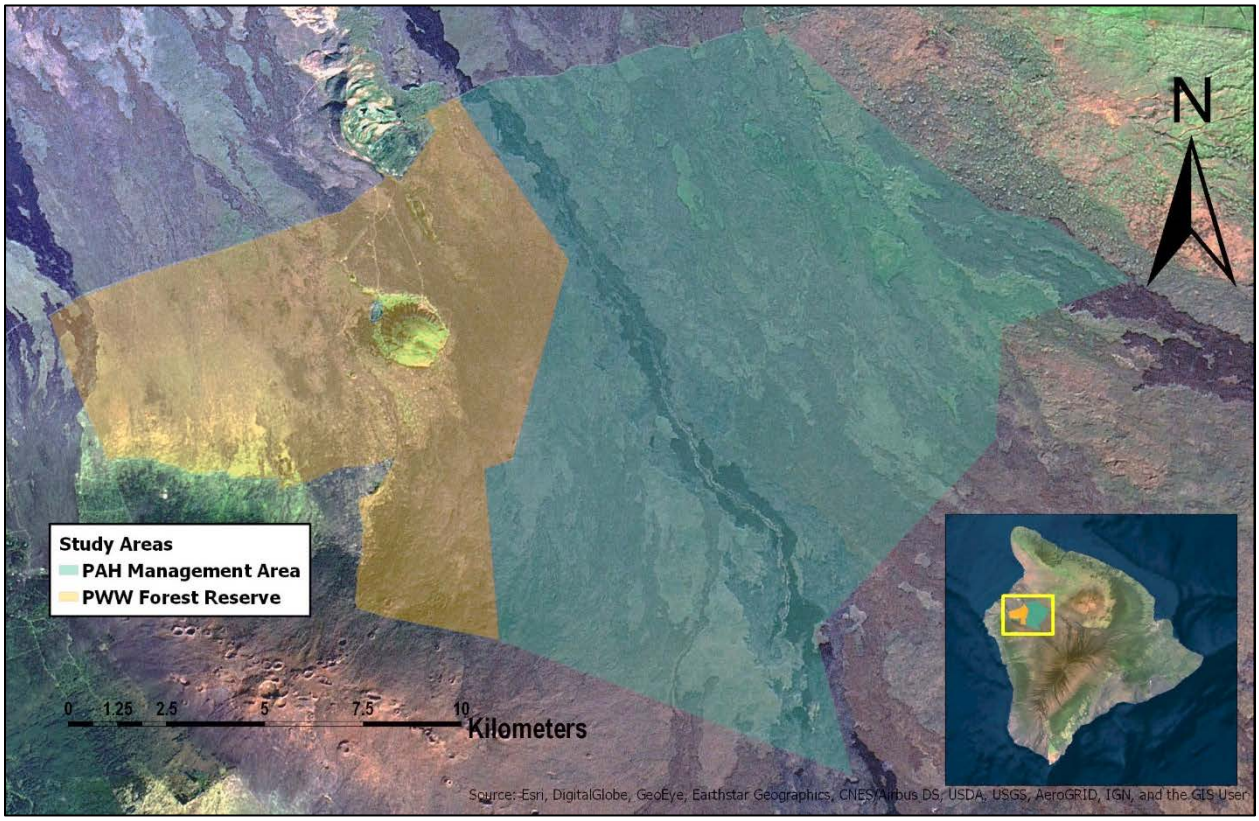

Figure 1: The study area encompassing PWW and PAH on the island of Hawai'i.

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