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Using Remote Cameras to Estimate the Abundance of Ungulates

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Master of Science

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ABSTRACT

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Many wildlife populations globally are experiencing unprecedented declines, and without accurate and precise estimates of abundance, we will not be able to conserve these vulnerable species. Remote cameras have rapidly advanced as wildlife monitoring tools and may provide accurate and precise estimates of abundance that improve upon traditional methods. Using remote cameras to estimate abundance may be less expensive, less intrusive, less dangerous, and less time consuming than other methods. While it is apparent that remote cameras have a place in the future of wildlife monitoring, research, and management, many questions remain concerning the proper use of these tools. In an effort to answer some of these questions, we used remote cameras to study a population of Rocky Mountain bighorn sheep (*Ovis canadensis*) in Utah, USA from 2012 to 2014. In Chapter 1, we compared methods using remote cameras against 2 traditional methods of estimating abundance. In Chapter 2, we evaluated the relationship between deployment time of cameras and proportion of photos needed to be analyzed to obtain precise estimates of abundance. We found that methods using remote cameras compared favorably to traditional methods of estimating abundance, and provided a number of valuable advantages. In addition, we found that remote cameras can produce precise estimates of abundance in a relatively short sampling period. Finally, we identified the optimal sampling period to produce precise estimates of abundance for our study population. Our findings can help researchers better utilize the potential of remote cameras, making them a more suitable alternative to traditional wildlife monitoring.

Keywords: camera trap, remote camera, population size, estimate of abundance, mark-resight, helicopter, optimization, bighorn sheep, *Ovis canadensis*, Antelope Island State Park

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I have found that it is impossible to stop my mind from admiring, examining, and exploring the natural world around me. I am grateful for the opportunities to embrace my fascination; may it never end.

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CHAPTER 1

Using Remote Cameras to Estimate the Abundance of Ungulates: A Comparison of Multiple Mark-Resight Methods

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ABSTRACT

Many wildlife populations globally are experiencing unprecedented declines. Precise estimates of abundance are needed in order to effectively conserve and manage vulnerable species. Remote cameras have advanced as wildlife monitoring tools and may provide improved methods of estimating wildlife abundance. We compared 4 different methods of estimating abundance on a population of Rocky Mountain bighorn sheep (*Ovis canadensis*) in Utah, USA from 2012 to 2014. These methods were 1) helicopter surveys, 2) ground surveys, 3) remote camera surveys with individually identified marked animals, and 4) remote camera surveys without individually identified marked animals. Remote camera surveys without individually identifying marked animals produced the most precise estimates of abundance. This method was also less expensive, less intrusive, and safer than helicopter surveys. In addition, this method was less time consuming than ground surveys and remote camera surveys with individually identified marked animals. Our results show that remote camera surveys without individually identified marked bighorn sheep can produce precise estimates of abundance that are safer, less intrusive, less expensive, and less time consuming than traditional methods. We believe that this method of estimating abundance may apply favorably to other species of ungulates that aggregate.

INTRODUCTION

Wildlife populations across the world are threatened with unprecedented declines from a variety of present-day challenges (May 2010). For example, shorebird populations worldwide are decreasing (Butchart et al. 2010), half of the world's ungulate and carnivore species are at a higher risk of extinction than in the 1970's (Di Marco et al. 2014), and 43% of all amphibian species have experiencing recent declines with 33% being at risk of extinction (Stuart et al. 2004). The challenges facing wildlife populations include habitat loss (Baker et al. 2004), global climate change (Hoegh-Guldberg and Bruno 2010), over-exploitation (Stuart et al. 2004), and increasing human-wildlife conflict due to growth in human populations (Virani et al. 2011). While there is much that is being done to mitigate these losses, it is likely that these challenges will only amplify in the future.

One of the difficulties we face in responding to these challenges is precisely estimating population size. Accurate and precise estimates of abundance are essential to effectively conserve and manage wildlife (Hofmeester et al. 2017). Having accurate and precise estimates of population size allows for the monitoring of survival rates, reproductive rates, sex ratios, and other meaningful metrics (Bowden et al. 1984). These metrics enable us to take the correct prescriptive actions in response to disease, human encroachment, and habitat loss. In addition, we are better able to develop trust with public stakeholders when using sound methods to estimate abundance (Freddy et al. 2004). For all these reasons, we should employ methods that provide the most precise estimates of abundance. Advances in technology and sensors coupled with drastic declines in cost provide an opportunity for development and use of innovative methods to estimate abundance.

One of the most promising technological innovations that is being used to estimate abundance is the remote camera. The technology used in remote cameras has advanced rapidly over the past two decades, allowing these cameras to become an integral tool in wildlife assessment and monitoring (Cutler and Swann 1999, Sanderson and Trolle 2005, Rowcliffe and Carbone 2008). While remote cameras are being utilized at a rapidly increasing rate (Rowcliffe and Carbone 2008), comparisons are still needed to determine if methods using remote cameras are truly suitable alternatives to traditional methods (Meek et al. 2015). Remote cameras have been used to monitor wildlife around the world in a variety of applications (Karanth et al. 2006, Michalski and Peres 2007, Tobler et al. 2009), including estimation of abundance, but to date this approach has mostly been done with uniquely patterned carnivores (Burton et al. 2015). Relatively little research has been done to explore the suitability of using remote cameras to estimate the abundance of ungulates (Rovero and Marshall 2009, Perry et al. 2010).

Traditional methods of estimating population size for ungulates include helicopter surveys (Krausman and Hervert 1983, Holl et al. 2004) and resight surveys performed from the ground (Bleich 1998, McClintock and White 2007). Helicopter surveys have been used increasingly during the past twenty years and are often the most common method used to monitor ungulates and to produce estimates of population size (Krausman and Hervert 1983, Bleich et al. 1994, McClintock and White 2007). Helicopter surveys, however, may have detrimental effects on the movements, foraging behaviors, and health of ungulates (Bleich et al. 1994, Bodie et al. 1995, Cote 1996, Frid 2003). For example, Rocky Mountain bighorn sheep (*Ovis canadensis canadensis*) that were surveyed using helicopters doubled their movements on the day of the survey (Bleich et al. 1990), suffered from decreased foraging efficiency on the day following the survey (Stockwell et al. 1991), and moved 2.5 times farther the day after the survey than the day before the survey (Bodie et al. 1995). The implications that these altered behaviors have on the predation rates and nutrition of bighorn sheep is not completely understood (Bleich et al. 1994). Moreover, conducting aerial surveys is the leading cause of work-related deaths among wildlife biologists (Sasse 2003). Consequently, resight surveys performed from the ground have been used as a less-intrusive and safer method for producing mark-resight estimates of bighorn population size (McClintock and White 2007), as well as other ungulates (Wingard et al. 2011, Corlatti et al. 2015). However, the large amount of time required to search for animals and observe marked individuals is a challenge associated with this method (e.g. McClintock and White 2007).

Remote cameras set at water sources have been used to increase our understanding of the ecology of bighorn sheep (Bleich et al. 1997, Whiting et al. 2009a, Whiting et al. 2010), as well as to estimate population size based on identifiable individuals (Jaeger et al. 1991, Perry et al. 2010). The estimates of population size produced from remote cameras in those examples, however, did not account for detection probability and individual heterogeneity which can limit both precision and accuracy of estimates. More recently, remote cameras placed at water sources were used to estimate the size of a population of mule deer (*Odocoileus hemionus*) with high precision after accounting for detection probability (Shields et al. 2012). Little is known, however, about how this method might apply to bighorn sheep and how it would compare to more traditional methods of estimating population size for bighorn sheep.

Herein, we used bighorn sheep as a model to analyze how remote cameras can be used to estimate the abundance of a mountain ungulate. Contemporary conservation and management of bighorn sheep populations often involves capturing and relocating individuals to augment existing populations in decline or to reintroduce bighorn sheep to historical habitat. These efforts depend on the number of individuals in source populations, and therefore it is necessary to have precise estimates of population size to effectively manage bighorn sheep (McClintock and White 2007). We compared the following three methods of estimating population size of bighorn sheep: helicopter surveys, resight surveys performed from the ground, and remote cameras deployed at water sources. Conducting those surveys allowed us to compare a relatively novel method of estimating abundance using remote cameras to conventional methods that have been widely accepted. We postulated that estimates of population size made using photos collected by remote cameras would be more precise than estimates made using either of the more conventional methods (sensu Perry et al. 2010). Our results will help wildlife biologists determine if new methods utilizing recent technological advancements can be used to estimate the number of individuals in populations of bighorn sheep. In addition, if this method is effective at estimating abundance of bighorn sheep at water sources, then the potential exists to use this method for other ungulates in similar settings.

METHODS

Study Area

Our study took place on Antelope Island State Park (AISP) (40°57’N, 112°13’W) in the Great Basin Desert of northern Utah, USA. Antelope Island is an 11,300 ha island located in the southeastern portion of the Great Salt Lake. This island is 24 km long, 11.3 km wide at the widest point, and ranges in elevation from 1283 m to 2134 m (Whiting et al. 2009a) (Fig. 1). The island is popular with recreationists and received approximately 300,000 visitors annually over the last decade (Olson et al. 2008). Human use of the island, however, was restricted to hiking, biking, and horseback riding on designated trails restricted to the outer edges of areas used by bighorn sheep (Fairbanks and Tullous 2002, Whiting et al. 2009b).

In 1996, 26 California bighorn sheep (*O. c. californiana*) captured in Kamloops, British Columbia, Canada (50°43’N, 120°25’W) were reintroduced onto Antelope Island (Whiting et al. 2009a). This population was later augmented in 2000 with 6 more California bighorn sheep from Winnemucca, Nevada (40° 58’N, 117°43’W) (Olson et al. 2008). Since 2001, the bighorn population on Antelope Island has provided over 209 animals to establish or augment three other populations in Utah (Olson et al. 2008). While all these animals were originally recognized as California bighorn sheep, morphometric evidence suggests that California bighorn sheep should not be considered different from Rocky Mountain bighorn sheep (Wehausen and Ramey 2000); and therefore, in this study we considered all animals as Rocky Mountain bighorn sheep (Whiting et al. 2009a). Limited hunting of bighorn sheep occurred during our study years and two male bighorn sheep were harvested during November on AISP each year beginning in 2011.

Other biota present on Antelope Island during our study were bison (*Bison bison*), mule deer, and pronghorn (*Antilocapra americana*). Potential predators of bighorn sheep on the island were bobcat (*Lynx rufus*), coyote (*Canis latrans*), and golden eagle (*Aquila chrysaetos*). Mountain lions (*Puma concolor*) were not observed or known to occur during our study. Predominant vegetation on the island consisted of big sagebrush (*Artemesia tridentata*), bluebunch wheatgrass (*Elymus spicata*), mountain spray (*Holodiscus dumosus*), purple three-awn (*Aristida purpurea*), Sandberg's bluegrass (*Poa secunda*), and wand mullein (*Verbascum virgatum*). The principle forage species utilized by bighorn sheep on Antelope Island were bluebunch wheatgrass, Sandberg's bluegrass, and wand mullein (Whiting et al. 2009b).

Capture and Marking

In February 2012, bighorn sheep were captured on AISP by contractors hired by Utah State Parks and fitted with radio collars. Prior to deployment, we marked each radio collar with a unique letter and number combination so that individual collared animals could be identified during subsequent sightings (Fig. 2). We used Lotek model 7000SU collars (Lotek Wireless Inc., St. John's, Newfoundland, Canada) with mechanical releases (n=20) and Lotek model LMRT collars (n=19). When a collar was inactive for ≥8 hours the radio emitted a pulse at a rate of 80 pulses/minute (mortality mode). Otherwise, the collar emitted a pulse at a rate of 40 pulses/minute (normal mode). Following deployment, we used a RA-150 directional antenna with a R-1000 telemetry receiver (Communications Specialists, Inc. 426 West Taft Avenue, Orange, CA 92865) to monitor the pulse rate of the collars. When a collar entered mortality mode, we assumed that the animal died and located the carcass as soon as possible to retrieve the collar. By November 2013, 18 of the model 7000SU collars had mechanically released and in February 2014, the bighorn sheep outfitted with the remaining 2 model 7000SU collars were recaptured and the collars were manually removed. In January 2014, 14 female, 6 male, and 5 lamb bighorn sheep were removed from AISP and translocated to the Oak Creek Mountain Range (39°19'N, 112°13'W) in central Utah, USA. To offset the marked animals that were lost to mortality and translocation, from January to March of 2014, we captured additional bighorn sheep on AISP and fitted each with radio collars. As in 2012, each of those radio collars was marked with a unique letter and number combination so that individual collared animals could be identified (Fig. 2).

Helicopter Surveys

We performed one helicopter survey in February of 2012, 2013, and 2014. We performed each survey in the same areas known to be used by bighorn sheep, and we attempted to spend a similar amount of time searching in each survey. Whenever bighorn sheep were sighted during those surveys, we classified animals into one of three groups; 1) adult female or yearling, 2) adult male, or 3) lamb (<1 year old). We also noted the presence of any collars on observed bighorn sheep. We performed telemetry surveys independent of the helicopter survey so that the number of collared bighorn sheep in the population that were available for resighting was known for each survey (Table 1). We used the Lincoln-Peterson method as described by

Pollock (1990) with the Chapman (1951) adjustment to estimate the abundance of adult females and yearlings, as well as adult males. We calculated error around estimates using the equations described by Bishop (1975) and produced 95% confidence intervals (CI) using the 0.5 transformed logit equations described by Sadinle (2009). To estimate abundances, error around estimates, and 95% CIs for lambs, we multiplied the respective estimate for adult females and yearlings by the overall observed ratio of lambs to adult females and yearlings for each survey (sensu McClintock and White 2006).

Ground Surveys

During July and August of 2012-2014, we conducted a total of 18 resight surveys (6 each year) from the ground using 10-12 power binoculars and 20-60 power spotting scopes with efforts focused on habitat used by bighorn sheep (hereafter referred to as ground surveys). In order to assume independence of samples, we allowed at least 10 days (SE = 0.68) to elapse between any two sequential surveys. To ensure that we knew the number of collared animals in the population during each survey, we monitored collars an average of 1.7 days (SE = 0.63) from the date of all resight surveys. No collared animals died during the sampling period for these surveys in any of the three years (Table 1). As with helicopter surveys, whenever bighorn sheep were sighted, observers classified those animals into one of three groups; 1) adult female or yearling, 2) adult male, or 3) lamb. Observers also noted the presence of collars on bighorn sheep. We used the logit-normal estimator without individually identifiable marks in Program MARK to estimate abundances, standard errors (SE), and 95% CIs for adult female and yearlings, as well as for adult males in each year surveyed (White and Burnham 1999, McClintock and White 2007;2012). To estimate abundances, SEs, and 95% CIs for lambs from ground surveys, we multiplied the respective logit-normal estimate for adult females and yearlings by the overall observed ratio of lambs to adult females and yearlings for each year (sensu McClintock and White 2006).

Remote Camera Surveys

We used information gathered from previous studies (Rogerson et al. 2008, Whiting et al. 2009a;b), as well as personal observation to identify water sources on Antelope Island most frequently used by bighorn sheep. We selected 5 natural water sources and monitored each with a Bushnell Trophy Cam^{TM} (Bushnell Outdoor Products, Overland Park, Kansas, USA) remote camera from July 1 to August 31 of 2012- 2014, the time of year that bighorn sheep on Antelope Island visit water most often (Whiting et al. 2009b). Additionally, those were the same time periods in which we performed our ground surveys. As described above, no collared bighorn sheep died during those periods and the number of collared bighorn sheep in the population was known during each period (Table 1). We placed cameras at the headwaters of the selected water source and focused the lens towards the area of greatest apparent wildlife activity. Digital cameras operated continuously, but only took images when motion was detected by Passive Infra-Red sensors. When motion was detected, the camera used infrared LEDs to take 8 MP images (Fig. 2). All cameras operated on medium sensitivity with a 20 second interval between images and used a single 8 GB secure digital high capacity (SDHC) card and 12 AA lithium batteries. SDHC cards and batteries were checked and replaced if needed every 40 days (SE = 2.3). Once images were collected, we analyzed each image and classified all visible bighorn sheep by age (adult/yearling/lamb), sex (female/male), presence or absence of a radio collar, and identified collared animals by their unique letter and number combination (Fig. 2).

After all images were analyzed, we selected the image with the greatest number of bighorn sheep for each age, sex, and collar combination at each of the five water sources for each day. For example, from all images collected at water source #1 on July 1, 2012, we selected 8 images for our analysis; the image that had the greatest number of 1) uncollared adult females, 2) collared adult females, 3) uncollared adult males, 4) collared adult males, 5) uncollared yearling females, 6) uncollared yearling males, 7) collared yearling males, and 8) lambs. By only using a single image, we eliminated the possibility of resampling the same individuals within a sampling period; we also avoided over representing the marked portion of the population which occurs when tallying unique individuals. That process of selecting images was repeated for each water source on each day of the sampling period. The 5 water sources used in our analysis were an average of 4.63 km (SE = 0.99) straight-line distance apart. Because of that distance, it was rare for an individual bighorn sheep to use more than one water source in a single day. Of the 1,054 times that an individual collared bighorn sheep was photographed at a water source during a day of our study, 95% of the time it was photographed at only 1 water source during that day. For that reason, we considered each day as an independent secondary sampling period, which allowed us to combine the selected images from each secondary sampling period (days) and thereby increase our secondary sample size and recapture rate. We recognize that this method of selecting images ignores the vast majority of the collected data, but it also provides a number of benefits; 1) reduced risk of sampling with replacement, 2) unbiased representation of marked and unmarked animals, and 3) rapid analysis by only considering the most valuable images. With the compiled data from our selected images, we estimated abundance using two separate analyses in Program MARK; 1) the logit-normal estimator using individually identifiable marks, and 2) the logit-normal estimator while ignoring individual marks. We chose the logit-normal estimator because our study met all the assumptions of this estimator, because the Bowden's estimator is no longer supported in Program MARK, and because the Poisson-log normal estimator only allows for 3-digit encounter histories. In our analyses, Program MARK produced estimates of abundance, SEs, and 95% CIs for adult females and yearlings, as well as for adult males in each year surveyed. To estimate abundance, SEs, and 95% CIs for lambs from camera surveys, we multiplied the respective logit-normal estimate for adult females and yearlings, by the overall observed ratio of lambs to adult females and yearlings for each year (sensu McClintock and White 2006).

RESULTS

Capture and Marking

In February 2012, we captured 26 adult females (approximately 30% of the adult female population) and 13 adult males (approximately 25% of the adult male population) on AISP and fitted each with a radio collar possessing a unique letter and number combination. From January to March of 2014, we captured an additional 8 adult females (approximately 10% of the adult female population), 17 adult males (approximately 30% of the adult male population), and 2 yearling males (approximately 15% of the yearling male population) on AISP and fitted each with collars similar to those used in 2012.

Helicopter Surveys

During helicopter surveys we noted if bighorn sheep were fitted with radio collars; but due to the distance from the animals and the nearly constant movement of the collars on running bighorn sheep, we were not able to identify individual collared animals by their unique letter and number combination while flying. In February of 2012 we observed 120 bighorn sheep, 19 of which were collared (Table 2). Using the Lincoln-Petersen estimator and observed ratios, we estimated the number of adult females and yearlings at 137 (SE = 35.3, CI = 98-250; Fig. 3), adult males at 58 (SE = 7.8, CI = 50-94), and lambs at 68 (SE = 14.9, CI = 42-106; Fig. 4). We estimated the bighorn sheep population at 263 animals (SE = 58.0, CI = 191-450; Fig. 5).

In February of 2013 we observed 62 bighorn sheep, 13 of which were collared (Table 2). We estimated the number of adult females and yearlings at 55 (SE = 9.5, CI = 45-91; Fig. 3), adult males at 29 (SE = 11.8, CI = 21-90), and lambs at 22 (SE = 2.9, CI = 14-28; Fig. 4). We estimated the bighorn sheep population at 105 animals (SE = 24.2, CI = 80-209; Fig. 5).

In February of 2014 we observed 117 bighorn sheep, 9 of which were collared (Table 2). We estimated the number of adult females and yearlings at 124 (SE = 35.2, CI = 90-248; Fig. 3), adult males at 224 (SE = 197.0, CI = 108-915), and lambs at 32 (SE = 8.1, CI = 21-57; Fig. 4). We estimated the bighorn sheep population at 381 animals (SE = 240.2, CI = 219-1,220; Fig. 5).

Ground Surveys

During ground surveys, we observed bighorn sheep at an average distance of 2.12 km (SE = 0.34). As a result of that distance, we were not able to consistently identify individual collared animals by their unique letter and number combination. However, we were able to determine whether any observed animal was collared or uncollared. During July and August of 2012 we observed a total of 263 bighorn sheep, 50 of which were collared (Table 2). Using the logit-normal estimator and observed ratios, we estimated the number of adult females and yearlings at 97 (SE = 12.4, CI = 73-122; Fig. 3), adult males at 47 (SE = 7.1, CI = 33-61), and lambs at 50 (SE = 6.4, CI = 38-63; Fig. 4). We estimated the bighorn sheep population at 194 animals (SE = 25.9, CI = 144-245; Fig. 5).

During July and August of 2013 we observed a total of 381 bighorn sheep, 37 of which were collared (Table 2). We estimated the number of adult females and yearlings at 99 (SE = 13.1, 73-125; Fig. 3), adult males at 58 (SE = 12.5, CI = 33-82), and lambs at 36 (SE = 4.7, CI =

27-45; Fig. 4). We estimated the bighorn sheep population at 192 animals (SE = 30.4, CI = 133-252; Fig. 5). During July and August of 2014 we observed a total of 317 bighorn sheep, 50 of which were collared (Table 2). We estimated the number of adult females and yearlings at 82 (SE = 9.7, CI = 63-101; Fig. 3), adult males at 55 (SE = 8.9, CI = 38-73), and lambs at 39 (SE = 4.6, CI = 30-48; Fig. 4). We estimated the bighorn sheep population at 176 animals (SE = 23.3, CI = 130-221; Fig. 5).

Remote Camera Surveys

We were able to identify 97% of the collared bighorn sheep captured in images by the unique letter and number combination on their collar, and 99% of the collared bighorn sheep that were selected for the analysis in Program MARK. During 2012, we sampled 310 camera days (camera active 24 hours at a water source) and collected a total of 11,806 images of bighorn sheep. From those images, we selected photos with 1,536 bighorn sheep for our analysis, 284 of which were collared and individually identifiable (Table 2). Using the logit-normal estimator with individually identifiable marks, we estimated the number of adult females and yearlings at 104 (SE = 10.8, CI = 83-125; Fig. 3), adult males at 39 (SE = 6.8, CI = 26-52), and lambs at 53 (SE = 5.5, CI = 42-64; Fig. 4). We estimated the bighorn sheep population at 196 (SE = 23.1, CI = 151-242) animals using the logit-normal estimator with individually identifiable marks (Fig. 5). Using the logit-normal estimator without individually identifiable marks, we estimated the number of adult females and yearlings at 106 (SE = 6.0, 94-118; Fig. 3), adult males at 33 (SE = 2.6, CI = 28-39), and lambs at 54 (SE = 3.1, CI = 48-60; Fig. 4). We estimated the bighorn sheep population at 194 (SE = 11.7, CI = 171-216) animals using the logit-normal estimator without individually identifiable marks (Fig. 5).

During 2013, we sampled 310 camera days (camera active 24 hours at a water source) and collected a total of 10,089 images of bighorn sheep. From those images, we selected images that contained 1,446 bighorn sheep for our analysis, 131 of which were collared (Table 2). We were able to individually identify all but 2 of the 131 collared bighorn sheep used in our analysis. Using the logit-normal estimator with individually identifiable marks, we estimated the number of adult females and yearlings at 104 (SE = 16.3, 72-136; Fig. 3), adult males at 112 (SE = 50.2, CI = 14-211), and lambs at 39 (SE = 6.1, CI = 27-51; Fig. 4). We estimated the bighorn sheep population at 255 (SE = 72.6, CI = 112-397) animals using the logit-normal estimator with individually identifiable marks (Fig. 5). Using the logit-normal estimator without individually identifiable marks, we estimated the number of adult females and yearlings at 93 (SE = 7.2, CI = 79-107; Fig. 3), adult males at 83 (SE = 18.8, CI = 46-120), and lambs at 35 (SE = 2.7, CI = 29-40; Fig. 4). We estimated the bighorn sheep population at 210 (SE = 28.8, CI = 154-267) animals using the logit-normal estimator without individually identifiable marks (Fig. 5).

During 2014, we sampled 293 camera days (camera active 24 hours at a water source) and collected a total of 10,614 images of bighorn sheep. From those images, we selected photos with 1,042 bighorn sheep for our analysis, 140 of which were collared (Table 2). We were able to individually identify all but 2 of the 140 collared bighorn sheep used in our analysis. Using the logit-normal estimator with individually identifiable marks, we estimated the number of adult females and yearlings at 97 (SE = 16.4, CI = 64-129; Fig. 3), adult males at 65 (SE = 12.1, CI = 41-89), and lambs at 44 (SE = 7.5, CI = 29-59; Fig. 4). We estimated the bighorn sheep population at 205 (SE = 36.0, CI = 135-276) animals using the logit-normal estimator with individually identifiable marks (Fig. 5). Using the logit-normal estimator without individually identifiable marks, we estimated the number of adult females and yearlings at 89 (SE = 7.8, CI = 74-104; Fig. 3), adult males at 66 (SE = 7.9, CI = 51-82), and lambs at 41 (SE = 3.5, CI = 34-47;

Fig. 4). We estimated the bighorn sheep population at 196 (SE = 19.2, CI = 159-234) animals using the logit-normal estimator without individually identifiable marks (Fig. 5).

DISCUSSION

Estimates obtained from remote cameras without individually identifiable marks were the most precise of all the methods used in our study. This method also provided the most consistent estimates of abundance across all three years surveyed, the smallest SEs and 95% CIs in 2012 and 2014, and the second smallest SE and 95% CI in 2013. For the SEs and 95% CIs in 2013, most of the variability in that estimate can be attributed to the low number of collared adult males. Low numbers of marked adult males in 2013 was a result of mortalities that occurred in the autumn of 2012, leaving only 5 adult males on Antelope Island, the lowest value during our study. With only 5 male bighorn sheep collared in 2013, only 5-10% of the male population was represented as available for resighting during ground and camera surveys. The portion of the population that is marked and available for resighting can directly affect the precision and accuracy of estimates (Roff 1973). Therefore, estimates of abundance produced by remote cameras were less precise when the portion of the marked population available for resighting was low. Even with that low number of collared males, however, estimates from remote cameras in 2013 compared favorably to estimates from the other two methods.

We predicted that using the individually identifiable marks on collared bighorn sheep would increase the precision of our estimates. However, we found that our estimates were more precise when we ignored the individual marks and only considered whether animals were collared or uncollared. A possible reason for this outcome was the low daily resight rate of individual collared bighorn sheep due to the process we used to select images for our analysis.

We employed a unique process of selecting remote camera images which allowed us to use a maximum of 3 images of collared bighorn sheep from each spring for each secondary sampling period (days) in our analysis. That method saved a great deal of time, because we analyzed only a small portion of the total collected images, but it meant rates associated with resighting bighorn sheep during secondary sampling periods was low. By ignoring the individual marks, we combined the low individual resight rates and created a modest group resight rate and thereby increased the precision of our estimates, which was encouraging, as the effort required to individually identify collared bighorn sheep was time consuming.

Using remote cameras without individually identifiable marks produced estimates that were particularly precise for the adult female and yearling portion of the population in our study area. That result likely occurred because adult female and yearling bighorn sheep travel in relatively large groups during the period that we sampled with remote cameras (Geist 1971), and because we had cameras placed at water sources spread out in the habitat predominantly used by adult females and yearlings. This sampling design resulted in us obtaining images with many bighorn sheep present and images of different groups of bighorn sheep, which allowed us to combine samples and increase the portion of the population that was resighted. The adult female and yearling portion of a population of ungulates is critically important as it drives population growth. Ungulate translocations are a vital practice used for a variety of purposes (Griffith et al. 1989, Seddon et al. 2007), but in most cases the adult female and yearling portion of the population is mainly used for translocations. Removing bighorn sheep can have detrimental effects on the source population (Stevens and Goodson 1993), and it can be harmful to remove animals when the source population is insufficiently large. Moreover, accurate and precise estimates of abundance of adult females and yearlings are needed in order to determine when a population of ungulates is sufficiently large to withstand a removal for translocation. Our method using remote cameras without individually identifiable marks provided a precise estimate of the number of adult females and yearlings in our population.

Our method using remote cameras without individually identifiable marks was also highly effective when estimating the number of bighorn sheep lambs in the population. Precise estimates of lamb abundance can be a valuable tool for monitoring the effects of disease and die-offs on bighorn sheep populations. Disease and die-offs regularly occur in bighorn sheep populations. Bighorn sheep lambs are often the most impacted group when a population of bighorn sheep contracts a disease (Forrester 1971, Woodard et al. 1974). By rapidly detecting a decrease in the abundance of lambs, we will be aware of die-offs sooner and better able mitigate the negative effects of the disease. Conversely, operating on inaccurate or imprecise estimates of lamb abundance will increase the risk of false-positive die-off detections; such a mistake could result in unnecessary expenses for a wildlife program.

Ground surveys were effective at estimating abundance during our study, which was consistent with their effectiveness in other studies ((McClintock and White 2007, Wingard et al. 2011). There were a number of factors that we found to be logistically difficult when performing ground surveys. First, a single survey typically required one technician working a full 10-hour day to complete. Second, we chose to have the same technician perform all ground surveys across all years of our study in order to avoid observer bias. Third, we allowed 10 days to elapse between sequential surveys in order to establish independence, but by having date-specific surveys we created the potential for conflicts with weather and work schedules. These complications should be considered when planning to use ground surveys to estimate ungulate abundance.

Helicopter surveys were not comparable to the other methods of surveying. The estimate of abundance produced from helicopter surveys did not fall within the 95% CIs of any of the other 3 methods during any of the years of our study. Helicopters were also far less precise than other methods in 2012 and 2014. One possible explanation for these imprecise estimates is the lack of independence between the method of marking and the method of resighting. The most fundamental assumption of mark-resight estimates is that the marked portion of the population is representative of the entire population in terms of sighting probability (Neal et al. 1993). If the marking of animals and the resighting of animals are performed via the same method (e.g. helicopter capture and helicopter resight), then there may be bias toward resighting marked animals at a higher rate than unmarked animals (Bowden and Kufeld 1995, McClintock and Hoeting 2010, McClintock and White 2012). The variability in our helicopter estimates may also be due the variability inherent to single-sample surveying. Estimates of abundance should be made with multiple surveys whenever possible (Sadinle 2009). In our study, the estimates produced from helicopter surveys were tenuous for making management decisions as they varied by hundreds of animals between years. When the variability of estimates is combined with the violation of mark-resight assumptions, the potential negative effects on wildlife, expense of surveying, and safety risk to observers, there are many reasons to carefully consider the appropriateness of helicopter surveys in estimating abundance.

Our study on AISP provided a unique opportunity with added geographic closure because it was separated from other mountain ranges. This allowed us to have an added amount of control in our study which strengthened our inferences. Bighorn sheep populations often occur in geographically isolated areas and our remote camera methods would apply in such areas. However, it would be beneficial to repeat these methods in other areas to better understand their efficacy in a variety of locations, especially when bighorn sheep populations interact more frequently. In such areas, remote cameras would likely produce better estimates of females than of males, as males travel between areas occupied by females.

We used remote cameras to estimate the abundance of ungulates with reasonable precision. Our method of surveying using remote cameras was more precise than helicopter surveys, as well as less time consuming and less logistically complicated than ground surveys. Our method meets all the assumptions of mark-resight surveying that are often violated with helicopter surveys. In addition, by selecting a small portion of the collected images, and by not investing time to individually identifying collared animals, time spent analyzing images can be reduced. Using remote cameras also creates a permanent record of photos that can be archived and reanalyzed to answer additional ecological questions at a future time. Our remote camera method could also be used for other ungulates that aggregate. In conclusion, we recommend that researchers and ecologists consider using remote cameras as an alternative method to estimate abundance of ungulates in an precise, safe, efficient, and inexpensive manner.

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FIGURES

Figure 1-1. Antelope Island State Park in northern Utah, USA, where we estimated population size of bighorn sheep (*Ovis canadensis*) from 2012-2014.

Figure 1-2. Image collected by a remote camera placed at a water source on Antelope Island State Park, Utah, USA showing a group of bighorn sheep. We classified each animal by its age and sex, and identified collared bighorn sheep by the unique letter and number combination on their collar.

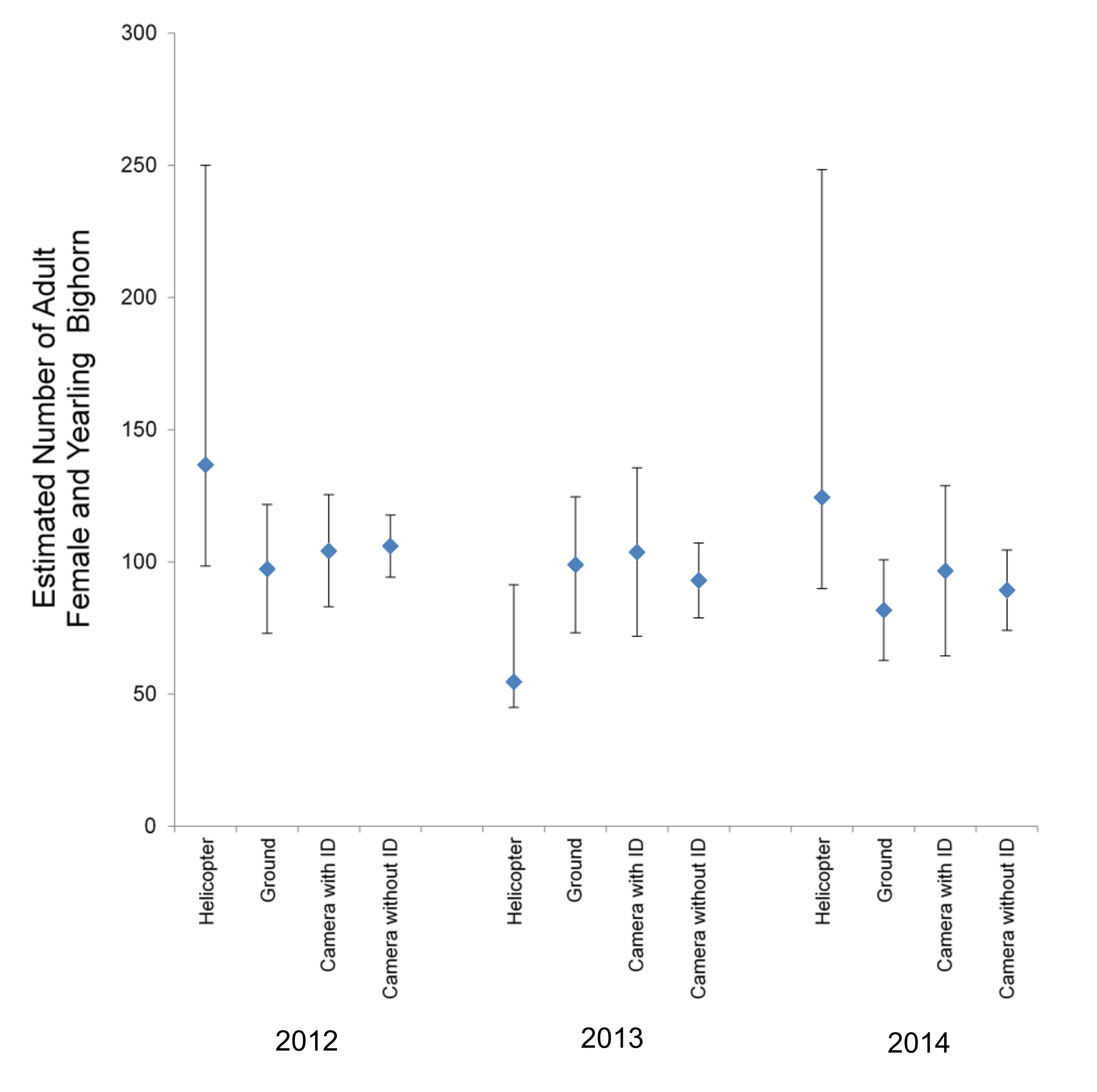

Figure 1-3. Estimates of abundance with 95% confidence intervals for adult female and yearling bighorn sheep (*Ovis canadensis*) on Antelope Island State Park, UT, from 2012-2014 using helicopter, ground, and remote camera surveys.

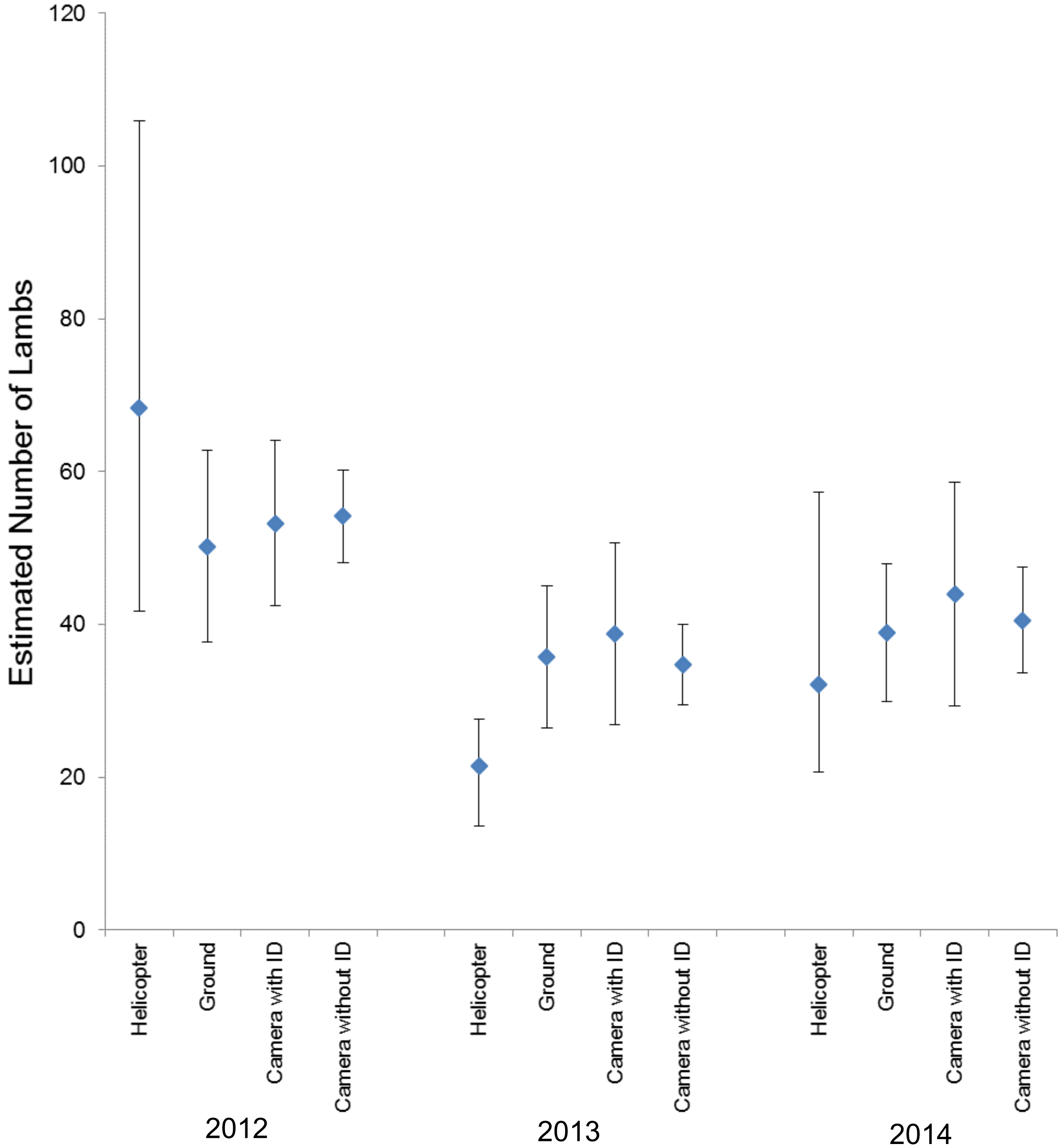

Figure 1-4. Estimates of abundance with 95% confidence intervals for the bighorn sheep (*Ovis canadensis*) lambs on Antelope Island State Park, UT from 2012-2014 using helicopter, ground, and remote camera surveys.

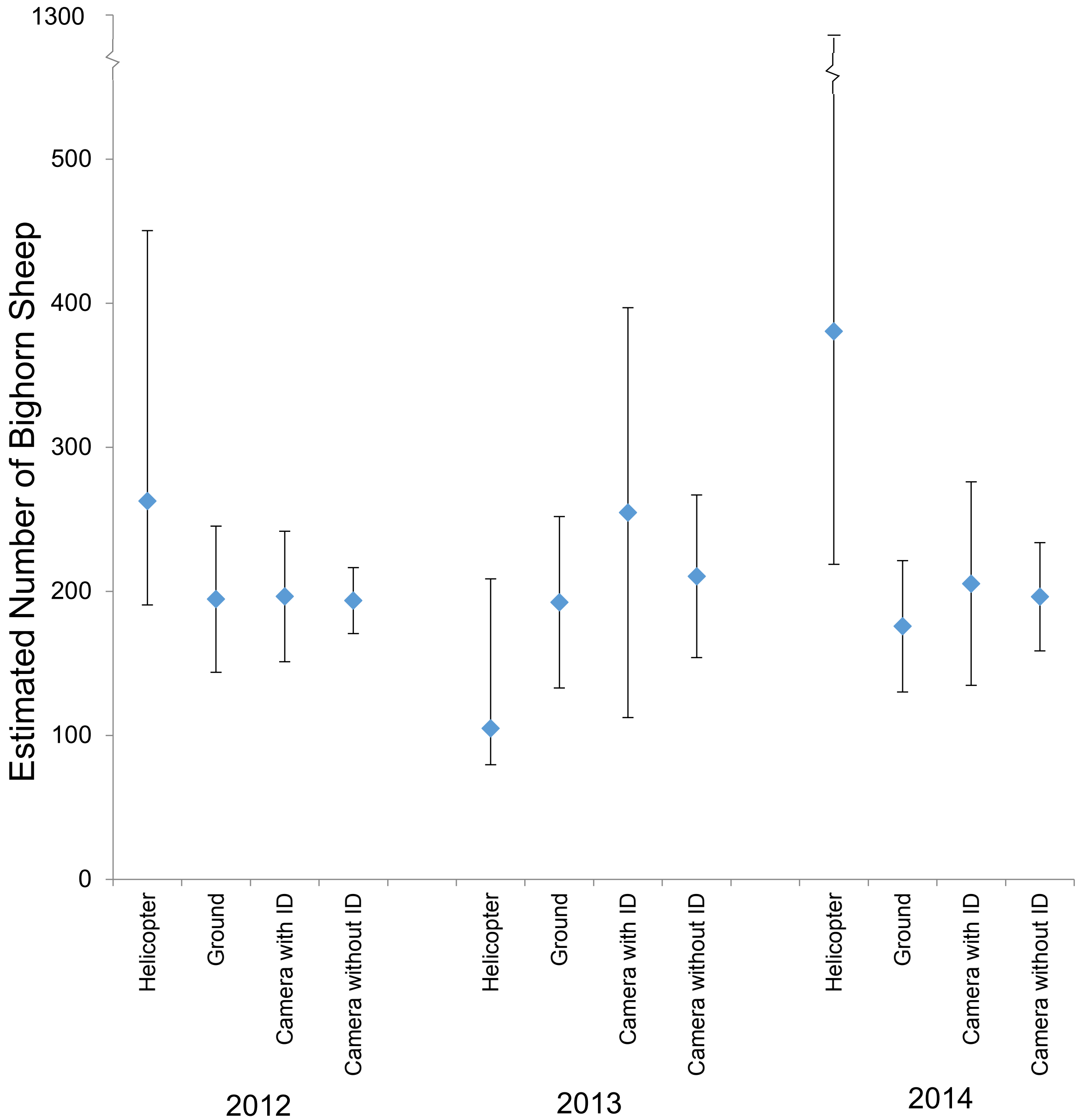

Figure 1-5. Estimates of abundance with 95% confidence intervals for the total population of bighorn sheep (*Ovis canadensis*) on Antelope Island State Park, UT from 2012-2014 using helicopter, ground, and remote camera surveys.

TABLES

Table 1-1. Collared bighorn sheep (*Ovis canadensis*) that were available for observation during helicopter, ground, and remote camera surveys performed on Antelope Island State Park, UT, from 2012-2014

Year	Survey	Collared Adult Females & Yearlings	Collared Adult Males
2012	Helicopter	26	13
	Ground	26	11
	Camera	26	11
2013	Helicopter	16	6
	Ground	14	5
	Camera	14	5
2014	Helicopter	16	14
	Ground	15	16
	Camera	15	16

Table 1-2. Total number of bighorn sheep (*Ovis canadensis*) observed from helicopter, ground, and remote camera surveys performed on Antelope Island State Park, UT, from 2012-2014.

	Adult Females & Yearlings		Adult Males			
Survey	Collared	Uncollared	Collared	Uncollared	Lambs	Total
Helicopter 2012	9	41	10	35	25	120
Ground 2012	34	94	16	53	66	263
Camera 2012	212	631	72	190	431	1536
Helicopter 2013	10	23	3	13	13	62
Ground 2013	26	159	11	118	67	381
Camera 2013	115	733	16	265	317	1446
Helicopter 2014	7	51	2	42	15	117
Ground 2014	31	139	19	47	81	317
Camera 2014	91	481	49	161	260	1042

CHAPTER 2

Optimizing the Use of Remote Cameras to Estimate Wildlife Abundance

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ABSTRACT

Remote cameras have rapidly become a powerful tool used to monitor, manage, and conserve wildlife. Projects using remote cameras often accumulate more images than it is feasible for technicians to analyze. These projects would benefit greatly by not analyzing more images than is needed to obtain satisfactory results, yet there is a lack of information about optimal sample size and deployment time required for remote camera surveys. We used remote cameras to estimate the abundance of adult females in a population of Rocky Mountain bighorn sheep (*Ovis canadensis*) in Utah, USA from 2012 to 2014. We then evaluated the relationship between deployment time of cameras and proportion of photos needed to be analyzed to obtain precise estimates of abundance. We compared estimates of abundance from 31 replicated analyses from each year, with each replicate considering a different number of days from a 62 day sampling period (July 1-Aug 31). In all years of the study, precise estimates of abundance were obtained with only 12 days of remote camera sampling. Very little precision was gained by analyzing more than 12 days of images. Our results indicate that an asymptotic relationship exists between camera deployment time and quality of estimates of abundance when using remote camera surveys; and we believe this a similar relationship exists for other wildlife

applications of remote cameras. Our findings can help researchers better utilize the potential of remote cameras, making them a more suitable alternative to traditional wildlife monitoring.

INTRODUCTION

Remote cameras are one of the most promising remote sensing tools for the future of wildlife ecology, conservation, and management (Cutler and Swann 1999, Rowcliffe and Carbone 2008, O'Connell et al. 2011, McCallum 2013, Burton et al. 2015). The widespread acceptance of remote cameras, combined with their improved technology, has triggered an increase in the number of scientific studies using these tools. For example, each year from 1998 to 2008, the number of published papers that addressed or used remote cameras increased 50% annually (Rowcliffe and Carbone 2008); from 2008 to 2013 this rate of growth continued to increase (Rovero et al. 2013); and this trend is projected to continue (O'Connell et al. 2011, Meek et al. 2014, Burton et al. 2015). This rapid growth, however, has resulted in a wide variety of methodologies lacking standardization, which has raised some concerns about incorrect applications and inappropriate inferences (O'Connell et al. 2011, Meek et al. 2015). As the use of remote cameras in wildlife research grows, researchers need to refine methodologies of using these instruments properly (Kays and Slauson 2008, Rowcliffe and Carbone 2008, O'Connell et al. 2011, Meek et al. 2014, Burton et al. 2015, Meek et al. 2015).

Efforts have been made to refine and standardize wildlife research using remote cameras. Some of these efforts include, defining common terminology and giving guidelines for standardized reporting of remote camera studies (Meek et al. 2014), providing guidelines to help researchers select the most appropriate camera model and settings (Rovero et al. 2013), and proposing a standardized data storage and analysis process (Harris et al. 2010). Those studies have been integral in guiding researchers in the proper terminologies, methodologies, and data storage and analyses, which has reduced the number of inconsistencies characteristic of previous studies. The need for such guidance is not abating; if researchers wish to maximize the potential of remote cameras, while avoiding the confusion, inaccuracy, and complexity inherent of these devices, then further research is needed to improve our methods of using them.

The popularity of remote cameras has been fueled by the rapid technological advancements that have made cameras smaller, more durable, more reliable, and more affordable (Sanderson and Trolle 2005). One of the greatest advancements of modern remote cameras is their ability to collect and store large quantities of images, facilitated by the transition to digital technology (Fegraus et al. 2011). However, the increasing ability to collect these images has outpaced our ability to analyze them (Sheil et al. 2013). Currently, the vast majority of images analyzed in studies using remote cameras must be individually classified by humans. While computer-automated systems are currently being developed to assist in differentiating species (Yu et al. 2013), distinguish pelage patterns (Hiby et al. 2009), and identify individual animals by unique morphological characteristics (Loos and Ernst 2013, Crunchant et al. 2017), these systems do not yet apply to the vast majority of remote camera research projects. Hopefully computer-automated systems will eventually progress to the point where they are widely used to analyze remote camera images, but it may take years before these systems are reliable and widely available. As researchers manually classify the potentially hundreds of thousands of images associated with their projects, analysis becomes a challenging and time-consuming task (Harris et al. 2010, Fegraus et al. 2011, McCallum 2013, Sheil et al. 2013, Yu et al. 2013). Researchers would benefit greatly by not analyzing more images than is needed to obtain satisfactory results, yet there is a lack of information when it comes to understanding the optimal sample size and deployment time required for remote camera projects (Meek et al. 2015).

One of the most common uses of remote cameras for wildlife research and management is to estimate wildlife abundance (e.g. Silver et al. 2004, Karanth et al. 2006, Head et al. 2013, Burton et al. 2015). Research using remote cameras to estimate abundance can collect large quantities of images, which require a significant investment in both time and money to analyze. In some circumstances, financial or temporal constraints may make it unrealistic for researchers to analyze every image that is collected by remote cameras. However, few guidelines exist concerning the minimum number of images needed in order to obtain a precise estimate of abundance. If researchers know the fewest number of images that are needed to precisely estimate abundance, they then can optimize their effort by not expending time or money when it does not improve the quality of their results.

We used remote cameras to estimate the abundance of a population of bighorn sheep in Northern Utah. We evaluated the relationship between deployment time of cameras and proportion of photos needed to be analyzed for precise estimates of abundance. We predicted that only a portion of the total collected images needed to be analyzed to produce results comparable to those produced using the complete data set and that an asymptotic relationship between number of photos and precise estimates of abundance existed. Our findings will help researchers decide how to most efficiently use remote camera data to obtain estimates of abundance; thereby, simplifying the process and magnify the value of remote cameras for wildlife research.

METHODS

Study Area

Our study occurred on Antelope Island State Park (AISP) located in the southeastern portion of the Great Salt Lake in northern Utah, USA (Fig. 1). The water level of the Great Salt

Lake fluctuates annually and so AISP can either be an island or a peninsula connected to the mainland (Kaze et al. 2016). AISP is 11,300 ha in size, 24 km long, 11.3 km wide at the widest point, and ranges in elevation from 1283 m to 2134 m. The state park receives approximately 300,000 visitors annually (Olson et al. 2008), and human use is restricted to hiking, biking, and horseback riding on designated trails (Fairbanks and Tullous 2002, Whiting et al. 2009b).

California bighorn sheep were recently introduced to our study area beginning in 1996 when 26 bighorn sheep were captured in Kamloops, British Columbia, Canada (50°43'N, 120°25'W) and released onto Antelope Island (Whiting et al. 2009a). Another 6 bighorn sheep were translocated from Winnemucca, Nevada (40° 58'N, 117°43'W) in 2000 (Olson et al. 2008). At the time, these animals were considered California bighorn sheep (*O. c. californiana*), but morphometric evidence suggests that California bighorn sheep should not be considered independent from Rocky Mountain bighorn sheep (Wehausen and Ramey 2000); and therefore, we considered all animals as Rocky Mountain bighorn sheep (Whiting et al. 2009a). Harvest of bighorn sheep on AISP was limited to 2 male bighorn sheep each November during our study.

Besides bighorn sheep, AISP is also inhabited by bison (*Bison bison*), mule deer (*Odocoileus hemionus*), and pronghorn (*Antilocapra americana*). Potential predators of bighorn sheep on AISP are bobcat (*Lynx rufus*), coyote (*Canis latrans*), and golden eagle (*Aquila chrysaetos*). Mountain lions (*Puma concolor*) were not detected during our study. Big sagebrush (*Artemesia tridentata*), bluebunch wheatgrass (*Elymus spicata*), mountain spray (*Holodiscus dumosus*), purple three-awn (*Aristida purpurea*), Sandberg's bluegrass (*Poa secunda*), and wand mullein (*Verbascum virgatum*) constitute the prevalent vegetation on AISP. Bighorn sheep in our study site predominantly forage on bluebunch wheatgrass, Sandberg's bluegrass, and wand mullein (Whiting et al. 2009b).

Capture and Marking

In February 2012, bighorn sheep were captured on AISP by contractors hired by Utah State Parks and fitted with radio collars. Prior to deployment, we marked each radio collar with a unique letter and number combination so that collared animals could be individually identified during subsequent sightings (Fig. 2). We used Lotek model 7000SU collars (Lotek Wireless Inc., St. John's, Newfoundland, Canada) with mechanical releases (n=20) and Lotek model LMRT collars (n=19). When a collar was inactive for ≥8 hours the radio emitted a pulse at a rate of 80 pulses/minute (mortality mode). Otherwise, the collar emitted a pulse at a rate of 40 pulses/minute (normal mode). Following deployment, we used a RA-150 directional antenna with a R-1000 telemetry receiver (Communications Specialists, Inc. 426 West Taft Avenue, Orange, CA 92865) to monitor the pulse rate of the collars. When a collar entered mortality mode, we assumed that the animal died and located the carcass as soon as possible to retrieve the collar. By November 2013, 18 of the model 7000SU collars had mechanically released and in February 2014, the bighorn sheep outfitted with the remaining 2 model 7000SU collars were recaptured and the collars were manually removed. In January 2014, 14 female, 6 male, and 5 lamb bighorn sheep were removed from AISP to be translocated to the Oak Creek Mountain Range (39°19'N, 112°13'W) in central Utah, USA. To offset the marked animals that were lost to mortality and translocation, we captured additional bighorn sheep on AISP from January to March of 2014 and fitted each with radio collars similar to those used in 2012.

Remote Camera Surveys

We used information gathered from previous studies (Rogerson et al. 2008, Whiting et al. 2009a;b), as well as personal observation to determine water sources on Antelope Island most used by bighorn sheep. We selected 3 natural water sources and monitored each with a Bushnell

Trophy Cam^{TM} (Bushnell Outdoor Products, Overland Park, Kansas, USA) remote camera from July 1 to August 31 of 2012-2014, the time of year that bighorn sheep on Antelope Island visit water source most often (Whiting et al. 2009b). We used radio telemetry to monitor the collared bighorn sheep in our study area and determined that no collared bighorn sheep died during the sampling period in any of the years sampled. We placed cameras at the headwaters of the selected water source and focused the lens towards the area of greatest apparent wildlife activity. Digital cameras operated continuously but only took images when motion was detected by Passive Infra-Red sensors. When motion was detected, the camera used infrared LEDs to take 8 MP images. All cameras operated on medium sensitivity with a 20 second interval between images and used a single 8 GB secure digital high capacity (SDHC) card and 12 AA lithium batteries. SDHC cards and batteries were checked and replaced if needed every 40 days (SE = 2.3).

Data Analysis

Due to time and budget constraints, we concentrated our efforts on adult female bighorn sheep as they are the most influential cohort on the growth of populations of bighorn sheep and therefore should be the focus of management efforts. Once images were collected, we analyzed each image and classified all visible adult female bighorn sheep by presence or absence of a radio collar. We were able to identify 97% of the collared bighorn sheep by the unique letter and number combination on their radio collar (Fig. 2). Identifying individual collared bighorn sheep allowed us to meet the requirement of sampling without replacement within secondary sampling periods, but we did not use the individual identifiers in our analysis because previous research has shown that it did not improve the precision of the estimates of abundance (Taylor et al. in prep.). After all images were analyzed, we selected 2 images per water source per day for our analysis, 1) the image with the greatest number of adult female bighorn sheep that were uncollared and 2) the image with the greatest number of adult female bighorn sheep that were collared. This process of selecting images was repeated for each water source on each day of the sampling period. The 3 water sources used in our analysis were an average of 1.6 km (SE = 0.4) straight-line distance apart. Because of that distance, it was rare for an individual bighorn sheep to use more than one of these water sources in a single day. Of the 762 times that an individual collared bighorn sheep was photographed at a water source during a day of our study, 94% of the time it was photographed at only 1 water source during that day. For that reason, we considered each day as an independent secondary sampling period. We then combined all of the selected images from each secondary sampling period (days) to compile the data for our analysis. This method of analyzing remote camera images has been shown to be effective for estimating population size for bighorn sheep (Taylor et al. in prep).

With those data, we used the logit-normal estimator without observing individually identifiable marks available in Program MARK to estimate abundances, standard errors (SE), and 95% confidence intervals (CI) for adult female bighorn sheep. Program MARK is an effective tool to estimate population size using mark-resight techniques (White and Burnham 1999, McClintock and White 2007;2012). We first estimated abundances, SEs, and 95% CIs for each of the 3 years using the complete 62 day sampling period (July 1 – August 31). We then systematically decreased the sample size of images by eliminating days from our model analysis in Program MARK. To test the most realistic scenario for sampling in a field setting, we chose to eliminate days from the end of the 62-day sampling period so as to maintain a contiguous sampling period for all replicates. We did this by eliminating 2 days at a time, so that our replicates were 62 days (July 1 – August 31), 60 days (July 1 – August 29), 58 days (July 1 – August 27), and so on until our final replicate of 2 days (July 1 – July 2). For all replicates, we chose to use a model in Program MARK where estimates of abundance varied by year, individual heterogeneity was fixed to zero, and resight probability varied by year and day. This model was most appropriate because the resight probability parameters applied to all replicates and because in the logit-normal estimator individual heterogeneity must be fixed to zero when marks are not individually identifiable.

RESULTS

Capture and Marking

In February 2012, we captured 26 adult female (approximately 30% of the adult female population) bighorn sheep on AISP and fitted each with a radio collar possessing a unique letter and number combination. From January to March of 2014, we captured an additional 8 adult female (approximately 10% of the adult female population) bighorn sheep on AISP and fitted each with collars similar to those used in 2012. Using telemetry surveys, we were successfully able to monitor the number of collared bighorn sheep that were available for resighting during all of our sampling periods. There were 26 marked adult females that were available for resighting in 2012, there were 14 available in 2013, and 13 available in 2014.

Remote Camera Surveys

During 2012, we sampled 186 camera days (camera active 24 hours at a water source) and collected a total of 10,150 images of bighorn sheep. From those images, we selected 583 adult female bighorn sheep for our analysis, 203 of which were collared. During 2013, we sampled 186 camera days and collected a total of 8,501 images of bighorn sheep. From those images, we selected 493 adult female bighorn sheep for our analysis, 121 of which were collared. During 2014, we sampled 169 camera days and collected a total of 10,003 images of bighorn sheep. From those images, we selected 371 adult female bighorn sheep for our analysis, 72 of which were collared.

Data Analysis

We performed 31 replicated analyses for each year for a total of 93 replicate logit-normal models in Program MARK. Estimates for 2012 ranged from 62 to 69 adult females and SEs for those estimates ranged from 3.5 to 17.4. Estimates for 2013 ranged from 44 to 58 adult females and SEs for those estimates ranged from 4.1 to 24.9. Estimates for 2014 ranged from 57 to 98 adult females and SEs for those estimates ranged from 6.2 to 58.6 (Fig. 3).

DISCUSSION

We obtained precise estimates of abundance with only 12 days of remote camera sampling. Very little precision was gained by analyzing more than 12 days of images in our study. In each year, all the estimates of abundance obtained from replicates with 12 or more days of sampling were contained in the 95% CI of the estimates obtained from the 62 day sampling period. Had only 12 days been analyzed as opposed to 62, in 2012 the estimated number of adult females would have changed by only 7 individuals, the SE would have increased only 3.25, and only 16% of the images would have been analyzed. In 2013 the estimate of abundance would have changed by only 3 individuals, the SE would have increased only 5.24, and only 8% of the images would have been analyzed. In 2014 the estimate of abundance would have changed by only 10 individuals, the SE would have increased 1.98, and only 29% of the images would have been analyzed (Fig. 3). Across the 3 years of the study, we could have avoided analyzing over 23,000 images, saving a significant amount of time and money while not significantly impacting the precision of the results.

It is important to note that not any 12 day sampling period would have produced satisfactory results, and that the optimal sampling period should be chosen carefully. In our study, we understood that we would maximize the number of bighorn sheep photographed at water sources by sampling in July and August as large groups visit these sites more frequently during these months (Whiting et al. 2009a). Researchers need to consider the type of site to be monitored with remote cameras (e.g. water source, mineral lick, trail intersection) and consider when they will photograph the greatest number of animals at this site. This will likely vary from location to location and potentially from year to year.

We were able to decrease the sampling period while not decreasing the precision of the estimates of abundance by increasing our resight probability in two ways. The first way that we increased our resight probability was by combining samples from multiple water sources within secondary sampling periods. By monitoring the individually identifiable marked bighorn sheep in our study area, we found that it was unlikely for a single bighorn sheep to use more than one of the water sources where remote cameras were deployed during secondary sampling periods (days). Because of this spatial independence, we were able to combine samples from all 3 water sources each day, thereby increasing our sample size and resight probability. The second way that we increased resight probability was by having more marks in the population. In 2012 we had the greatest number of adult female bighorn sheep marked during our study (n=26, approximately 30% of the adult female population); and even with only 2 days of sampling, we obtained an estimate that differed by only 4 individuals from the estimate obtained when using 62 days of sampling. When using remote cameras to estimate abundance by mark-resight analysis, researchers should attempt to increase their resight probability and thereby decrease the number of images that need to be analyzes to obtain precise results. Keep in mind the most fundamental assumption of mark-resight analyses is that the marked portion of the population must be representative of the entire population in terms of sighting probability. Any efforts that are made to increase the resight probability of marked animals must equally increase the sighting probability of unmarked animals. For example, reflective marks should not be placed on animals if they increase the ability of observers to see marked animals more easily than unmarked animals.

Our study on AISP provided a unique opportunity with added geographic closure because it was separated from other mountain ranges. This allowed us to have an added amount of control in our study which strengthened our inferences. Bighorn sheep populations often occur in geographically isolated areas and our remote camera methods would apply in such areas. However, it would be beneficial to repeat these methods in other areas to better understand their efficacy in a variety of locations. One of the critical elements that aided our success on AISP was having an in depth knowledge of which water sources were most utilized by bighorn sheep. This knowledge allowed us to optimize our sampling and photograph sufficient numbers of bighorn sheep while monitoring only 3 water sources. Researches using remote cameras to estimate abundance of ungulates should identify aggregation sites using GPS data or observations to optimize their sampling efforts.

Remote cameras are used for a variety of purposes related to wildlife ecology and conservation including observing behavior (Hall et al. 2013), discovering new species (Rovero et al. 2008), estimating species richness (Tobler et al. 2008), and estimating abundance (Perry et al. 2010). In this study we showed that it is possible to reduce sampling period when using remote cameras to estimate abundance, this is likely also the case for all the other applications where remote cameras are used for wildlife research. For example, measuring species richness has an inherent asymptotic relationship between detection effort (camera days) and the number of species detected. At some level of effort, not enough new species will be detected to justify additional sampling. This relationship will vary in different habitat types and in different wildlife communities. In order for researchers and ecologists to optimize their sampling effort with remote cameras, they will need to understand the asymptotic relationship for their area of interest and for their question of interest. Much work must be done in order to better understand these relationships for the many studies that are using remote cameras.

Precise estimates of abundance were obtained from 12 days of sampling with remote cameras, only a small portion of the complete 62 day sampling period. By understanding the relationship between sampling duration and information loss, researchers can save significant amounts of time and money by analyzing only the necessary portion of the images the collect. Our ability to collect images has already outpaced out ability to analyze them (Sheil et al. 2013), and remote cameras will only be used more frequently in the future (Meek et al. 2014, Burton et al. 2015). As more researchers use this advancing technology, it is important that time and money are not wasted on analysis that does not improve results. Indeed, there are discoveries in the fields of ecology and conservation that have not yet been made simply because researchers do not have the time to analyze all of their images collected by remote cameras. As we learn to better optimize the sampling design of studies using remote cameras, we will be better able to utilize their potential making them a more suitable alternative to traditional wildlife monitoring.

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Figure 2-1. Antelope Island State Park in northern Utah, USA, where we documented deployment time, proportion of photos analyzed, and loss of information using pictures of adult female bighorn sheep (*Ovis canadensis*) from 2012-2014.

Figure 2-2. Image of a group of bighorn sheep collected by a remote camera used in our study on Antelope Island State Park, Utah, USA. We classified each animal in the images we collected by their age and sex; we also identified collared bighorn sheep by the unique letter and number combination on their collar.

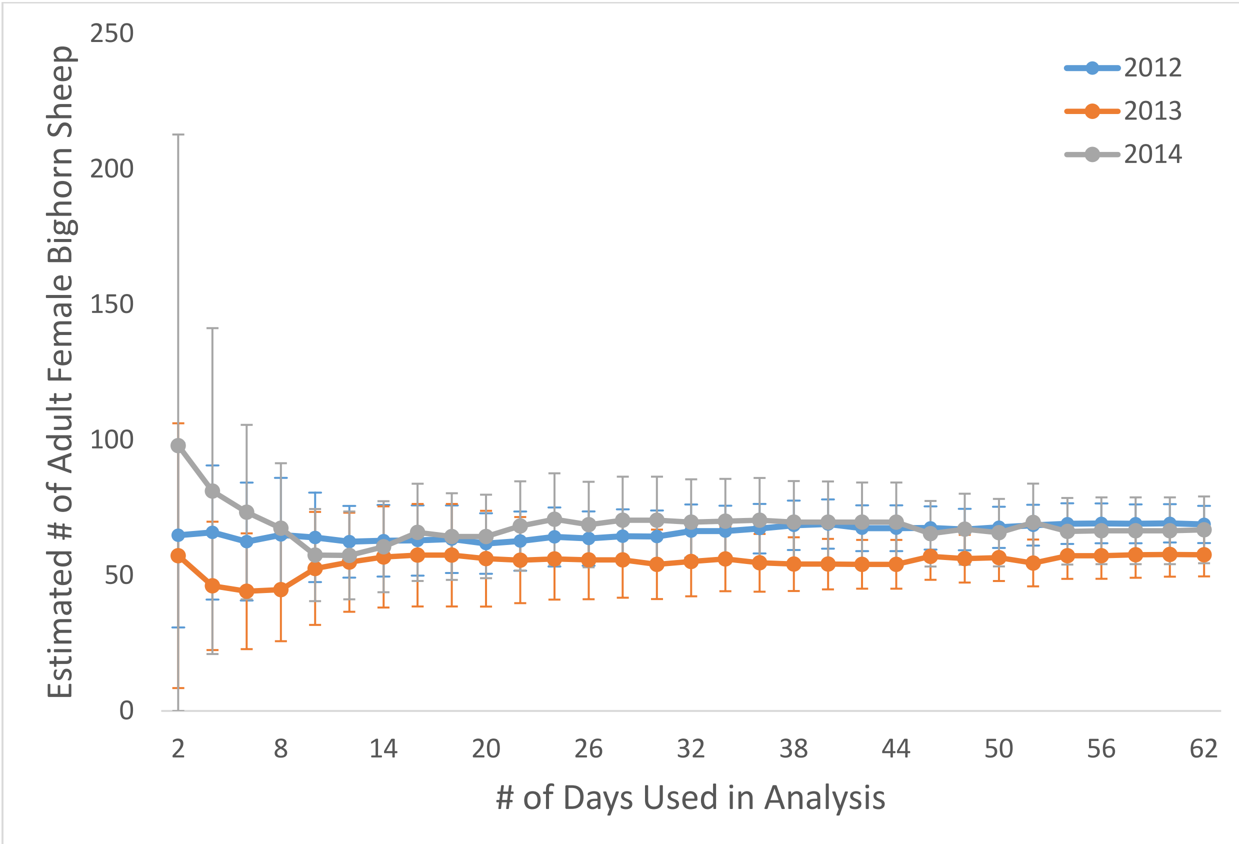

Figure 2-3. Estimated number of adult female bighorn sheep (*Ovis canadensis*) with 95% confidence intervals. Thirty-one replicated analyses were performed for each year 2012-2014; each replicate considered a different number of days (2-62) of sampling using remote cameras on Antelope Island State Park, Utah, USA.