Wild Furbearer Management and Conservation in North America

Edited by Tim L. Hiller, Roger D. Applegate, Robert D. Bluett, S. Nicki Frey, Eric M. Gese, and John F. Organ

Chapter 15: Survey and Monitoring Methods for Furbearers

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ACKNOWLEDGEMENTS

We thank the numerous entities and individuals that helped with the success of this project, including chapter authors and reviewers, financial supporters, in-kind supporters, and all others that contributed. We also thank Milan Novak, James A. Baker, Martyn E. Obbard, and Bruce Malloch, editors of the 1987 book by the same title. Their achievement set the bar very high.

This project was financially supported by Alberta Conservation Association, Alberta Trappers Association, Association of Fish and Wildlife Agencies, Colorado Trappers and Predator Hunters Association, Fur Takers of America, Government of Saskatchewan Ministry of Environment Fish and Wildlife Development Fund, Illinois Department of Natural Resources (State Furbearer Fund Grant Program), Iowa Trappers Association, National Wildlife Control Operators Association, North Carolina Trappers Association, U.S. Fish and Wildlife Service, Vermont Trappers Association, Wildlife Ecology Institute, Wisconsin Department of Natural Resources, Wisconsin Trappers Association, and Fred Fouse. We thank you for your financial support.

We are grateful for the in-kind support provided by Alan Sinner (Alan Sinner Photography; numerous wildlife images), and Jay Villemarette and Josh Villemarette (Skulls Unlimited International; images of skulls for each furbearing species). We also thank Tom Walker (illustrations of each furbearing species); Jamie McFadden (Wildlife Ecology Institute; construction and revision of distribution maps for each furbearing species based on available information and input from chapter authors); James Baker and Pierre Canac-Marquis (Fur Institute of Canada; furbearer harvest data from Canada); and Jeff Bowman, Martyn Obbard (Emeritus), and Peter Carter (Ontario Ministry of Natural Resources and Forestry) for providing background information and assistance associated with the 1987 book of the same title.

This project was funded by a Multistate Conservation Grant F19AP00097, a program funded from the Wildlife and Sport Fish Restoration Program, and jointly managed by the U.S. Fish and Wildlife Service and the Association of Fish and Wildlife Agencies.

This project has received educational grant funding support from the Illinois Department of Natural Resources - State Furbearer Fund. The Furbearer Fund provides grants to appropriate not-for-profit organizations, governmental entities, educational institutions, and corporations to benefit furbearing mammals and improve furbearer hunting and trapping opportunities.

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Citation:
Gese, E. M., P. A. Terletzky, H. S. Cooley, F. F. Knowlton, and R. C. Lonsinger. 2024. Survey and monitoring methods for furbearers. Pages 15.1–15.44 *in* T. L. Hiller, R. D. Applegate, R. D. Bluett, S. N. Frey, E. M. Gese, and J. F. Organ, editors. Wild furbearer management and conservation in North America. Wildlife Ecology Institute, Helena, Montana, USA. https://doi.org/10.59438/QXDE4827

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First edition published 2024
by Wildlife Ecology Institute
PO Box 4725, Helena, Montana 59604-4725, USA
web page: www.wildifeecology.org

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https://doi.org/10.59438/QXDE4827

Published 9 February 2024.

Front cover image of a Pacific marten (*Martes caurina*) at baited camera station with hair-snagging device designed for wolverines (*Gulo gulo*) in the Cascades of Oregon, USA; image courtesy of Tim L. Hiller, Wildlife Ecology Institute, Montana, USA,

Typesetting by Tim L. Hiller, Wildlife Ecology Institute, Helena, Montana, USA.

15

SURVEY AND MONITORING METHODS FOR FURBEARERS

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There is a continuing need to assess the status (distribution and population abundance) of furbearing species throughout North America for state, provincial, tribal, and federal agencies for effective management and conservation of furbearers. With an expanding population of humans and continued changes in land-use practices, loss and fragmentation of habitat, declines in prey populations, increases in disease transmission from domestic species, and increasing competition with other species, many fish and wildlife agencies have prioritized the management and conservation of some furbearing species. Paramount to making informed decisions regarding management of furbearer populations is accurate information about their current distribution and population status. Assessing the status of furbearer populations can be a daunting endeavor because many furbearing species are mobile, elusive, cryptic, and behaviorally responsive to many human-associated activities. Challenges increase for those furbearing species with relatively low population densities. Lastly, limited financial and staffing resources, restricted access to private lands, political and social considerations, and government regulations may further constrain the level of feasibility or acceptability of these efforts.

Prior to a biologist or manager implementing a monitoring plan, a diligent and thorough planning effort can improve efficacy by determining the: 1) population parameter(s) necessary to address the management or research need, 2) level of precision and accuracy needed for these parameter estimates, 3) level of assurance needed to acquire a reasonable and acceptable answer, and 4) appropriate time interval to repeat monitoring to produce updated information. Thus, requirements for planning include: 1) precise identification of the questions to be answered, 2) knowledge of biology and behaviors of the furbearing species of interest, 3) recognition of the physical and social environment in which the data are to be collected, 4) an understanding of threats or stressors to the furbearing species of interest, 5) development of suitable analytical processes that will provide some assurance surrounding the estimate(s) provided, and 6) selection of the appropriate survey method(s) to conduct in the field. All of these requirements may not apply to every monitoring situation, and some can be handled in a perfunctory manner. Other requirements, however, may be associated with considerable effort to arrive at some suitable compromise to ensure an appropriate estimate or answer will result from a survey. A pilot season or collection of preliminary data can provide much needed clarity toward developing a suitable monitoring program.

WHY MONITOR?

Monitoring the distribution, population size, population trend, or dynamic parameters (i.e., those that drive the state of a population; e.g., survival, reproduction) of a furbearer population is one of the prerequisites to making informed management decisions or formulating conservation plans with some level of certainty (Nichols and Williams 2006, Sauer and Knutson 2008). Establishing levels of sustainable harvest for populations of furbearing species requires knowledge of population status and temporal variation in population abundance (Jonzén et al. 2002, Haydon and Fryxell 2004). Knowing the legal requirements for species management is needed in many circumstances and monitoring furbearing species to ensure population persistence is fundamental to maintaining ecosystem function. Paramount to recovery, reintroduction, or development and evaluation of management plans and policies, is having reliable and accurate information regarding the status, health, and well-being of the population of interest (Gese 2001). Many state, provincial, tribal, and federal agencies are tasked with managing furbearing species at some level, and our ability to define success of management actions is important, but can prove nebulous. For example, determining whether investments in recovery of a rare species are paying off as expected (e.g., Batson et al. 2015); whether control of an invasive species or epizootic infections (e.g., rabies) is making

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a difference (e.g., Slate et al. 2005, Bos et al. 2020); and whether changes in the historical distribution and abundance of some furbearing species are having spill-over effects on the environment, other species, and humans (e.g., Hody and Kays 2018).

While a statistically robust sampling design for monitoring a species is typically desired, outcomes of the prescribed program do not always hinge upon science. Even when substantial funding is spent on a monitoring program, court cases and legal wrangling, social concerns, or political directives may still take precedence over biological opinion or fact. Addressing the status of a population of a furbearing species may be one of the most difficult tasks assigned to a biologist. A-priori knowledge of the status of a population will be useful in designing a monitoring program. For small populations, a census (a complete count of an entire population; Garton et al. 2005) may be considered, whereas large populations would require sampling via either relative or absolute abundance techniques, or estimation of alternative metrics (e.g., which may serve as a surrogate for abundance; MacKenzie and Nichols 2004). The level of precision and accuracy required to detect changes in rare or endangered species is much greater than for an abundant species. Errors and limitations of the confidence intervals assigned to estimates of population parameters would be more consequential for a rare species.

WHAT DO WE MONITOR?

Commonly asked questions associated with management and conservation of furbearing species include: 1) where are the animals (distribution or patterns of occurrence), 2) how is their distribution changing, 3) how many animals are there in a specific area (abundance), 4) what is the population trend (e.g., change in abundance), and 5) why is the distribution or abundance changing (i.e., population demography)? These questions often place biologists and managers in the difficult position of determining the status of a population. Biologists need a sound sampling design and reliable survey methods that provide accurate and precise data on the distribution (or occupancy), abundance, and trend of a species to make informed decisions and recommendations to policy makers (Thompson et al. 1998, Gese 2001, Williams et al. 2001). In some cases, it might be appropriate to monitor threats or stressors that could have delayed or secondary effects on the population (e.g., harvest levels, habitat loss, or recreational-use patterns). All too often, naïve biologists race into the field and start counting sign or marking or radio-collaring animals without carefully considering the objective or question they are attempting to answer.

As previously mentioned, careful consideration of what question is being asked, what population parameter(s) needs to be estimated to address the question, and what level of precision and accuracy is required of the estimate to answer the question(s) can improve the likelihood of success (Lancia et al. 1994, Zielinski and Stauffer 1996, Thompson et al. 1998). This aspect is particularly important depending on whether the monitoring effort is for a rare, moderately abundant, or abundant species. For species that are difficult to detect (as is commonly the case with furbearing species), occupancy-based monitoring that estimates the proportion of sites occupied by the species (MacKenzie et al. 2002), or changes in patterns of occurrence (MacKenzie et al. 2003), while accounting for imperfect detection, may be easier to accomplish than estimating abundance or trend in abundance of the species. With an appropriate sampling design, occupancy-based monitoring can serve as a surrogate for abundance (MacKenzie and Nichols 2004), and can be more robust than alternative indices of relative abundance (Lonsinger et al. 2016). In addition, cost comparisons of various monitoring techniques that measure relative and absolute abundance (e.g., Schauster et al. 2002), or different population parameters (e.g., occupancy vs. abundance vs. demography; Lonsinger et al. 2020) is also useful because a less expensive technique may provide the needed levels of accuracy and precision, or prove as accurate and precise as a more expensive technique.

QUESTIONS TO BE CONSIDERED

Identification of the specific question(s) to be answered is an important starting point for the planning exercise. Is detecting a species adequate? Or is it necessary to estimate detection rate and account for detection probabilities <1? In this context, we acknowledge that establishing the absence of any particular species usually can only be inferred from a lack of determining presence with a given amount of effort (i.e., zeros can be valid data points). For example, in the U.S. some management programs for prairie dogs (*Cynomys* spp.) require assurance that rare and endangered black-footed ferrets (*Mustela nigripes*) are not in the vicinity. A second question involves whether an estimate of a population parameter (e.g., occupancy, total population size) is sufficient or whether one needs to determine a trend in the population parameter. If the latter, repeated sampling is generally needed along with some knowledge of the basic seasonal or cyclic patterns of abundance for the species of interest. Is that particular species cyclic in distribution or abundance on a seasonal basis or over some other time frame? An issue frequently overlooked is whether a measure of the stock (breeding) population is desired or whether some other aspect of the population suffice. Because many furbearing species are synchronous seasonal breeders with population numbers fluctuating in cyclic seasonal patterns, this may be more than a casual element. Most members of the Family Canidae in the northern hemisphere breed in late winter or early spring. As a result, population indices during fall represent a conglomerate enumeration of the breeding population, surviving members of the most recent reproductive effort, and non-breeding adults.

When designing a monitoring program, biologists must not only make a multitude of decisions regarding funding, logistics, available personnel, the species in question, and the feasibility of meeting the study objectives, but they must also address a series of issues when setting up the sampling design (adapted from Garton et al. 2005). While in the planning stages of a monitoring program, key questions to address include: 1) what is the survey objective; 2) what is the frequency and duration of monitoring needed; 3) does the monitoring program need a population estimate, an index of abundance, or some other population metric (e.g., occupancy, survival); 4) if an index suffices, does the method chosen actually reflect or correlate with population size; 5) if a count is needed, how will you enumerate animals, and will complete or incomplete

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counts be satisfactory for monitoring; 6) what is the sample unit and population of interest; 7) how large of an area needs to be sampled; 8) will random, systematic, cluster, adaptive, sequential, or stratified sampling be used; 9) what levels of sensitivity, accuracy, and precision of their survey method are required to detect population change or trends; 10) what is an adequate sample size (e.g., number of animals, scats, sites, transects) for your survey method and can you attain that sample; 11) does the chosen study design and monitoring techniques actually answer the question(s) required for sound management decisions; 12) what are the repercussions to the population if the study design and methodologies used to collect data and formulate management decisions are imprecise, inaccurate, or biased; and 13) is the budget sufficient to accomplish the project?

We do not intend to provide a recipe for designing a monitoring program, as this will vary based on objectives. Rather, we describe various scenarios, pose questions, and raise issues that a biologist should consider and attempt to address prior to and during the design phase of a monitoring program. This chapter is not meant to include every study on every furbearing species using every monitoring technique, but rather provide the reader with a basic conceptual framework to design and implement a survey and monitoring program.

Biological Considerations

Even when a study design is carefully crafted and planned, the popular credence, scat happens, can make best laid plans go astray, dooming a fledgling monitoring program. Some of these events can be planned for through careful thought during the design phase and asking questions of other researchers or biologists working in similar environments or political-social-cultural landscapes. Again, many of these issues are intuitive to experienced scientists, whereas others may elicit a mental head slap when they occur to a naïve biologist, followed quickly by exclamations (e.g., why didn't I think of that, no one mentioned that before!), as a lot of hard-earned trust, sweat, money, and toil goes down the drain. In other words, talk to other biologists and ask lots of questions. Many failed monitoring efforts are never published. Investigating whether a new monitoring technique or sampling design has been attempted and failed can reduce headaches and frustration.

Animal Dispersion

The manner in which individuals of a species distribute themselves across a landscape in both time and space can be an important consideration while designing an inventory method. Some species with relatively large geographic distributions may have enough diversity in distribution patterns, threats, or other factors to warrant variation in monitoring across their distributions. Are they solitary or gregarious? Do they deposit or leave behind sign that is conspicuous and discernible? Does recognition of one individual influence the recognition of others (e.g., if they travel in groups or are solitary)? Is there interest in surveying individuals or groups of individuals? Are seasonal movements such as migrations or movements to areas with differing habitat conditions that are characteristic of the species, and do these characteristics apply equally to all sex and age classes? Is the species of interest territorial, which may cause them to be distributed in some regular pattern? If territorial, how large are the territories and how does this relate to the size of area in which you are trying to make an assessment?

Relative Responsiveness of Individuals

Does the survey method apply equally among sex, age, and any social hierarchy? To illustrate, in the early stages of developing inventory procedures for coyotes (*Canis latrans*), the use of elicited vocalizations to assess abundances was considered (F. Knowlton, National Wildlife Research Center, personal communication). In early trials, two issues were identified. First, there was a 4-fold difference in response rates resulting from the use of three different sirens used to elicit the vocalizations, which demonstrated that coyotes were likely to respond at times they were active, but unlikely to respond when they were inactive (Carley 1973). Subsequently, Wolfe (1974) reported that whereas dominant (alpha) individuals were likely to respond, transient individuals were less likely to respond. Petroelje et al. (2013) also reported higher response rates for coyotes that were residents compared to transients. In addition, a wide variety of environmental and observer characteristics influenced detection of responding vocalizations. These issues lead to serious concerns about the reliability of such procedures.

Linhart and Knowlton (1975) proposed and implemented a broad survey of relative abundance of predatory species throughout the western U.S.; their survey employed a series of artificial scent stations (>400 sampling points comprised of 50 stations each), and was directed primarily toward coyotes (Roughton and Sweeny 1979). In subsequent evaluations, it was revealed that coyotes were appreciably less likely to investigate and leave tracks at such scent stations when scent stations were encountered within familiar areas as opposed to when scent stations were encountered within less familiar areas (Windberg and Knowlton 1988, Windberg 1996, Harris and Knowlton 2001).

These situations indicate that trying to assess abundance by eliciting responses to some provocation may be subject to severe limitations. Sensitivity to such events may vary widely among furbearing species. For instance, we suspect that felids and mustelids may be much less sensitive to elicited responses than canids; felids may be attracted to visual lures (e.g., hanging feathers or aluminum foil) than olfactory lures (Ferreras et al. 2018), and solitary-living carnivores tend to have lower rates of long-distance vocalizations than group-living species (Suraci et al. 2017).

Using techniques that are directly associated with normal and natural behaviors of a species of interest is much less subject to the influences of unexpected factors. In addition, the activity or distribution of many furbearing species can be influenced by sympatric species. Consequently, results of monitoring may be confounded if the presence of dominant sympatric species or predators varies in space or time. Lonsinger et al. (2017) determined that detection rates of kit foxes (*Vulpes macrotis*) were higher in areas with higher levels of activity of coyotes. This also highlights the need to establish some method of validating the procedures used.

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Stereotypic Behaviors

For some furbearing species, especially when documenting presence is the primary objective, it is possible to take advantage of stereotypic behaviors. For instance, many felids seem to have an affinity for traveling within narrow canyons or along specific ridges, whereas many canids have a proclivity for traveling along trails and low-use roadways. Similarly, because the distribution of black-footed ferrets seems to be limited primarily to prairie-dog colonies, assessments can be geographically limited (Eads et al. 2011, Boulerice et al. 2024 [Chapter 49]). In addition, the nocturnal activity patterns of black-footed ferrest, coupled with the luminescent and reflective nature of their eyes and apparent innate curiosity, indicates the potential utility of spotlighting surveys within the specific areas of interest (Biggins et al. 2006).

Potential Wariness and Habituation

Many furbearing species have an innate curiosity toward novel situations in their environment (e.g., Windberg 1996, Harris and Knowlton 2001, Ferreras et al. 2018). As objects or situations become familiar through repeated exposures, the same stimuli tend to elicit lower degrees of interest and attention. Sometimes simply relocating the stimulus relatively small distances can temporarily revive interest. Sensitivity to such situations varies widely among and within furbearing species. For example, coyotes react strongly and warily (neophobia) to novel stimuli (Windberg and Knowlton 1988, Windberg 1996, Harris and Knowlton 2001), whereas bobcats (*Lynx rufus*) are much less reactive to novel situations and can be repeatedly captured in the same locations with the same attractants; trappers often hang a feather or reflective object as a visual attractant for bobcats (Ferreras et al. 2018).

Some furbearing species tend to be neophobic and wary of changes within their home ranges (e.g., coyotes; Windberg 1996). Harris and Knowlton (2001) reported that coyotes exposed to artificial scent stations were much more apt to visit those encountered at the periphery of their territories or when they were traveling in adjacent territories of other coyotes than when scent stations were encountered within the familiar confines of their own territory. Animals occupying areas of high use by humans often become habituated, or desensitized, to human presence than individuals of the species occupying remote areas with little or no presence of humans. Hence, knowledge of the general repertoires of and within a furbearing species can be important in selecting the means of monitoring.

Logistical Considerations

In addition to considering the biology and behavior of the furbearing species of interest, biologists and managers must also account for extrinsic factors that could influence their ability to successfully conduct a survey. These factors can be placed into broad categories of characteristics of the study area, timing of assessments, human and animal safety, human interference, potential for disease transmission, social and cultural values of humans within the study area and about the species, and regulatory requirements. Of course, available funding and other resources (e.g., vehicles, personnel) also need to be considered.

Characteristics of the Study Area

The physical attributes of the area being monitored, starting with spatial extent, topography, and environmental conditions, play an import role in determining which activities are feasible and practical. This starts with a clear designation of the area or areas to be monitored. It is essential to determine if complete enumeration (e.g., census) is feasible or if sampling will be required. A complete enumeration is rarely feasible or may be impractical.

Spatial extent of the area is an important aspect, but topographic and vegetative features should also be considered. The demarcation of the boundary of the study area to be assessed can be critical if an estimate of population density is part of the protocol, but increased use of spatially explicit capture-recapture (SECR) models to define the effective sampling area makes this less of a concern. SECR is a set of methods for modeling capture-recapture data collected with an array of detectors (Efford 2023). Detectors may be any device or survey capable of uniquely identifying individuals, including cage or box traps, noninvasive genetic sampling (NGS; e.g., scat surveys, hair snags), or remote cameras (e.g., for individuals that can be identified from natural or artificial marks). SECR methods use spatially disparate detections of individuals to address capture heterogeneity associated with proximity to detectors and to estimate the effective sampling areas, reducing the problem of edge effects common in conventional capture-recapture estimation of population size (Efford 2023).

Reviewing what is known about the area of interest may help identify potential issues with access for conducting the planned surveys. Will private lands be included, and if so, will access be a potential problem? If public lands are included, are any special permits required? Are roads adequate to provide access to all portions of the area, or are some portions impassable during some seasons or only accessible by foot? Some furbearing species occupy areas with rugged terrain, dense vegetation, extreme weather conditions (e.g., Arctic, desert), few or no roads, or high elevations.

The influence of extreme temperatures on the performance of personnel and equipment should be considered during the planning stages. Difficult access to locations to conduct surveys or capture animals may make a well-planned study irrelevant. If aerial-based monitoring is involved, having experienced pilots familiar with the wind currents or sudden storms in mountainous terrain cannot only increase the confidence in the data collected, but more importantly, get everyone home safely. Even innocent-looking snow cover in the morning can turn avalanche prone as weather conditions change during the day. Sometimes using the terrain to our advantage is practical. For example, placing remote cameras in places where animal movements are concentrated along a specific portion of trail or through a mountain pass, may increase success of sampling efforts. Stratifying efforts for track or scat sampling to be more intensive along trails commonly traveled by the species of interest may increase monitoring efficacy. Scrapes (e.g., ground scratches left by mountain lions [*Puma concolor*] or coyotes) and scent marks made by a species on prominent features on the landscape can serve as an index of relative abundance.

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The spatial extent of sampling must always be considered given that time and funding usually prohibit sampling the entire area (Lancia et al. 1994, Macdonald et al. 1998). Are we sampling the population of interest, or are we trying to conduct a complete census, which is rarely practical or possible? In addition, the costs, logistics, personnel, and time constraints must be considered in deciding the utility of a specific method to monitor a population. Because increasing the sample size generally reduces bias and increases accuracy and precision, multiple small-sample units may be more useful for statistical comparisons. Thus, a biologist must weigh the advantages of multiple small sample areas versus a few large sample areas.

The effect of the spatial distribution and movement capabilities of the species of interest will influence the size of the sampling area, with wide-ranging species requiring larger sampling extents than species with more limited movements and smaller home-range sizes. Stratification of a study area may also be useful to concentrate monitoring efforts and reduce costs. Sampling strategies may vary for rare species compared to common species. For example, when estimating patterns of occurrence for a common species, optimal sampling design may involve surveying fewer sites more frequently; in contrast, surveying for rare species may be better accomplished by surveying more sites less intensively (MacKenzie and Royle 2005).

Timing of Surveys

Identifying when to conduct surveys will determine not only the merits of the information obtained, but also the inferences that can be made from the data. Does the activity or visibility of the animals change seasonally? How long will it take to sample the population and will the population likely be closed to changes (e.g., abundance, occupancy) during the sampling period? What stages in the seasonal pattern of population phenology might be involved? Among species with seasonal breeding patterns, characterizing the breeding population may be more important than making assessments at times that will include young of the year. In instances where some measure of reproductive performance is desired, conducting the assessments at a different time may be necessary, provided young animals can be differentiated from adults.

Activity periods of many species are influenced by the prevailing weather (e.g., striped skunks [*Mephitis mephitis*], red foxes [*Vulpes vulpes*], northern raccoons [*Procyon lotor*]; Ruzicka and Conover 2011) or lunar patterns (e.g., coyotes; Bender and Bayne 1996). These aspects become increasingly important when trying to assess population trends, such that conditions associated with surveys should be standardized or explicitly accounted for in the analysis to reduce errors and biases in the data. For example, in an attempt to determine the seasonal relative abundance of black-tailed jackrabbits (*Lepus californicus*) in Texas, USA, roadside-based counts commenced at sunrise. Subsequently, Haug (1969) determined that jackrabbits changed activity periods seasonally not only by extending their activity periods into the daylight period during summer, but accentuated their activity by being active for longer periods of time during summer.

Coyotes typically vocalize when they are active, but may adjust their circadian activity to ambient light or temperature conditions (e.g., Hidalgo-Mihart et al. 2009, Melville et al. 2020). Alternatively, inclement weather can influence the ability of an observer to detect animals or identify their sign. Many species are more easily detected with snow cover, especially where deciduous vegetation is involved. This is especially true in cases where aerial-based surveys are used to count individuals. Also, wind or even light rain can easily degrade animal tracks. Standardizing conditions appropriate for surveying will help work around such issues and improve the utility of the information obtained.

Human and Animal Safety

A high priority of any monitoring program is the safety and well-being of participants, followed by animal safety and welfare (see Kreeger 2023 [Chapter 17]). Assessing the risks that animals, terrain, or methods of monitoring pose to ourselves and others assisting the research should be examined and carefully outlined in safety protocols, even if collecting only animal carcasses, which might have disease causing pathogens transmittable to humans (see Gillin et al. 2024 [Chapter 7]). Reviewing protocols frequently and training new participants on procedures can help maximize safety, particularly when using methods such as aerial-based capture or telemetry, use of chemical immobilization, handling a species that poses direct or indirect (e.g., disease) risk to handlers, or movement through hazardous terrain. Advising participants on proper gear for extreme temperatures and maintaining hydration and nutrition may seem trivial, but success of the study may depend on the attitude, safety, and comfort of those collecting data. Do not assume that every participant is equally adept to changing environmental conditions and has sufficient levels of knowledge and experience to avoid getting into trouble. This may not only endanger themselves, but could consequently put the animals at risk as well.

Influences of Surveys on Animal Behavior

A question that needs to be addressed is whether the animal's behavior or fitness is being influenced by the survey method. If the monitoring program requires the capture and marking of animals, there is an obligation and responsibility to treat the animals with respect and minimize harm or impacts on them. Animals can habituate or become averse to monitoring techniques, thereby biasing results. The act of ear tagging and radio-collaring an animal may seem harmless, but could increase the visibility or vulnerability of that animal to legal or illegal harvest, or predation, thereby reducing fitness and biasing estimates of survival. Although ecotourism may offer some benefits to conservation, the public is often eager to see a rare or endangered species, which may in fact cause undue disturbance and stress to the animal (Buckley et al. 2016).

Transmission of Pathogens

A subject often overlooked when monitoring furbearing species relates to the role of infectious disease-causing pathogens in monitoring programs. The increase in intensity of interactions among furbearing species, humans, pets, and livestock escalates the possibility of disease transmission (e.g., Fayer et al. 1982,

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Rupprecht et al. 1995, Marcek et al. 2023). There are serious consequences for rare or endangered species exposed to disease agents. Importing outside or exotic diseases, changing pathways of transmission, adding stress to animals during capture, and even bringing domestic pets to the study site are issues to be considered when initiating a monitoring program. In one instance, introduction of canine distemper virus caused a rapid population decline of black-footed ferrets, and almost caused their extinction (Williams et al. 1988; see also Boulerice et al. 2024 [Chapter 49]).

Prior to initiating a study, the possible need for a disease-monitoring program and handling protocol (for animals and samples collected), or a biosecurity protocol, should be addressed (see Gillin et al. 2024 [Chapter 7]). Physical examination of living animals, blood collection for serological analysis, and post-mortem examinations of animals (e.g., collected from trappers or recovery of radio-marked animals) can be used in a disease-monitoring program. Consultations with veterinarians affiliated with a diagnostic laboratory or university can help identify which diseases should be screened for and then design an appropriate monitoring program. Disease transmission to biologists or other participants (e.g., volunteers) must also be considered and proper protocols established for handling samples, even during post-mortem examination of animals.

Social and Cultural Values

The ability to conduct a monitoring program may be severely curtailed if the social, political, or cultural environment prohibits the presence of personnel or equipment. Remote cameras used to monitor the presence of jaguars (*Panthera onca*) along the southern border of the U.S. are frequently vandalized, destroyed, or stolen (Culver 2016). Areas with poor economies may resent people driving expensive vehicles and displaying excessive wealth via equipment and clothing. Animosity towards agency personnel can occur frequently in areas with intense anti-government sentiments.

Cultural mystics and social taboos related to animals should be respected, particularly where locals retain religious or cultural ties to the species being monitored (Hiller and Vantassel 2022). However, conducting research and monitoring in these environments can be richly rewarding in terms of the cultural experience. Traditional ecological knowledge can be gained in discussions with members of the local community. Making appropriate political and social connections within the community is a prerequisite for gaining trust and lessening troubles along the way. Further considerations include working with local communities to ensure research efforts do not interfere with subsistence activities, particularly for shaping values and attitudes towards the species of interest, as well as public acceptance of the monitoring program. Returning to these communities after the study is completed and discussing the results with the local community and regional officials may pay dividends on influencing future management and policy.

Regulatory Requirements

Obtaining the necessary permits to conduct surveys can be a time-consuming process and is an important consideration early in the planning process. In advance of any study, gaining permission and approval (and potentially, partnership) from agencies involved in the area is paramount. Often times, regulatory agencies must be consulted and involved in the study design to help ensure it supports their statutory obligations for management. Terms and methodologies need to be clearly defined in protocols and proposals to avoid confusion later and potential roadblocks to the survey plan. The amount of detail required by agencies in the permitting process varies widely; some agencies require a relatively simple application, whereas other agencies may require a lengthy and detailed proposal. Monitoring programs involving international shipment of samples (e.g., genetic, serological) to diagnostic labs will often need to secure proper permits for exporting or importing samples (e.g., international trade in species such as bobcats and North American river otters [*Lontra canadensis*] are regulated under the provisions of the Convention on International Trade in Endangered Species of Wild Fauna and Flora; see Hiller et al. 2023 [Chapter 10] for more details).

During the past 40 years, the prevalence of animal care and use committees (ACUC) has increased at universities; research facilities; non-governmental organizations; and state, provincial, and federal agencies. Gaining approval for capturing, handling, and monitoring activities is now required by most ACUCs, with the desire to see humane and ethical methodologies and treatment of animals becoming prerequisite for sound science (Paul et al. 2016). Acquiring the necessary permits to conduct surveys is also necessary and often entails concurrent ACUC approval. Having standard operating procedures for common procedures can increase efficiency if shared by researchers within and among organizations.

The ACUC of the American Society of Mammalogists publishes guidelines for the use of wild mammals in research that can help guide appropriate handling procedures (Sikes et al. 2016). If the monitoring program will involve capturing and handling furbearers, then completion of a handling and immobilization course from a qualified veterinarian should be considered, with the knowledge that documentation of training may facilitate approval by ACUC. Additionally, if chemical immobilization will be used, knowledge of and adherence to all legal requirements for permits (e.g., Drug Enforcement Administration within the U.S. Department of Justice), and obtaining, storing, and using drugs is required (see Kreeger 2023 [Chapter 17]).

Statistical Considerations

A key consideration in wildlife research is that the vast majority of studies are observational rather than experimental because wildlife researchers typically do not manipulate systems in the experimental sense, nor can they typically integrate all three of the cornerstones (controls, replication, and random assignment of treatments) associated with experimentation (Shaffer and Johnson 2010). Throughout this section, we describe approaches used by wildlife researchers to help avoid the pitfalls associated with observational science to help ensure valid results that support informed management decisions.

A monitoring technique is only as useful as a tool for estimating the state or trend of a population as the design behind it. A poor sampling design will ultimately fail to answer the primary questions

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posed regarding the population of interest. In this next section, we provide some groundwork to be considered during the design of a monitoring program (Williams et al. 2001, Gregory et al. 2004, Garton et al. 2005). Many of these considerations seem quite intuitive and fundamental, but success may hinge on careful examination, planning, and development of an appropriate, unbiased, and repeatable study design (Skalski and Robson 1992, Macdonald et al. 1998, Thompson et al. 1998). Statistical assumptions associated with each method should be considered before implementing a monitoring program (Peterman 1990, Hayes and Steidl 1997, Van Strien et al. 1997). There are several books (e.g., Macdonald et al. 1998, Thompson et al. 1998, Williams et al. 2001) that provide thorough reviews of statistical considerations.

Frequency and Duration of Monitoring

Decisions regarding frequency and duration of monitoring should appropriately reflect the biology and (potentially) threats associated with the species of interest. If assessing a population trend is desired, researchers should consider life history of the species, such as age of sexual maturity (if reproducing individuals are being monitored) and ability to detect change in abundance over time. If researchers are interested in assessing the response of a population to threats or stressors, such as habitat loss, disease, or management regulations, information on the timing and nature of threats must be considered.

For short-lived species, a shorter duration of monitoring will be adequate; but for long-lived species, a longer duration may be required to detect changes in population trends, habitat, or threats. Factors that support an extended monitoring duration will often justify a multi-year monitoring interval (e.g., monitoring every 2–3 years rather than every year). Frequency of surveys may also balance cost factors with desired levels of precision, ability to detect trends in abundance with a specified probability, and limitations on the probability of incorrectly estimating the size of the population relative to management goals. A change in monitoring methods may also be necessary over time. For example, status of occupancy may be the most appropriate method to use initially after reintroducing a species, but more rigorous methods for estimating population size may be needed following species recovery.

Defining the Sample Unit

Before beginning a study, it is important to clearly identify the statistical population of interest (i.e., the collection of units over some defined region). The statistical population is not always the same as the biological population, so careful consideration should be given to the relationship between these two populations (Krebs 1999). For example, if the biological population is much larger than the population actually studied, the biologist must extrapolate to draw inferences, which can lead to invalid conclusions. This problem can be avoided by sampling the population from which you would like to draw inferences (Krebs 1999). If it is desired to draw inferences from the larger biological population, then the larger population needs to be sampled, or available to sample, when sampling units are selected. The spatial and temporal heterogeneity of the animals must also be considered with regard to how that influences the distribution of your samples and data collection (Williams et al. 2001).

Limited time and budgets usually preclude making observations on the entire population (i.e., a census). Instead, observations are made on a smaller segment of the population (i.e., a sample), and conclusions or estimates concerning the entire population are derived on the basis of the sample (Williams et al. 2001, Garton et al. 2005). Hence, defining and measuring the sample are crucial to ensuring valid conclusions. The sample unit, or the experimental unit, will depend on the objectives of the study, and should be the unit of statistical analysis.

Sample units should be independent, unbiased, and randomly chosen, whenever possible, to help ensure validity of results. However, defining the sample unit can be difficult. For estimating population abundance or demographics, the experimental unit may be an individual animal, a group of animals, or all animals within the boundaries of a geographic area. In contrast, for estimating occupancy parameters, the experimental unit may be a discrete habitat patch or a section (or cell) within contiguous habitat, and the spatial scale of a unit can vary based on the ecology of the species of interest. Discussions of the types of experimental units and their appropriateness have been included in other sources (e.g., Dean and Voss 1999, Keppel 1991, Wu and Hamada 2000).

Independence of Samples

Statistical tests usually require independence of observations, whether those observations are survey points, transects, or animal locations. Independence requires that a completely separate sampling effort is conducted within each group and selection of units in one group has no effect on the units selected in any other group (Garton et al. 2005). However, animal locations and animals are usually spatially and temporally correlated, and thus may occur as non-independent data. For example, the current location of an animal is correlated with its recent location, so if two observations are recorded within a very brief period of time, they are likely dependent (spatially and temporally autocorrelated). As the amount of time between locations increases, locations approach independence. White and Garrott (1990) suggested independence between locations is assumed if enough time has passed for the animal to have moved across its home range. More recent developments in home-range analysis have allowed the use of a set of alternative estimators (e.g., autocorrelated kernel density estimation) designed to be statistically efficient for addressing the complexities of complex movement data now acquired from global positioning system (GPS) collars (Fleming et al. 2015, Silva et al. 2022).

Animals living in groups or with littermates could also be considered non-independent because behavior of an individual in a group may be influenced by individuals within that group (Hurlbert 1984). Thus, treating individuals in groups as independent (pseudoreplication) often produces artificially inflated estimates of precision, increasing the chance of concluding samples are statistically different when they are not (Type I error; Garton et al. 2005). These consequences can be avoided by properly defining the experimental unit, having independent samples, and applying a sampling design that uses randomization, replication, and controls for variation (Williams et al. 2001, Garton et al. 2005). Accounting for spatial influences on capture probabilities can also be considered (e.g., Royle et al. 2011).

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Sample Size and Power

After the sampling unit is defined, the next step is to determine the sample size necessary to achieve study objectives. Sample size refers to the number of independent, randomly sampled units collected from a population, and sample size may be the number of replicates within each experimental treatment (Williams et al. 2001, Garton et al. 2005). Typically, larger sample sizes give more precise estimates and provide increased sensitivity to detect change.

The precision, or reliability, of the estimate can be measured by constructing confidence or credible intervals around the estimates. A confidence or credible interval is a range of values based on the mean of the estimate plus and minus the variation of that estimate, and which is expected to include the true population value within a given probability if the assumptions are met (Garton et al. 2005). Intervals can be constructed for any probability, but 95% has been somewhat arbitrarily selected as the most commonly used. Because the entire population is not sampled, it is unknown whether the true size of the population falls within the confidence interval, only that on average 95% of the confidence interval will include the true population value. A narrow confidence interval indicates a high level of precision, likely resulting from a large sample size, whereas a wide confidence interval indicates a low level of precision, likely due to a small sample size.

Power is the probability of rejecting the null hypothesis (e.g., no difference between populations or over time) when it is in fact false and should be rejected (Williams et al. 2001). Power analyses can be used to quantify the sample size necessary to achieve a desired level of precision under a particular sampling design, and to quantify the level of sensitivity necessary to detect change to answer study questions (Steidl et al. 1997, Mills 2007). Thus, conducting a power analysis before initiating data collection can help a biologist determine how much sampling effort may be necessary. The power of a monitoring program is influenced by many factors (Gerrodette 1987), including count error and variability, sample size, survey length, magnitude of trend to be detected (i.e., the effect size), and the statistical level of significance (i.e., α, the a-priori probability of erroneously rejecting the null hypothesis when it is in fact true, or Type I error).

Power is positively related to α, sample size, and effect size (e.g., differences among treatments, slope of trend lines), and negatively related to variation. Thus, small sample sizes, effect sizes, Type I error rates, or large variances yield low power, which may prevent detecting a difference that is real (Garton et al. 2005). This can be a problem with studies of furbearing species, which often have inherently small sample sizes. Steidl et al. (1997) and Ellis (2010) provide useful reviews of statistical power analysis in research designs. In terms of retrospective analysis, power analysis can be useful to estimate the number of samples or effect size that would have been necessary to reject the null hypothesis with greater certainty.

Sample Bias, Accuracy, and Precision

Furbearer research often requires long periods of field work to collect data on a limited number of individuals (e.g., resulting from species that are difficult to detect, occur in low population abundance or density, occupy remote environments, or for which we have little ecological knowledge), stressing the importance of careful consideration of the sampling design to achieve success. Knowing how to sample and how to design a research project that provides unambiguous results are crucial to scientific advancement. Sampling is the technique of drawing a subset of sampling units from the complete set and then making deductions about the whole from the part. It is consistently used in wildlife research and management, but often incorrectly (Sinclair et al. 2004). If we are to make deductions about the population from a sample, the sample must be random and an accurate representation of the population, both to the extent possible, with some amount or pattern of variability (Williams et al. 2001, Garton et al. 2005). If we are to draw conclusions about a population based on samples, every attempt must be made to use samples that are randomly collected and representative of the population of interest, which can be challenging given that we are conducting observational, not experimental, science.

Accuracy is a measure of how close an estimate tends to be to the true value (e.g., population size or occupancy; Garton et al. 2005). Bias is the difference between the expected value of an estimator and the true value (Sinclair et al. 2004, Garton et al. 2005). Precision is the variation in estimates obtained from repeated samples (Garton et al. 2005). A sampling design may provide precise estimates that are not accurate. It is also true that a sampling design may provide accurate estimates that are not precise.

Sampling strategies should be designed to increase levels of both accuracy and precision, yet reduce bias. Precision is achieved primarily by considering a large sample that is a randomly chosen set of independent samples with replication. In addition, the degree of variation among replicate samples can be reduced through standardized protocols of the survey technique. Standardization of monitoring methods can be equally useful when comparing across years and different studies. All too often there is the difficulty of comparing results amongst years and studies due to the lack of standardized monitoring protocols; are differences in the results truly due to the population, or simply due to different methods being used?

Probability of Detection

An issue that must be addressed when developing a monitoring program is the detectability, observability, sightability, or catchability of the animal. Low values of detectability, observability, sightability, or catchability for a species will result in small sample sizes and wide confidence intervals around the population estimate, likely making the estimate imprecise and inaccurate. The probability of detection for an individual or species may be considered complete, less than complete but constant, or variable, but is generally <1 (Williams et al. 2001, MacKenzie et al. 2002). Many techniques, such as population estimates from mark-recapture approaches, may assume an equal probability of capture and recapture (or resight). However, this assumption may be violated depending upon the furbearing species and the monitoring technique. For example, the use of foothold traps may

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cause an aversion among some furbearing species (e.g., coyotes) to being trapped or recaptured again, whereas for other furbearing species (e.g., North American river otters), recapture with traps is effective. Therefore, if animals are directly captured, then a second, noninvasive technique (e.g., scat sampling, hair snares, remote cameras) may prove effective and unbiased for recapture or resight of that animal. However, even some noninvasive techniques may be avoided by certain segments of the population. For example, dominant coyotes were more likely to avoid remote cameras compared to subordinate coyotes (Larrucea et al. 2007). Similarly, Murphy et al. (2018) determined that hair snares and scat sampling disproportionately detected different population cohorts in a population of coyotes.

Madsen et al. (2020) provided evidence that weather conditions influenced detection probabilities of coyotes during surveys that utilized remote cameras. One should also consider detectability or catchability in terms of the location of the capture device (e.g., Windberg and Knowlton 1990, Windberg 1996), seasonal changes in the behavior of the animal, and trap-shy or trap-happy animals that either bias population estimates or present an inaccurate trend in measures of abundance or other parameters. For harvested furbearers, harvest can be incorporated as a recapture event following initial capture through invasive (e.g., live-capture) or noninvasive (e.g., NGS) approaches. For example, Dreher et al. (2007) used hair snares and genetic identification of samples to capture individual black bears (*Ursus americanus*) and used genetic samples from harvested bears as the recapture event. Using alternative techniques for capture and recapture reduced the influence of individual capture heterogeneity associated with a single capture method (Dreher et al. 2007).

Spatial and Temporal Scales

The behavior of furbearing species is strongly influenced by the spatial and temporal variability of environmental conditions (e.g., habitat conditions, amount of snowfall, ruggedness of terrain) across the landscape. Determining the spatial and temporal scales appropriate to address the study objectives can help account for these environmental patterns. For example, habitat selection may need to be sampled at micro-scale and macro-scale to understand the phenomenon fully (Johnson 1980).

Just as with spatial scale, the appropriate choice of temporal scale depends on the question being asked (e.g., open or closed population) and the species of interest. For example, a relatively short sampling period and study duration may be appropriate for certain behavioral responses (e.g., determining reproductive success via the presence of wolf pups at a rendezvous site), or be required to meet modeling assumptions of population closure (e.g., for estimating abundance or occupancy). In contrast, a longer study duration may be necessary to assess population dynamics or estimate parameters in a large population, which may require a multi-year effort if sampling varies among years. An even longer time scale may be required for studies of genetic change and evolution (Wiens et al. 1986). If threats or stressors are causing a population decline for a species, researchers might also consider the scale at which those threats or stressors are occurring when they select an appropriate monitoring scale.

Failing to consider the effects of scale, and instead sampling at an arbitrary scale, may create problems in interpreting the results (Addicott et al. 1987). Comparisons among different systems may become difficult or impossible because arbitrary study units may represent different scales. For example, estimates of occupancy are intrinsically associated with the scale of the sampling units and spatial extent of sampling, such that estimates from studies employing disparate scales (even within the same system) cannot be directly compared (Lonsinger et al. 2020). Additionally, a given study area may not correspond to the scale appropriate to examining a particular question or for informing decisions for harvest management. Therefore, it will be difficult to relate field research to specific models or questions, and conclusions could be flawed. Finally, different demographic processes in the same system may operate at different scales, so it may not be sufficient or adequate to examine only one scale.

Biological vs. Statistical Significance

Statistical hypotheses are widely used because they provide objective, standardized criteria for decision-making. However, the method has received much criticism during the past couple decades (e.g., Johnson 1999, 2002; Anderson et al. 2000, Ellison 2004, Guthery 2008). Null-hypothesis testing is uninformative in some cases (Johnson 1999), and often results in conclusions that may lack meaningful insights for conservation, planning, management, or further research (Guthery 2008). Additionally, the significance level (α) used in a test is often based on convention (e.g., 0.1 or 0.05), classifying results into biologically meaningless categories (significant and nonsignificant; Anderson et al. 2000). There may be times when faced with a test statistic of $0.05<P<0.10$, but one may decide the result is biologically meaningful, or the result is indicative of a relationship. Finally, because the P-value is dependent on sample size, one can always reject the null hypothesis, even if the true difference is trivial, if the sample size is large enough.

Anderson et al. (2000) suggested that information-theoretic methods offer a more useful, general approach than null-hypothesis testing. This technique uses values such as the Akaike Information Criterion (AIC), Bayesian Information Criterion (BIC), and others to identify a set of models supported by the data from a set of a-priori candidate models, which represent plausible competing hypotheses. Inferences are based on the set of biologically realistic models supported by the data, rather than on a single best-performing model. As with other modeling approaches, information-theoretic methods such as AIC and BIC should be carefully designed and interpreted to avoid misuse (see Arnold 2010).

Sometimes descriptive statistics provide adequate information, and with the added benefit of being relatively straightforward to interpret or compare to other studies. Simply looking at the data can prove invaluable when first attempting to realize how covariates may be influencing the response variable. Simple box plots and regressions can point to meaningful relationships within the data. We again emphasize that the initial study design is critical, and quantitative analyses are generally useful and pertinent to any

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study, but conclusions should be based upon biological reality. With the large amount of variability in most study systems and general lack of controls or replication (i.e., observational studies), statistical tests producing *P*-values <0.10 should be examined thoroughly, as a biological relationship may exist among the parameters measured and sample size may be limiting a definitive statistical conclusion.

SAMPLING STRATEGIES

One is seldom certain that any sample is unbiased; we can only increase the chances of obtaining an unbiased sample by the way we select the units (e.g., individuals) or measurements that comprise it (Williams et al. 2001, Gregory et al. 2004, Garton et al. 2005). Usually, this involves selecting the independent sample randomly, so the sample is representative of the population. Random sampling designs are intended to minimize possible biases from the observer, which otherwise could occur in field studies (Schefler1980). Obtaining a random sample is not always simple or possible. For example, how does one trap a random sample of animals that exist at very low population densities and are highly mobile? Are the captured individuals somehow different from the other individuals within the population? Does the sampling strategy require a level of experimental control (difficult or impossible to achieve in a field study)? Complete randomness may never be achieved, but can be thought of as an ideal toward which the investigator strives. Next, we provide a summary of common sampling designs. These may apply to any sample unit, whether they are plots, transects, points, roads, individual animals, or groups of animals. For a thorough review of sampling design, see Krebs (1999), Williams et al. (2001), or Garton et al. (2005).

Simple Random Sampling

Simple random sampling is most appropriate when you are studying a homogeneous population. A group of observations or individuals (a sample) for study is selected from a larger group (a population) such that each individual is chosen entirely by chance and each member of the population has an equal chance of being selected (Krebs 1999, Williams et al. 2001, Garton et al. 2005). There are a number of ways that sampling units can be selected randomly, including the use of random number generators via computer, scientific calculators, database packages, or random-number tables in most statistics books (Gregory et al. 2004). Sample plots within the study area could be located by randomly selecting geographic coordinates. Truly random samples may occasionally produce biased estimates by chance due to unsatisfactory spatial coverage of the area or population of interest (Garton et al. 2005), particularly if the sample units, survey plots, or land-cover types are small and patchy in distribution. Proportional stratified sampling may be more appropriate in cases to ensure all land-cover types are sampled (see section on Stratified Random Sampling).

Stratified Random Sampling

Stratified random sampling allows the researcher to depart from the guidelines of random sampling by parsing, or stratifying, the area or population of interest into smaller, exclusive subunits with similar characteristics (Krebs 1999, Williams et al. 2001, Garton et al. 2005). Stratified sampling is generally used when the population is heterogeneous and certain homogeneous subpopulations (strata) can be isolated and identified. These strata may be land-cover types, age classes, or sex classes, provided these strata have similar characteristics and the substrata themselves differ from each other in the characteristic of interest (Garton et al. 2005). The principles of simple random sampling are then used to draw samples within each stratum separately. Stratified sampling takes advantage of prior knowledge about a species or area to sample more effectively than simple random sampling. For example, if a species is known to select certain land-cover types within the available set, stratifying the study area by land-cover type may be most appropriate. In addition to land-cover type, stratifications could be by elevation, population density, accessibility of survey sites, administrative boundaries, or any other variable likely causing variation among populations or areas.

Delineating strata that minimize the variation between or among sampling units within a stratum and maximize the variation between or among strata will increase the precision of estimates (Gregory et al. 2004). The number of strata required differs according to the sampling situation, but should be between 2 and 6 (Cochran 1977, Krebs 1999). Because a point of diminishing returns is quickly reached, the number of strata should normally not exceed 6 (Cochran 1977), with often fewer strata being desirable, but this will depend on the strength of the gradient dividing the strata (Krebs 1999). Once strata are identified, sample units can be allocated to each stratum, commonly assigning a proportion of sample units to each stratum according to the proportion of the population of a species occurring in each stratum (Sutherland 2000); however, alternative sampling-allocation methods exist (Snedecor and Cochran 1980).

Common reasons for using stratified random sampling rather than simple random sampling include: 1) estimates of the population parameters may be required separately for each subpopulation, 2) sampling challenges may vary in different areas (e.g., population density, terrain), 3) stratification may increase precision of estimates when strata are chosen well, and 4) convenience of sampling may be increased and cost may be reduced (Cochran 1977, Krebs 1999). For analyses (e.g., occupancy modeling) assuming the use of simple random sampling during data collection, strata should be analyzed separately or, if combined into a single analysis, accounted for through the inclusion of covariates to represent each stratum (MacKenzie et al. 2018).

Systematic Sampling

Systematic sampling is the process of selecting the first sampling unit at random from the set, and thereafter selecting sampling units at a predetermined regular interval as they are encountered (Garton et al. 2005). A valid application of systematic sampling requires the random placement of the first plot followed by a systematic placement of subsequent plots, usually spaced along a transect or in a grid pattern (Garton et al. 2005). For example, traps placed on a line or square grid at 500-m (1,640 ft) intervals after the randomly determined location of the first plot. This method is often

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used because of its simple application in the field, and its ability to sample evenly across an area. Systematic sampling is not the same as random sampling, but it generally produces an unbiased sample, and is an acceptable sampling strategy. Beware, however, that periodic effects (i.e., populations with regular or repeating cycles) may bias the estimates, as there may be some unknown pattern within the population (Krebs 1999, Garton et al. 2005).

Cluster Sampling

Cluster sampling is simple random sampling where each sample unit is a cluster or collection of observations (Williams et al. 2001, Garton et al. 2005). The structure of many furbearing species includes social or family groups, thus this method of sampling has application to several species. Cluster sampling increases efficiency (reduced costs and time), particularly in areas where the furbearing species of interest is territorial and has large home ranges. Garton et al. (2005) stated the three-step procedure for cluster sampling consists of: 1) identifying the appropriate clusters and listing all clusters, 2) drawing a random sample of all the clusters (this may vary by group size), and 3) measuring all elements or parameters of interest within each randomly chosen cluster. They recommended that if sample units in a cluster are similar (i.e., little variation within a cluster), cluster size should be small. If units within a cluster are heterogeneous (i.e., high variability within a cluster), then cluster size should be large. If groups of animals are the sample unit, then group size is not under control of the researcher and is a characteristic of the population (Garton et al. 2005).

Adaptive Sampling

Because of the difficulty in detecting and estimating the population abundance or distribution of rare species and species with elusive behaviors, an adaptive sampling strategy may yield more efficient and precise estimates (Thompson 1992). In contrast to the previously discussed sampling designs in which the selection of samples is done a priori to survey initiation, adaptive cluster sampling strategies allow for increased sampling intensity depending on observations made during the survey (Garton et al. 2005). For example, when an individual is encountered during a survey for a rare species, one may intensively sample the adjacent areas to determine if other individuals of that species occur in a clump. Garton et al. (2005) recommended that the initial sample be drawn at random and adjacent units also sampled. The initial and adjacent sampling units form neighborhoods analogous to clusters and are treated similarly to cluster sampling. Size of the clusters need not be constant nor known in advance. For spatially clustered animals, the neighborhood consists of adjacent spatial sample units (Garton et al. 2005). The primary purpose of adaptive sampling is to acquire more precise estimates for a given sample size.

Sequential Sampling

Garton et al. (2005) described sequential sampling as the procedure in which samples are collected one at a time and after each sample is taken, the biologist decides whether a conclusion can be reached; therefore, sample size is not predetermined in advance. After an initial sample is collected, successive samples are added until the biologist determines the estimate has adequate precision, a null hypothesis is rejected, or a maximum sample size has been obtained. The primary advantages are less time and money compared to other sampling designs because sample sizes are minimized, perhaps less than one third of the standard sampling design, to the smallest extent needed for the survey (Krebs 1999, Garton et al. 2005). The sample is still required to have a random distribution throughout the entire population in order to be representative of the population of interest (Garton et al. 2005).

SURVEY METHODS FOR MONITORING

Surveys of a furbearing species may be conducted at various levels of resolution and will answer different questions regarding the population of interest (Gese 2001). Biologists and managers may need to know only where a species occurs (i.e., distribution); this fundamental question is usually needed to determine the presence and distribution of rare, threatened, or endangered species. Methods employed to determine the distribution of a species include habitat mapping; questionnaires, interviews, and sighting reports from the public, agency staff, or others; and confirmation of sign (e.g., tracks, scats, hair, burrows) made by the species.

In their most rudimentary form, surveys of animal sign provide distributional information. With standardization of methods and documentation of the amount of effort, sign-based surveys may also be used as an index of relative abundance (Long et al. 2008). If certain areas are repeatedly surveyed over time and the amount of search effort is recorded, then biologists may standardize survey results (e.g., number of tracks identified/hour, number of scats identified/hour), allowing for trend information over time or comparisons between areas (Gese 2001). Samples from hair snares or scat collections, combined with genetic techniques, can also be used to identify individual animals and subsequently allow for estimation of population size based on a capture-mark-recapture approach (CMR; e.g., Kohn et al. 1999, Lonsinger et al. 2015*a*, Eriksson et al. 2020).

Accurate and consistent identification of species based on tracks, scats, burrows, and hair samples can be difficult to achieve (Gese 2001). Species identification from scats can be assisted by the use of fecal bile acid patterns detected by thin-layer chromatography (Major et al. 1980, Johnson et al. 1981). Examination of hair samples with a light microscope and comparison to a hair key (e.g., Adorjan and Kolenosky 1969, Moore et al. 1974) or reference collection can help with identification of species. Use of genetic techniques also allows for accurate identification of species based upon scat or hair samples (e.g., Foran et al. 1997*a*,*b*; Paxinos et al. 1997; Kohn et al. 1999; Reding et al. 2023 [Chapter 16]). The amount of sign made by an animal may not correlate with population density for some carnivores, such as American badgers (*Taxidea taxus*) and North American river otters (Messick and Hornocker 1981, Melquist and Hornocker 1983, Messick 1987). Additionally, simply because one fails to find sign does not necessarily indicate absence of the species of interest.

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For rare or elusive furbearing species, biologists and managers may be interested in assessing the proportion of area occupied (i.e., occupancy) within a study area, and determining what factors influence the probability of occurrence (or pattern of occupancy) for a species. Biologists can combine replicated surveys for a species (or based on sign associated with that species) with occupancy modeling to evaluate patterns of occurrence, while investigating the influence of environmental factors and accounting for detection probabilities <1 (MacKenzie et al. 2002). The sampling required to assess occupancy and patterns of occupancy requires only identification of animals (or sign) to species, and therefore may be more practical than the sampling required to enumerate individuals within a population, particularly at large spatial scales or for long-term monitoring (MacKenzie and Nichols 2004).

With an appropriate spatial scale (e.g., where the sampling unit approximates the territory size of a species), biologists may be able to use occupancy as a surrogate for population abundance. Estimates of occupancy can also be generated over time to evaluate trends in occurrence and assess spatial dynamics, including rates of site colonization and local extinction, as well as environmental factors that influence spatial dynamics (MacKenzie et al. 2003). Importantly, estimates of occupancy are typically related to the proportion of units that are occupied and, therefore, comparison of occupancy estimates among different regions or study areas can be difficult if the units were sampled with disparate sizes. The assumptions of occupancy modeling should be carefully reviewed (MacKenzie et al. 2002), and are best addressed through appropriate sampling designs that consider the ecology of the species of interest (MacKenzie and Royle 2005).

Once the presence of the species of interest is confirmed in a particular area, the biologist or manager may then want to estimate population abundance and trends in abundance. Population abundance may be monitored indirectly (e.g., track counts, visitation rates at scent stations) to estimate relative abundance, or by directly (counting animals) to estimate absolute abundance (Macdonald et al. 1998). Relative abundance is an index of population abundance that can be compared over time or between or among areas, but does not estimate true abundance. Absolute abundance involves using methods to count or identify animals and then estimate the abundance or population density of animals in the population.

Directly counting animals includes dead animals (e.g., mortality samples, vehicle strikes, harvest data), live animals (e.g., trapping, sightings), or sign that can be assigned to a unique individual (e.g., noninvasive genetic sample). The assumptions of direct counts and the estimators used to determine population abundance should be carefully reviewed (Caughley 1977, Burnham et al. 1980, Skalski and Robson 1992). Counts may involve surveying the entire area of interest, or a subsample of the area and extrapolating those data to the entire area of interest. A probabilistic sampling design and stratification of subsamples to different land-cover types (or other attribute) may increase the validity, usefulness, and precision of the surveys (see sections on Sampling Strategies).

The most appropriate sampling strategy for any species will tend to exploit species-specific tendencies or behaviors. Because of their generally low population density, elusive nature, cryptic coloration, and certain habitat characteristics, direct counts of furbearing species are seldom practical (Gese 2001). Biologists and managers typically utilize a variety of techniques to quantify indirect measures of abundance for animals that are, or have been, in the vicinity. Many of these techniques are useful for detecting presence of a species, but detecting relative or absolute abundance of the population of interest can become somewhat more subjective. Relative abundance is frequently inferred by the frequency with which such sign is encountered. Determining absolute abundance from such information is much more difficult. Some information may be gleaned from different-sized tracks, or pitch and tonal differences in vocalizations (see White et al. 2024 [Chapter 14]). Identification of individuals can be achieved via remote cameras if individuals have unique natural markings (e.g., pelage patterns) or artificial marks (e.g., ear tags). Whereas animal sign may be used to assess distribution and relative abundance of a species, DNA analyses of some types of sign (e.g., hair follicle, saliva, urine, blood, tissue, scat) can provide identification of individuals (e.g., Lonsinger et al. 2015*a*,*b*; Lonsinger et al. 2018*a*; Åkesson et al. 2022).

Detecting changes in population abundance over time with some degree of accuracy, precision, and power requires consistent and standardized application of a technique (Macdonald et al. 1998). Whether using sign-based surveys, indices of relative abundance, or measures of absolute abundance, caution should be exercised when examining population trends (Gese 2001, Bauder et al. 2021). Biologists and managers should be aware of the influence of other variables on survey results, including characteristics (e.g., behavior, color, size) of the species of interest; topography and vegetation characteristics of the survey area; temporal factors (e.g., nocturnally vs. diurnally active species); observer experience, ability, and fatigue; equipment defects, malfunctions, or complexity of use; and spatial distribution of the species (i.e., low versus high population density). Before embarking on population trend analyses, biologists can examine the assumptions and estimate the power of the survey technique to detect population changes (see section on Sample Size and Power).

Considerations for assessing changes, either temporally or spatially, generally require a rigorous standardization of procedures. Perhaps most important among these is accounting for instances where counts accumulate over time (e.g., track counts, scat-deposition rates). For example, Sittenthaler et al. (2020) demonstrated that the abundance of fresh scat increased with population abundance of Eurasian otters (*Lutra lutra*), but there was no relationship between old scats and population abundance. Making comparisons between areas has an added caveat regarding whether there are differences in the spatial-use patterns of the animals between the areas involved.

With repeated sampling over time, both relative indices and absolute estimates of population abundance can be used to monitor population trends (Gese 2001). For many species, this amount of information may be adequate. However, if the population trend indicates an increasing or declining population, then it may be important for the biologist to determine the cause of the change. This

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involves examining the demographic processes that increase (i.e., births and immigration) or decrease (i.e., deaths and emigration) local populations. Population modeling enables biologists to test the influence of competing variables and hypotheses, as well as simulate the effect of alternative management scenarios.

The following descriptions of survey techniques are not meant to include every method being used on every furbearing species, but rather a description of the methods with considerations for their implementation. These descriptions are updated from a previous review of these methodologies (Gese 2001). We again emphasize careful consideration of the question(s) and study design to address the needs of the monitoring program and which method most appropriately meets those needs.

Habitat Mapping

Biologists should not necessarily race out into the field and start looking for animals or signs of them. Careful consideration regarding the habitat requirements for a species followed by examination of landcover maps or aerial or satellite images can save time (e.g., Macdonald et al. 1998). Habitat suitability models have been developed for many species. Predictive models can be used to identify habitat for a species (e.g., Engler et al. 2004, Hatten 2014) and allow for optimization of survey effort (e.g., Greenspan and Giordano 2021). Technological advances that facilitate predictive modeling continue to improve, including the development of high-resolution imagery (e.g., Landsat, Lidar, IfSAR, QuickBird), platforms for acquiring remotely sensed imagery (e.g., satellite, airplane, helicopter, unmanned aerial vehicles [UAVs; i.e., drones]), remote-sensing techniques, and Geographic Information Systems (GIS). Surveys can then be stratified by land-cover types or other landscape attributes (e.g., Macdonald et al. 1998). In Great Britain, use of landscape data from the Countryside Information System (CIS), plus existing mammal records and knowledge of habitat requirements, were used to predict mammal distribution at a national scale (Macdonald et al. 1998). Use of satellite imagery and GIS has also been instrumental in identifying potential habitat for reintroduction (e.g., red wolves [*Canis rufus*]; Van Manen et al. 2000) and recolonization (e.g., gray wolves [*Canis lupus*]; Mladenoff et al. 1999) of furbearing species.

Questionnaires, Interviews, and Sighting Reports

One of the simplest methods of determining the distribution of a species, and possibly gaining a subjective estimate of relative abundance, is collecting sightings and general impressions from various people in the field (e.g., Allen and Sargeant 1975, Hatcher and Shaw 1981, Balčiauskas et al. 2021*a*). Questionnaires, interviews, and sighting reports from hunters, trappers, rangers, mail carriers, tourists, guides, farmers, field personnel, and public citizens have been used with some success to measure distributions, and sometimes population abundance, of different species of furbearers (e.g., Clark and Andrews 1982, Erickson 1982, Strickland and Douglas 1984, Melquist and Dronkert 1987, Fanshawe et al. 1997). Questionnaires were successfully used in Great Britain to detect the presence of elusive carnivores, such as European pine marten (*Martes martes*; Strachan et al. 1996), European polecats (*Mustela putorius*; Birks and Kitchener 1999), and European wildcats (*Felis silvestris*; Balharry and Daniels 1998). Nagy et al. (2012) reported that citizen science provided reliable estimates of habitat-use patterns of coyotes in urban areas. Mueller et al. (2019) stated that community-generated reports of coyotes and red foxes may prove useful for managers to monitor these species in urban environments. Balčiauskas et al. (2021*a*) described reports from interested citizens, including hunters, foresters, and farmers, that provided useful information on distribution, pack size, and pack numbers of gray wolves.

More in-depth questionnaires or interviews with persons who possess intimate knowledge of an area and who spend considerable time in the field may not only provide information about distribution and population status (Fuller et al. 1992), but may also be used to obtain a general, subjective estimate of relative abundance (e.g., Allen and Sargeant 1975, Balčiauskas et al. 2021*a*). Many government agencies compile reports on population status of many furbearing species using this method, particularly in regions where agencies are unable to invest the considerable resources required for robust population assessments, as well as when a species is so abundant that anything other than a major change in abundance is of minor concern. Questionnaires have been used when agencies require a large-scale assessment of the distribution of a species (e.g., Fuller et al. 1992), or in circumstances when little is known about the ecology of the species of interest. This is especially useful for rare species with large geographic distributions. Questionnaires are often sent to trappers and field personnel to monitor population trends of furbearing species (Hatcher and Shaw 1981, Clark and Andrews 1982, Strickland and Douglas 1984). Challenges of this technique (see review by White et al. 2005) include misidentification of species, low response rates to the questionnaire, and concentration of animal sightings along roads or near human habitation (i.e., rare species inhabiting areas of low population densities of humans may go undetected or unreported).

Presence of Sign

In the absence of visual confirmation of the animal, biologists and managers may resort to surveys of animal sign to determine presence of a species. Sign-based surveys have been used to determine distribution of most furbearing species, such as Eurasian otters (Macdonald and Mason 1982), North American river otters (Melquist and Hornocker 1983), and American badgers, American mink (*Neogale vison*), coyotes, northern raccoons, red foxes, and striped skunks (Sargeant et al. 1993, Macdonald et al. 1998).

Different methods of sign-based surveys include counting and identifying tracks, scats, scratches, burrows or dens, and hair samples. For example, during diurnal periods, surveys for sign of black-footed ferrets have been conducted throughout the prairie ecosystem to locate remnant populations (Richardson et al. 1985). Trained scent-detection dogs have even been used to search for ferrets and improve detection of burrows occupied by black-footed ferret (Boulerice et al. 2024 [Chapter 49]; E. Dean, Southwestern Research Institute, unpublished report). Conspicuous burrows of American badgers have been used as an indicator of presence. Surveys at bridges over rivers have been used to determine presence of North American river otters (Melquist and Dronkert 1987). Spraint (defecation) surveys for Eurasian otters provided distribution information in Great Britain, but the abundance of spraint seemed to be unrelated to population abundance (Kruuk et al. 1986, Conroy and French 1987). Due to their increased availability and decreasing costs, UAVs have been increasingly used to document conspicuous structures for determining species presence or estimating occupancy patterns. Examples include presence of dams and lodges of North American beavers (*Castor canadensis*), and use of infrared cameras to assess whether huts are occupied or have been abandoned by muskrats (*Ondatra zibethicus*; Fig. 1).

When sign includes relatively fresh biological material (e.g., scats, feces, hair, saliva), genetic techniques may be used to identify species and individuals within a species, which in turn offers methods to estimate population size and demographics (e.g., Kohn et al. 1999, Frantz et al. 2003, Lonsinger et al. 2015*a*, Eriksson et al. 2020, Reding et al. 2023 [Chapter 16]; see section on Noninvasive Genetic Sampling). Practitioners may also be able to extract DNA of the species of interest from surfaces (e.g., snow) or substrates (e.g., soil), or within mediums (e.g., water); these sources of DNA are commonly referred to as environmental DNA (eDNA). For example, eDNA collected during snow-track surveys and snow collected at remote cameras was used to detect and identify 3 rare forest carnivores (Canada lynx [*Lynx canadensis*], fishers [*Pekania pennanti*], wolverines [*Gulo gulo*]; Franklin et al. 2019).

Track Counts

Tracks made by carnivores along river beds, dry washes, sandy fire breaks or roads, or on snow-covered roads and trails (Fig. 2) have been used as a relatively simple and inexpensive measure of relative abundance for several species of canids (e.g., Crête and Messier 1987, Palomares et al. 1996, Moran et al. 2016, Pozzanghera et al. 2016, Droghini and Boutin 2018), felids (e.g., Van Sickle and Lindzey 1992, Stander 1998, Squires et al. 2012, Walpole et al. 2012, Montgomery et al. 2014), and mustelids (e.g., Melquist and Dronkert 1987, Crowley et al. 2012, Haskell et al. 2013, Sirén et al. 2017, Jun et al. 2021). Furbearing species that occupy regions with snowfall have been monitored through the use of counting tracks along established transects within 1–2 days following fresh snowfall. Track counts during winter along standardized transects have been routinely used to index the relative abundance and population trends of several furbearing species (e.g., Slough and Smits 1985, Jun et al. 2021, Powers et al. 2021). Snow-tracking datasets were used to identify habitat use by Canada lynx in Ontario, Canada (Phillips et al. 2021), and in northeastern Minnesota, USA (Hostetter et al. 2020). Snow-tracking has also been used effectively for monitoring the behavioral responses of carnivores to snowmobiles (Gese et al. 2013), determining how coyotes negotiate deep-snow landscapes (Dowd et al. 2014), and examining how carnivores respond to newly constructed wind farms (Sirén et al 2017) and widening of highways (Boyle et al. 2020).

Franklin et al. (2019) improved identification of carnivore species by collecting DNA during snow-tracking surveys. Golden (1987) was able to conduct aerial-based track counts for wolverines in unforested areas of Alaska, USA. Becker (1991) conducted aerial-based surveys along transects to determine the number of Canada lynx and wolverines occupying an area on the Kenai Peninsula, Alaska. Ballard et al. (1995) reported acceptable precision between line-intercept sampling of tracks and estimates of population density of gray wolves based on telemetry methods. Kawaguchi et al. (2015) indicated that snow-track counts were likely valid sources to infer population dynamics of American marten (*Martes americana*), American red squirrel (*Tamiasciurus hudsonicus*), and weasels (primarily *Mustela erminea* and

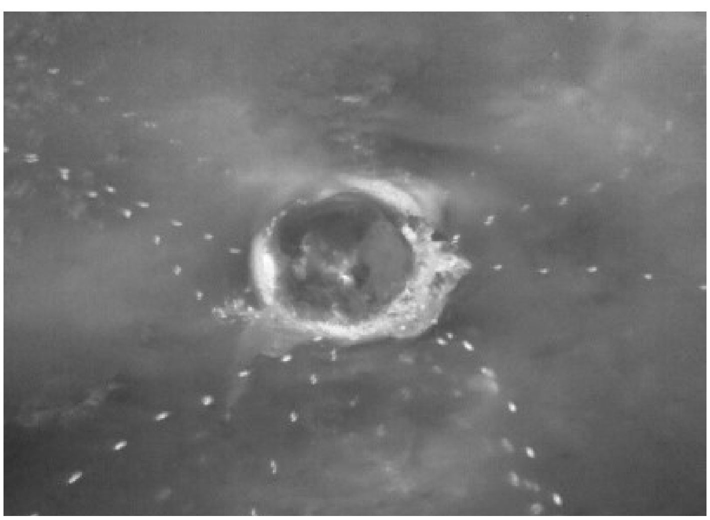

Fig. 1. Infrared cameras have been used during surveys of furbearing species, including mounted to unmanned aerial vehicles (e.g., drones) to assess whether the cameras can be used to assess whether huts of muskrats (*Ondatra zibethicus*) are occupied or have been abandoned. Image courtesy of Indiana Department of Natural Resources, USA.

Fig. 2. Animal tracks in the snow (or other appropriate substrate) can be used to identify species to confirm presence and to index population abundance of a furbearing species. Image courtesy of C. Bromley, National Park Service, USA.

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Neogale frenata); the technique was repeatable, efficient, reasonably accurate, and relatively inexpensive. Recently, Åkesson et al. (2022) demonstrated using snow-tracking to collect and analyze noninvasive genetic samples (i.e., scats, urine, hair, blood) from gray wolves to identify individuals, and detect territorial pairs, packs, and reproductive events with a high degree of reliability.

Biologists using track counts should be aware of some potential pitfalls. Misidentification of species based on tracks and low power to detect population changes can occur (Van Sickle and Lindzey 1991, Kendall et al. 1992, Ballard et al. 1995, Beier and Cunningham 1996). Precision can be increased by increasing sampling effort (e.g., increase the number of transects, increase length of transects) when surveying a species (e.g., mountain lions; Van Sickle and Lindzey 1991) with large home-range sizes (but see Kendall et al. 1992). Also, combining snow-tracking surveys with remote cameras increased detection of Canada lynx in central British Columbia, Canada (Crowley et al. 2013).

Much of the power of standardized track surveys is dependent upon a high rate of encountering sign along the transects (Kendall et al. 1992). When conducting surveys in areas with snowfall, biologists and managers must consider the condition, consistency, and presence of snow; ambient temperature; and time of year (Pozzanghera et al. 2016, Sirén et al. 2017). The level of experience of observers for accurately identifying species based on tracks is also crucial for consistent and reliable monitoring. With continued warming of the climate, agencies must also consider the utility of snow-tracking surveys as a monitoring tool in many areas should the duration and quality of snow cover become less reliable.

Surveys of Dens and Burrows

Ground-based and aerial-based surveys for active dens conducted along transects is a method of indexing relative abundance of some furbearing species (e.g., Gallant et al. 2012). As mentioned previously, use of UAVs is gaining in popularity for wildlife agencies to count conspicuous structures to determine presence or estimate occupancy of a furbearing species. Annual surveys of dens have been used to monitor populations of Arctic fox (*Vulpes lagopus*) in northern dry tundra (Macpherson 1969, Garrott et al. 1983), but seem to have little application in areas of coastal wet tundra (Anthony 1996). Ground-based and aerial-based surveys for dens have been used to monitor populations of kit foxes in desert environments (O'Farrell 1987) and red foxes in prairie ecosystems (Trautman et al. 1974). In the Arctic, dens of polar bears (*Ursus maritimus*) have been located with forward-looking infrared (FLIR) cameras, but with mixed results depending on detection platform (helicopter, airplane, or ground-based), environmental conditions (e.g., ambient temperature, wind speed, precipitation, amount of sunlight) and thickness of den ceiling (Amstrup et al. 2004, Robinson et al. 2014, Pedersen et al. 2020, Smith et al. 2020, Woodruff et al. 2022).

The key to effective surveys for dens and burrows is a species that makes conspicuous dens or burrows. These surveys can be relatively expensive (e.g., aerial-based surveys, although UAVs can substantially reduce costs) and labor intensive (e.g., ground-based surveys). In general, surveys of dens and burrows entail personnel walking or flying along a route or transect searching for active dens. The presence of feces or tracks at burrows or dens can assist in species identification and confirmation that the site is active. Ground-based surveys conducted along transects can also be used to calculate the density of dens if biologists record the perpendicular distance from the transect to each den (Burnham et al. 1980). Conspicuous burrows dug by American badgers have been used to indicate presence, but there seems to be no correlation between density of burrows and population abundance (Messick and Hornocker 1981, Messick 1987, Bylo et al. 2014). However, burrows of American badgers can be indicative of denning and reproductive activity (Duquette et al. 2014).

Although not technically dens, lodges constructed by North American beavers have been counted to determine the distribution and relative abundance of this furbearing species (King et al. 1998, Swimley et al. 1999). This technique would likely be ineffective for indexing population abundance of carnivores with large social units. For example, coyotes, regardless of pack size, typically have one natal den to rear offspring (i.e., a pair of coyotes uses the same number of dens as a pack of 7 coyotes). For animals in packs or clans, the number of dens would more likely indicate the number of social units present within that area, but not the number of animals in each social unit.

Scent-station Surveys

A common method used for indexing population abundance is scent-station surveys. Scent-station surveys have been used to estimate the relative abundance of several species of canids (e.g., Linhart and Knowlton 1975; Roughton and Sweeny 1979, 1982; Morrison et al. 1981; Travaini et al. 1996; Sergeyev et al. 2020), felids (e.g., Conner et al. 1983, Fredebaugh et al. 2011, Kapfer and Potts 2012), mustelids (e.g., Humphrey and Zinn 1982, Melquist and Dronkert 1987, Hein and Andelt 1995, Loughry et al. 2012, Burr et al. 2017), and procyonids (e.g., Clark and Andrews 1982, Conner et al. 1983, Smith et al. 1994, Kowalski et al. 2015, Rockhill et al. 2016). Scent-station surveys involve placing an olfactory attractant within a 1-m circle of sifted dirt, a substrate suitable for well-defined imprints of tracks from species that visit stations (Fig. 3).

Webster and Beasley (2019) reported that skunk essence, a commonly used attractant in contemporary research on furbearing species, had the highest visitation rates within the set of attractants tested. Tracks within the sifted dirt are identified to species, which confirms presence; however, lack of detection of any particular species may or may not be absence, but rather considered non-detection. Typically, scent stations are spaced at a predetermined interval along roads or trails and monitored for 3–4 consecutive nights and checked by a biologist each day to identify tracks; the sifted area is swept smooth after each night to ensure removal of all tracks before the next day. When determining the spacing of stations, an important consideration for biologists is the movement patterns and home-range size of the species of interest (i.e., close spacing for close-ranging species, increased spacing for species with larger home ranges or longer daily movements; Rockhill et al. 2016). The frequency of visitation by animals to operable stations (i.e., those not disturbed by wind, rain, or vehicles) is used

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Fig. 3. A scent station consists of a 1-m (3 ft) circle of sifted dirt and an olfactory attractant placed in the center. Image courtesy of C. Thompson, Forest Service, USA.

as an index of relative abundance (Webster and Beasley 2019), and to determine detection rates, estimate species richness, and compare distributions of species (Rockhill et al. 2016). More detailed information about study design and data interpretation associated with scent-station surveys is available in Smith et al. (1994) and Sargeant et al. (1998).

Scent-station surveys have been used successfully to detect northern raccoons (Rockhill et al. 2016), and some biologists reported that scent-station surveys reflected changes in population abundance of this species; however, Smith et al. (1994) claimed that no association between visitation rates and population density of northern raccoons existed. Knowlton (1984) calculated a positive correlation (r^2 = 0.79) between population indices derived from scent-station surveys and estimated population density of coyotes. Seasonal changes in habitat use and visits to multiple stations by a single animal can contribute to invalid correlations of population density and visitation rates. Misidentification of species based on tracks, problems with weather (e.g., wind, precipitation), avoidance of sifted substrate by some animals, and a relatively labor-intensive technique are challenges when considering scent-station surveys.

Although scent-station surveys can be economical in areas with low population density of the species of interest, detection probabilities can be low (Rockhill et al. 2016). A variation of the scent-station survey used to estimate a population index for dingoes (*Canis familiaris dingo*) in Australia was the activity index (Allen and Engeman 1995, Allen et al. 1996). This index of animal visitation simply uses a sifted-dirt area on a road without an olfactory attractant. The number of linear sets of tracks crossing the sifted area is used to assess relative abundance and calculate a variance estimate (Engeman et al. 1998).

Track Plates

The use of track plates to determine presence of furbearing species is gaining popularity, particularly for detection of carnivores in forested areas (e.g., Zielinski 1995, Ray and Zielinski 2008, Barrett et al. 2012, Zielinski et al. 2013, Lindsay and Ash 2021), but also in urban (e.g., Riem et al. 2012, Jordan and Lobb-Rabe 2015) and suburban (Lumpkin et al. 2012) environments. Track plates provide a reliable measure of distribution or presence of species, but may be unreliable for determining relative abundance. Track counts in prepared beds (i.e., plywood coated with chalk dust) have been used to estimate the distribution, but not abundance, of American mink (Burgess and Bider 1980, Humphrey and Zinn 1982). Similarly, soot has been applied to track plates to ease identification of tracks of American marten (Barrett 1983, Zielinski and Truex 1995), fishers (Zielinski 1995, O'Neil and Swanson 2010, Matthews et al. 2011, Triska et al. 2011, Loughry et al. 2012), and weasels (Barrett 1983, Clark and Campbell 1983). Track plates have also been used to identify larger-bodied carnivores, including coyotes (Reed 2011, Barrett et al. 2012, Melville et al. 2015), foxes (*Vulpes* spp.; Reed 2011, Barrett et al. 2012, Melville et al. 2015), Canada lynx (Melville et al. 2015), and bobcats (Reed 2011, Melville et al. 2015).

A detailed description of track plates and the implementation of both enclosed track-plate boxes and unenclosed track plates is provided by Zielinski (1995). In general, track surfaces may be produced from smoked or carbon-sooted aluminum plates, contact paper (tacky white paper), chalk, or ink. An attractant (visual, olfactory, or both) may be used to increase the probability of investigation of the tracking station by an animal; while investigating the attractant, tracks are made on the tracking surface by the animal (Jordan and Lobb-Rabe 2015, Lindsay and Ash 2021; Fig. 4). Transportation of track plates without damage, protecting track plates from the weather, ensuring an investigating animal steps on the plate, and identification of species via tracks are all challenges that require prior planning when using track plates (see Zielinski 1995 and

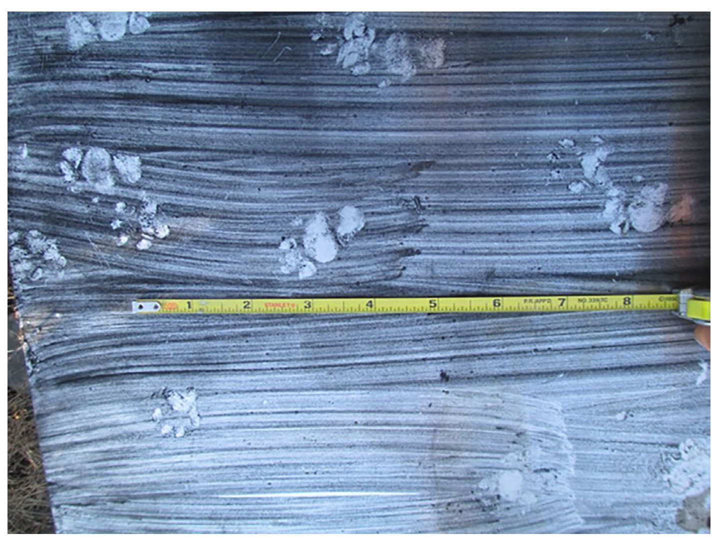

Fig. 4. Tracks from a striped skunk (*Mephitis mephitis*) on a track plate. Image courtesy of T. Lysak, Cascadia Wild, USA.

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Zielinski and Truex 1995). Identification of individuals may also be possible. For example, new techniques with track plates have been able to identify individual northern raccoons (Ellison and Swanson 2016) and fishers (O'Neil and Swanson 2010) by measuring their metacarpal pads. If tracks can be measured and individuals reliably and repeatably identified, mark-recapture methods could be applied to track data to estimate population size.

Scat-deposition Transects

The rate at which scats are deposited along roadways has been used as an estimate of relative abundance for canid species, including coyotes (Clark 1972, Davison 1980, Andelt and Andelt 1984), foxes (*Vulpes* spp.; Schauster et al. 2002, Dempsey et al. 2014), and gray wolves (Crête and Messier 1987). The method involves designating transects or routes along a roadway, removing all scats from transects along the road, and returning to collect all scats encountered (typically after a 14-day period). The scat index is computed as the number of scats collected/transect/14-day period (Davison 1980). If transects vary in length, or the time periods vary in the number of days between collections, then the index can be standardized to number of scats/km/day. Rates of scat deposition for coyotes were correlated ($r^2 = 0.97$) with estimates of population density derived from mark-recapture techniques using radioisotope tagging of feces (Knowlton 1984).

For long-term monitoring, scat-deposition transects should be conducted along the same routes at the same time of year to avoid introducing biases associated with differential digestibility of seasonal prey (hence, differential rates of scat deposition) and seasonal changes in food items consumed (Andelt and Andelt 1984). Misidentification of species based on scats and high (or variable) volume of vehicles on roadways can also be problematic when using scat-deposition transects (Kluever et al. 2015, Lonsinger et al. 2016). Use of genetic techniques for identifying species from scats may alleviate the problems of misidentification (Foran et al. 1997*a*,*b*; Lonsinger et al. 2015*a*). In addition, genetic identification of individual animals collected during scat-deposition transects can be used to estimate population size (Paxinos et al. 1997; Kohn et al. 1999; Lonsinger et al. 2015*b*, 2018*a*), though collection of relatively fresh scats is generally required to obtain reliable amplification of DNA for subsequent capture-recapture models (see section on Noninvasive Genetic Sampling, and Reding et al. 2023 [Chapter 16] for more details on collecting and utilizing samples for DNA amplification and individual identification).

Vocalization-response Surveys

For social species of carnivores that utilize long-range vocalizations (e.g., roars, howls, whoops) to communicate, biologists have been able to use the response rate to simulated vocalizations to estimate relative abundance. Howling surveys for coyotes (e.g., Wenger and Cringan 1978, Okoniewski and Chambers 1984, Hansen et al. 2015, Cherry et al. 2016) and gray wolves (e.g., Carbyn 1982, Harrington and Mech 1982, Fuller and Sampson 1988, Gable et al. 2018, Garland et al. 2020, Boyd et al. 2023 [Chapter 32]) have been used for estimating population abundance. Vocalization-response surveys typically employ recorded vocalizations, although human imitation of sounds is sometimes effective. Traveling along roads or trails and stopping at predetermined intervals, observers produce vocalizations and then listen for a specified amount of time for a response from the species of interest.

Biologists may conduct a survey during several nights and use the rate of vocalization response to estimate relative abundance of the species. O'Gara et al. (2020) located gray wolves in northern Wisconsin, USA, using acoustic triangulation with similar precision as ground-based telemetry, although the two methods resulted in different predicted locations. Standardization and consistency of vocalization-response surveys is necessary for reliable and comparable results for trend analyses (Gable et al. 2018). Biologists should be aware of the seasonal, social (e.g., group size), temporal (e.g., time of day, wind speed and direction), and spatial factors influencing vocalization rates (Laundré 1981, Harrington and Mech 1982, Walsh and Inglis 1989, Gese and Ruff 1998, Ausband et al. 2020). For accurate results, biologists need to intensively survey the area of interest to obtain adequate coverage (Fuller and Sampson 1988). Notably, Hansen et al. (2015) cautioned that howling surveys should be considered an index rather than a true estimate of abundance.

Frequency of Depredation Complaints

The frequency of livestock-depredation complaints may be useful as an indicator of presence of a given species. For example, Moheb et al. (2012) used a combination of public reports, field evidence, and documentation of depredations to identify status and distribution of brown bears (*Ursus arctos*) in Afghanistan. Because this relationship has not been robustly tested, biologists should be cautious of this technique, as depredation rates are subject to variable behavior of carnivores (i.e., not all individuals kill livestock, scavenging of carcasses depredated by another individual or species), changes in stocking rates of livestock, reporting rates from producers, land-cover type, size of area used, husbandry practices, environmental variables, and reporting accuracy (Fritts 1982, Mech et al. 1988).

Hair Sampling

Collection of hair samples and subsequent identification of species based on characteristics of the hair (and comparison to reference collections) or genetic analysis can be used to assess distribution of species (e.g., Foran et al. 1997*a*,*b*; Paxinos et al. 1997; Kohn et al. 1999; Trapp and Flaherty 2017). Hair may be collected via various methods often collectively referred as hair snares, snags, or tubes, and involve removal of very small amounts of hair from animals. Specific devices include a barbed-wire corral around a bait or barbed wire wrapped around a tree, gun-cleaning (or other) brushes or spring-loaded alligator clips in a tube, carpet tacks protruding from a piece of carpet, or using sticky paper that will pull some hairs when contact occurs (Fig. 5).

Some techniques passively collect the hair (i.e., animal walks under a wire fence), or require the animal to actively rub on the device and needs a scent to illicit a rubbing behavior from the animal. The type of snare or snag is often dependent

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on the species. An olfactory attractant is generally required to illicit a facial-rubbing response from felids, whereas more passive techniques may be used for canids, mustelids, and ursids (e.g., Belant 2003, Long et al. 2008).

Collection of hair has been used on a variety of furbearing species. Hair snares were used across 4 states to determine occupancy by wolverines (Lukacs et al. 2020). Similarly, hair snares were used to determine occupancy and habitat-patch use by fishers (Ellington et al. 2017, Linden et al. 2017). Trapp and Flaherty (2017) used hair snares to describe respective distributions of American red squirrels, northern flying squirrels (*Glaucomys sabrinus fuscus*), and southern flying squirrels (*Glaucomys volans*), each with low population densities, by differentiating hairs based on morphological measurements.

Hair snares, hair tubes, and hair snags can provide hair samples with follicles that can be genetically analyzed to identify individuals and estimate population size (e.g., Foran et al. 1997*b*, Paxinos et al. 1997, Kohn et al. 1999, Frantz et al. 2004). Roundsville et al. (2022) reported a novel design of hair snare for collecting genetic material to noninvasively detect bobcats. Hairs from very large carnivores are routinely collected with hair snares (barbed-wire corrals) to collect genetic material for determining population size and genetic diversity (e.g., Kendall et al. 2015, Hooker et al. 2020, Tamendemberel et al. 2021, Baciu et al. 2022) or diet composition via isotope analysis (Ro et al. 2021). Additionally, hair snares have been used to collect hair from Eurasian red squirrels (*Sciurus vulgaris*) to assess physiological condition via cortisol analysis of hair (Cordeschi et al. 2021).

Collection of hair follicles for genetic analysis is often species or system specific. For example, Phoebus et al. (2020) recommended using hair snares as the primary method for DNA inventories for grizzly bears (*Ursus arctos horribilis*), whereas collecting and analyzing scats increased the precision of population estimates. Genetic data from hair snags were used to successfully identify 100% of individuals, whereas genetic data from scats were used to successfully identify 14% of individuals; however, the use of scats had a higher success rate (98%) when identifying species compared to hair (80%). For other species, there is evidence that the use of hair snares is not very effective, and scat-based sampling is a more reliable approach to acquiring usable DNA samples.

Scent-detection Dogs

Use of professionally trained domestic dogs and handlers (Fig. 6) to detect the locations of substances (e.g., carcasses, scats, hair, predation sites) associated with a furbearing species has been a successfully applied field technique, including to support analyses of population size (e.g., Smith et al. 2003*a*, MacKay et al. 2008, Wilbert et al. 2019, Petroelje et al. 2021). Detection dogs can detect mammalian scats more efficiently and accurately than human surveyors (e.g., Smith et al. 2003*a*, 2005, 2006; Grimm-Seyfarth et al. 2019). Additionally, detection dogs can be trained to detect up to 40 odors, including minute amounts of scat and individual identification with a species, and recall each odor for ≥12 months (e.g., Wasser et al. 2009, Rosell et al. 2020, DeChant and Hall 2021, Waggoner et al. 2022). Dogs have been effective in locating scats of species with low population

Fig. 5. The use of bait stations with hair snares (e.g., gun-cleaning brushes affixed to corrugated plastic and attached to the base of a tree) allow biologists to collect hair samples to determine gene flow and population structure, such as from Sierra Nevada red foxes (*Vulpes vulpes necator*) in the Cascades of Oregon, USA. Image courtesy of T. Hiller, Wildlife Ecology Institute, USA.

Fig. 6. The use of a professionally trained detection dog and handler is gaining popularity for finding fresh biological samples (e.g., scats) useful for collecting data for determining presence and population abundance of furbearing species. Image courtesy of Bureau of Land Management, USA.

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densities (e.g., Long et al. 2007, Davidson et al. 2014), both in rugged terrain and complex vegetative ecosystems (e.g., Wasser et al. 2004), as well as detecting scats of rare species (e.g., Reindl-Thompson et al. 2006, Hatlauf et al. 2021). Once a dog locates and indicates to the handler that it has identified a target of interest (e.g., a scat of the species of interest), the handler can collect a fecal sample, which can then be analyzed with genetic techniques to identity both the species and individual that deposited the scat for occupancy modeling and mark-recapture estimates, respectively (e.g., Long et al. 2011, Barry et al. 2021, Ruprecht et al. 2021), depending on the objectives of the project.

Multiple factors need consideration when using detection dogs, including wind speed, precipitation, temperature, humidity, countermarking of scats by nontarget species, and experience and training of the dog and handler (e.g., Reed et al. 2011, DeMatteo et al. 2018, Mutoro et al. 2021, Rutter et al. 2021). Bennett et al. (2020) recommended that future studies using detection dogs include reporting of the level of effort as the total area and time spent searching, and estimates of sensitivity and precision. Rutter et al. (2021) advised that the dog-handler team be trained in the spatial scale and environmental context similar to their working conditions. Genetic material collected from scats located by detection dogs combined with other techniques (e.g., remote cameras) has proven successful in monitoring several furbearing species, such as American black bears, American marten, bobcats, Canada lynx, coyotes, fishers, and gray wolves (e.g., Long et al. 2011, Mumma et al. 2015, Moriarty et al. 2018, Cozzi et al. 2021).

Remote Cameras

A method continuing to gain popularity is the use of remote cameras (e.g., Kays and Slauson 2008, Gould and Harrison 2018). Remote cameras have been used successfully to estimate the number of individuals in a population since the late 1990s (e.g., Karanth 1995, Karanth and Nichols 1998), and have been deployed to detect various furbearing species (e.g., Kucera et al. 1995, Foresman and Pearson 1998, Ruprecht et al. 2021), including elusive or nocturnal felids (Joslin 1982, Rappole et al. 1985, Soisalo and Cavalcanti 2006) and mustelids (Happe et al. 2020, Barry et al. 2021). Remote cameras can be modified to be activated by pressure-sensitive plates, motion or heat detectors, or an infrared beam, or may collect time-lapse imagery.

Remote-camera systems are used primarily to detect the presence of species (Kucera et al. 1995, Naves et al. 1996, Foresman and Pearson 1998), or identify predators at bait stations or nests (Savidge and Seibert 1988). Cameras can also be used to collect data to estimate population abundance if individuals can be identified by artificial tags (e.g., ear tags, radio-collars) or natural features (e.g., pelage characteristics; Karanth 1995, Karanth and Nichols 1998, Dixon 2003, Soisalo and Cavalcanti 2006), and then applying mark-recapture techniques to estimate population size. Species that lack individually distinguishable patterns associated with pelage are generally not candidates for the use of remote cameras to collect data to estimate population size, as these data can produce unreliable estimates (e.g., Alexander and Gese 2018).

Remote cameras are increasingly used to corroborate other survey methods for the collection of data used for estimating population abundance (Ruprecht et al. 2021), occupancy modeling (Barry et al. 2021, Happe et al. 2020), habitat use (Parsons et al. 2019, Happe et al. 2020), and behavior (e.g., activity patterns, vigilance; Chitwood et al. 2020, Kemna et al. 2020). Remote cameras can also be used to assess animal health, such as visible infections of mange in coyotes (Murray et al. 2021, Reddell et al. 2021), reproductive characteristics (Fig. 7A), and prey provided by adults to young at the den (Fig. 7B). Remote cameras have the added benefit of a permanent photographic record that is available for examination and potential use by other researchers.

The principal challenge of using remote cameras is consistent triggering of the cameras when an animal is present, similar to problems associated with track plates. Seasonal differences, the presence or absence of an attractant, and if used, the specific type of attractant, can all influence detection rates of different furbearing species, and should be considered in survey design (Ferreras et al. 2018, Heinlein et al. 2020, Iannarilli et al. 2021, Dart et al. 2023). Although remote cameras may be used to collect data that provide valid estimates of population abundances for relatively rare species (e.g., Moriarty et al. 2018), these estimates may be highly variable (e.g., Greenspan et al. 2020).

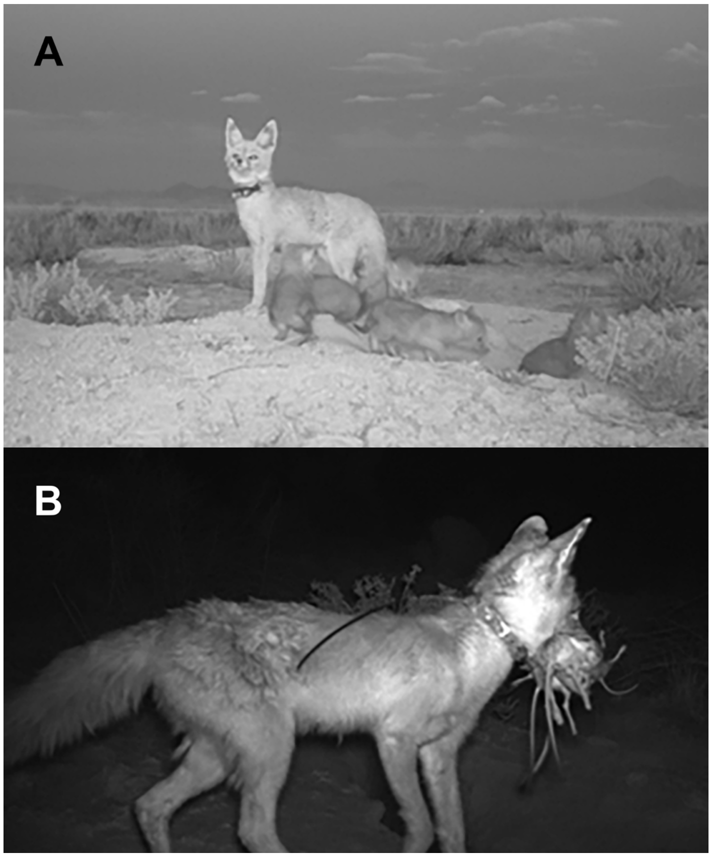

Fig. 7. Remote cameras provided evidence of A) reproduction, and B) prey delivery to the den, among kit foxes (*Vulpes macrotis*) in the west desert of Utah, USA. Images courtesy of B. Kluever, Utah State University, USA.

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Recent developments in quantitative approaches also provide opportunities for estimating abundance of unmarked animals with data collected from cameras through space- or time-to-event models (Moeller et al. 2018), or camera-based distance sampling (Howe et al. 2017). Generally, a large number of cameras are required for adequate coverage of a large area to provide reliable estimates based on capture-recapture techniques. The optimal pattern and density of deployment for remote cameras varies based on the monitoring objective and parameters of interest; e.g., higher densities of cameras may be required for sampling associated with spatial capture-recapture techniques and estimation of population density compared to sampling to estimate patterns of occupancy.

A large variety of camera models are commercially available from several manufacturers and are relatively inexpensive. Models of cameras differ in their functionality (e.g., motion triggered versus timelapse options) and performance. For example, there is evidence that some models of cameras often fail to detect smaller-bodied species, which could bias monitoring results and inferences (Urbanek et al. 2019, Dart et al. 2023). Computer programs employing machine-learning algorithms can process the large volume of digital images obtained to aid with identifying species of interest, or for some species, individuals within a species (e.g., Norouzzadeh et al. 2018, Tabak et al. 2019), though these algorithms are more reliable at excluding digital images with information that is not useful (e.g., no species recorded in image).

Harvest Data

Current and historical harvest data can be a valuable resource associated with assessing distribution and abundance of furbearing species, depending on the quality and quantity of those data (see Hiller et al. 2023 [Chapter 10]). In the Canadian provinces, mandatory pelt tagging or sealing reports have also been used to estimate population trends of furbearing species (Novak 1987). In the United Kingdom, a decline in population abundance of Eurasian otters was observed through a decrease in harvest success (Strachan and Jefferies 1996). While detailed information from harvested animals can be used to construct models for population estimation (Clark and Andrews 1982), other data may need to be included with harvest data to produce a reliable estimate of population trends.

Pelt prices, differing harvest methods, and environmental and social factors all influence harvest rates. Clark and Andrews (1982) speculated that harvest data may be used to assess population trends of furbearing species with consistently low pelt values because harvest trends of these species would be less affected by management actions and annually changing harvest effort. Gompper and Hackett (2005) used long-term harvest data collected during the 20th century to assess population trends of eastern (and plains) spotted skunks (*Spilogale putorius* and *Spilogale interrupta*, respectively) across 10 states; after accounting for variation in harvest pressure, their analysis indicated a substantial population decline. Other problems associated with harvest data include a proportion of inaccurate and incomplete harvest reports, depending on whether submission is mandatory or voluntary (Sanderson 1951*a*, Clark and Andrews 1982, Beaman et al. 2005). For some species, harvest data may be too sparse to produce reliable estimates of population trends (Kaufman 1987).

One method for estimating harvest rate (i.e., proportion of population that is harvested) and population size of bobcats uses the total number of harvested animals, sex-specific age distribution of the harvest, and estimates of harvest effort collected during several years (Paloheimo and Fraser 1981, Rolley 1987). When using harvest data, the validity of the underlying assumptions should be evaluated (Gilbert et al. 1978). Population trends of furbearing species have been examined in relationship to past and current harvest data for many species of canids (e.g., Clark and Andrews 1982, Erickson 1982), felids (e.g., Erickson 1982, Quinn and Parker 1987, Rolley 1987), mustelids (e.g., Barbour and Davis 1974, Clark and Andrews 1982, Linscombe et al. 1982, Strickland and Douglas 1984, Melquist and Dronkert 1987), mephitids (e.g., Gompper and Hackett 2005), and procyonids (e.g., Sanderson 1951*a*, Clark and Andrews 1982, Kaufman 1987, Novak 1987).

Additional testing needs to be conducted to confirm the relationship between population density and harvest statistics. Bauder et al. (2021) described that broad-scale (i.e., statewide) trends in harvest-based indices were related to other survey indices for some furbearing species (e.g., coyotes, northern raccoons), but not for other species (e.g., red foxes). Mowat et al. (2022) reported harvest success and effort were not good proxies to index trends in abundance of gray wolves due to uncertainty in number of hunters and their harvest effort. They reported that harvest of gray wolves can be used to index large changes in population abundance, but not at finer spatial scales (i.e., management unit).

Species with required pelt tagging or mandatory checks by state wildlife agencies can provide an opportunity for biologists to collect genetic samples from harvested furbearers, which can be used to improve population monitoring. For example, Reding et al. (2013) used genetic samples from harvested bobcats in Oregon, USA, to evaluate patterns of population structure and gene flow. Genetic samples may also facilitate the application of close-kin mark-recapture approaches to monitor population abundance exclusively through the use of samples from harvested individuals.

Samples from Animals Killed on Roadways

The frequency of animal carcasses on roadways has been proposed as a measure of population trends for some species, usually as an index of relative abundance. For example, the numbers of northern raccoons and striped skunks struck and killed by vehicles have been used as measures of relative abundance (Clark and Andrews 1982, Bartlett and Martin 1982, Gehrt 2002). In Great Britain, population densities of red foxes were positively related to vehicle collisions (Baker et al. 2004). McClintock et al. (2015) were able to produce the first statistically defensible population estimates of radio-collared Florida cougars (*Puma concolor coryi*) by incorporating telemetry with surveys of animals killed by vehicles. In addition, animals killed by vehicles can be examined for disease prevalence (Calabuig et al. 2019) and genetic information (Allio et al. 2021).

In eastern and southern Africa, researchers utilized vehicle-killed animals to assess genomic differences between the bat-eared fox (*Otocyon megalotis*) and the aardwolf (*Proteles cristatus*; Allio et al. 2021). Researchers used genetic samples from American badgers (Kierepka and Latch 2016) and European badgers (*Meles meles*;

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Frantz et al. 2010) killed by vehicles to assess gene flow and patterns of dispersal. While using mortality data associated with vehicles is intuitively simple and appealing, differences in animal behavior and movements, habitat, traffic volume, type of road surface, and density of roads on the landscape likely influence kill rates of some furbearing species. In Tasmania, there was no correlation between the number of marsupial species killed by vehicles and the abundance of local populations (Nguyen et al. 2019).

Several studies have included recommendations that use of carcass counts along roadways be used only with caution (Stevens and Dennis 2013, Quiles et al. 2021, Silva et al. 2021), whereas other studies have included suggestions that these surveys are effective and valid (Schmidt et al. 2021). Researchers have determined that effectiveness of carcass surveys along roadways was also dependent on monitoring scale (Balčiauskas et al. 2021*b*), although the need for standardization and consistent methods was recognized (Schwartz et al. 2020, Silva et al. 2021). Birks and Kitchener (1999) calibrated mortality of European polecats along roadways with numbers estimated from intensive live trapping. In its simplest form, samples from animals struck and killed by vehicles can be used to confirm species presence.

Spotlighting Surveys

Spotlighting surveys are a cost-effective method for some furbearing species and is typically used for assessing the relative abundance of nocturnal species. Estimates of relative abundance for nocturnally active species, such as northern raccoons (Andrews 1979, Rybarczyk et al. 1981, Clark and Andrews 1982, Ruzicka and Conover 2011, Melville et al. 2015), American badgers (Hein and Andelt 1995, Bauder et al. 2021), kit foxes (Ralls and Eberhardt 1997, Dempsey et al. 2014), red foxes (Weber et al. 1991, Ruzicka and Conover 2011, Bauder et al. 2021), swift foxes (*Vulpes velox*; Schauster et al. 2002), coyotes (Melville et al. 2015, Bauder et al. 2021), bobcats (Melville et al. 2015), black-footed ferrets (Campbell et al. 1985; Eads et al. 2012*a*, 2015), American mink, and skunks (*Mephitis mephitis* and *Spilogale* spp.; Schowalter and Gunson 1982, Rosatte 1987, Ruzicka and Conover 2011, Bauder et al. 2021) have been determined with spotlighting surveys. These surveys usually involve 1–2 observers in a vehicle (e.g., truck, all-terrain vehicle) driving slowly (16–24 km/hr [10–15 mi/hr]), scanning the survey area for animals using spotlights of >500,000 candlepower (Fig. 8). There may be considerable variation in how spotlighting surveys are completed (e.g., number of observers, type of vehicle, handheld or vehicle-mounted spotlights, routes on or off roads). In some roadless areas, spotlighting surveys are conducted on foot, with spotlights powered by large batteries carried via backpack.

When an animal is detected, usually through light reflected by their eyes, the driver stops the vehicle and the observer(s) attempt to identify the species, sometimes with the aid of binoculars or a spotting scope. The distance traveled and time of detection are often recorded for each sighting. An index of number of animals observed/survey distance traveled is then calculated. The relationship between indices from counts of individuals via spotlighting surveys and known population densities of each of swift foxes (Schauster et al. 2002) and kit foxes (Dempsey et al. 2014) was weak, indicating that counts via spotlighting surveys were poorly related to abundance; low rates of detection for each species were the primary reason for weak correlations. Aubry et al. (2012) suggested that lack of precision in data collected during spotlighting surveys has likely been due to underestimation of variance. Bauder et al. (2021), using data collected during several decades, indicated that indices derived from counts via spotlighting surveys would be most appropriate for evaluating general trends in abundance rather than refined estimates of abundance.

Seasonal and biological limitations of spotlighting surveys were highlighted when Winchester et al. (2021) compared data collected on American mink during spotlighting surveys with data collected via floating camera stations in wetlands in Florida, USA; spotlighting surveys were only effective during high water levels when American mink moved into tall vegetation not inundated with water. In addition, detection probabilities associated with data collected by inexperienced observers can be lower compared to experienced observers or those with hunting experience (Sunde and Jessen 2013). Ability to detect may also depend on environmental conditions, especially lunar phase and season. Eads et al. (2012*b*) determined that when the moon was visible above the horizon and during August–September in South Dakota, USA, black-footed ferrets were more readily detected. Wind speed and direction can also influence detection, with higher wind speeds resulting in decreased detections (Ruzicka and Conover 2011, Sokos et al. 2015).

Count data collected during spotlighting surveys may be used to estimate population size with line-transect methodology if the perpendicular distance to the observed animal is recorded (i.e., distance sampling; Thompson et al. 1998). Transects need to be relatively lengthy (>10 km [6 mi]), and because vegetative cover and topography influence visibility, which in turn influences survey results, these variables should be considered in survey design

Fig. 8. Spotlighting surveys typically involve 1–2 observers using spotlights of >500,000 candlepower from a vehicle while slowly driving and searching the survey area for furbearing species of interest. Image courtesy of C. Thompson, Forest Service, USA.

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(O'Farrell 1987, Whipple et al. 1994, Ralls and Eberhardt 1997, Winchester et al. 2021). Surveys are conducted over several nights (repeated counts) to obtain a measure of sampling error. Large samples with replication are needed to detect changes in population size with an appropriate level of statistical power (Ralls and Eberhardt 1997). Surveys can be conducted seasonally or annually to assess population trends.

Changes in vegetative cover through time may also need to be considered for its potential influence on detection, and therefore monitoring long-term trends in an area. Spotlighting surveys may not be effective in areas containing low population densities of animals (e.g., Dempsey et al. 2014). Spotlighting surveys may also be used to acquire an estimate of the relative abundance of certain prey species (Barnes and Tapper 1985, White et al. 1996), monitor survival of young (Chipault et al. 2012), or assess space use (Eads et al. 2012*a*). However, Ralls and Eberhardt (1997) stated that the use of spotlighting surveys was not a sensitive method for assessing population abundance of certain prey species (e.g., desert cottontails [*Sylvilagus audubonii*], kangaroo rats [*Dipodomys* spp.]).

Catch Per Unit Effort

Live capture of an animal certainly gives a positive confirmation of species presence and hence distribution, assuming that animal was not a captive individual that escaped or was released. The number of animals captured/trap night can also be used as an index of relative abundance of a species. Smith et al. (2003*b*) determined population density of Blanford's foxes (*Vulpes cana*) using catch per unit effort (CPUE). Engeman et al. (2003) reported that CPUE marginally correlated with population trends of striped skunks, but track counts showed the greatest sensitivity to changes in population abundance. Matthews et al. (2011) calculated estimates of population density of fishers based on data associated with CPUE, remote cameras, and track plates. All methods detected a decline in relative abundance, but CPUE detected the decline at a lower magnitude. Live-capture methods can be expensive and labor intensive, and can be ineffective in areas with low population densities of the species of interest. In addition, standardization of capture procedures and variation among individual trappers can cause problems with this methodology.

Catch per unit effort has been used to assess the relative abundance of coyotes (e.g., Clark 1972, Davison 1980, Knowlton 1984), island foxes (*Urocyon littoralis*; Crooks 1994), kit foxes (e.g., Cypher and Spencer 1998, Dempsey et al. 2014), swift foxes (e.g., Schauster et al. 2002, Criffield et al. 2010), felids (e.g., Rolley 1987, Choate et al. 2006, Hötte et al. 2016), mustelids (e.g., Bjorge et al. 1981, Hein and Andelt 1995, Matthews et al. 2011, Yamaguchi et al. 2020), and muskrats (Bos et al. 2020). In Colorado, USA, CPUE was determined to be a valid method for estimating low-density populations of black bears (Baldwin and Bender 2012). For weasels, the number of animals captured/trap night seemed to be linearly related to population density (Caughley 1977). Schauster et al. (2002) and Dempsey et al. (2014) examined the relationship between known population densities of swift foxes and kit foxes, respectively, and indices derived from CPUE surveys; the results of both studies included low correlations between population density and indices derived from CPUE-based surveys. Wolfe et al. (2016) reported that pursuit success (number of mountain lions treed/day) of handlers with trained dogs during the non-harvest pursuit season was an informative metric of population trends of mountain lions. Allen et al. (2020) reported that trends in CPUE may be similar to population trends, but was generally dependent on sampling method and population trajectory. Bos et al. (2020) reported that CPUE data for muskrats varied consistently between seasons. Population estimates derived from CPUE have not always correlated with results from other survey methods. For example, Choate et al. (2006) relied on CPUE to estimate population abundance of mountain lions, and this estimate was different from estimates derived from other methods. In contrast, Schauster et al. (2002), Criffield et al. (2010), and Ross and Reeve (2011) reported similar estimates of population abundance from data associated with CPUE and other survey techniques, and estimates were stable across years (Skinner and Skinner 2008).

Capture-mark-recapture

A useful method for estimating population abundance of furbearing species is CMR; this method involves capturing and marking individuals in a defined area, then capturing a set of individuals in that same area to estimate population size based upon the ratio of marked to unmarked animals (e.g., Pollock 1981, Seber 1982, Montgomery 1987). While mark-recapture can be time consuming, labor intensive, and costly, it provides a reliable estimate of population size (i.e., absolute abundance) for many furbearing species. However, low rates of recapture can yield unreliable estimates of abundance because sparse data will result in wide confidence intervals.

CMR has been used to reliably estimate abundance of canids (Clark 1972, Todd et al. 1981, Roemer et al. 1994, Schauster et al. 2002), felids (Quinn and Parker 1987), mustelids (e.g., Messick and Hornocker 1981, Douglas and Strickland 1987, Rosatte 1987, Strickland and Douglas 1987), and procyonids (Sanderson 1951*b*, Kaufman 1987). CMR can provide relatively accurate estimates of population size if sample sizes are adequate, data collection techniques are unbiased, and the basic assumptions for the population estimator are not violated (see Caughley 1977, Wilson et al. 1996, Thompson et al. 1998). Schauster et al. (2002) reported that estimates of population abundance of swift foxes from CMR-based surveys were highly correlated ($r = 0.71$) with estimates of population density (determined independently from radio-collared animals), and that CMR was the most appropriate method for monitoring abundance among 6 techniques evaluated.

Types of marks used to identify animals include ear tags, radiocollars, dyes, and physiological markers such as radioactive isotopes. Recapture of marked animals may involve physical recapture of the animal, resighting of the animal (Todd et al. 1981, Miller et al. 1997), identification of animals harvested by trappers and hunters (Sanderson 1951*b*), recapture via fecal analysis for a physiological marker, or a combination of these methods. Use of genetic techniques to initially identify (mark) and identify individuals after some period of time (recapture) from samples (e.g., scats, hair, urine) is becoming increasingly important for

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monitoring furbearing species (Reding et al. 2023 [Chapter 16]). Kohn et al. (1999) estimated population size of coyotes by identifying individual animals through DNA analysis of fecal samples combined with mark-recapture methodology. Boulanger et al. (2018) used SECR methods to explore factors influencing population density and distribution of grizzly bears. Azad et al. (2019) used hair snares and SECR models to estimate population density of black bears. Similarly, Roffler et al. (2019) used hair snares and SECR models to determine abundance of gray wolves, particularly in a densely forested area where visual surveys would be impossible.

Several different models for population estimation (e.g., Petersen, Jolly-Seber, Schnabel, SECR) can be used to calculate population size (Caughley 1977, Jolly 1982, Seber 1982, Thompson et al. 1998, Efford 2023). If the area of interest or capture effort is known, then estimates of population density can be derived. Reviewing capture-recapture methodologies (e.g., Caughley 1977, Thompson et al. 1998, Efford 2023) can assist biologists in study design prior to implementation. Various capture methods have been used with mark-recapture estimators. Roemer et al. (1994) used a trapping grid to estimate population size of island foxes. Clark (1972) captured and marked coyote pups at dens in the spring, then conducted trapping sessions to recapture pups during late summer.

Prior to the use of genetic techniques, physiological markers have been used to mark animals and then later identify marked animals (recapture) to estimate population abundance with mark-recapture estimators. This method involves capture of the animal, injection or oral dosing of the marker (e.g., iophenoxic acid, rhodamine B [Knowlton et al. 1988], chlorinated benzene [Johnston et al. 1998]) into the animal, then resampling the animal at a later date either by direct recapture and blood sampling, collection of labeled scats, or examination of harvested animals. Radioactive isotopes have been used to determine population densities of furbearing species (Pelton and Marcum 1977, Kruuk et al. 1980). Radioactive zinc has been injected into captured European badgers, with the isotope later detected in feces to estimate the population size from the ratio of radiolabeled to unlabeled feces (Kruuk et al. 1980, Kruuk and Parrish 1982). Kruuk et al. (1993) used radioactive isotopes to mark spraints and then identify which individual Eurasian otter deposited that spraint.

With the added responsibility and permitting needed to handle and store radioisotopes, researchers have examined other compounds to serve as individual markers for carnivores. Knowlton et al. (1988) reported that oral doses of iophenoxic acid were detectable or traceable in coyotes ≤16 weeks post ingestion. Johnston et al. (1998) tested the use of chlorinated benzenes as physiological markers for coyotes and stated that injection or ingestion (oral dose) of some compounds were detectable ≤100 days later in feces and blood serum. Fisher (1995) provided a review of the use of rhodamine B as a biomarker and reported detectable levels in several furbearing species, depending on species, dosage, and tissue sampled (e.g., blood, hair). Tetracycline has also been used as a biomarker in black bears and polar bears (e.g., Garshelis et al. 1990, Taylor and Lee 1994). Biomarkers have been used to estimate population abundance of canids (Davison 1980, Knowlton 1984), mustelids (Kruuk et al. 1980, Kruuk and Parrish 1982, Knaus et al. 1983, Melquist and Dronkert 1987), ursids (Garshelis and Noyce 2006), and procyonids (Conner et al. 1983, Conner and Labisky 1985).

Direct Counts by Removal from a Population

For furbearing species that are abundant or considered pests, the removal method (i.e., permanent removal of individuals within a species from the population through human-caused mortality) has been used to estimate population abundance (e.g., Skalski et al. 1984, Rosatte 1987). Disadvantages of this technique are the lack of knowledge of the proportion of the population that was not included or not captured, the size of the area affected by the removal efforts, and the consequences that the resulting estimate is associated with the population size at initiation of removal efforts rather than the current population size. Due to the economic importance of furbearing species, intrinsic values, and the social and political ramifications, the removal method is rarely employed, perhaps with management of non-native species being the exception.

Transect, Strip, or Area Sampling

In certain circumstances, it may be possible for a biologist or manager to directly count the number of animals along transects or strips, in quadrants, or within a defined area and estimate population size or density (Burnham et al. 1980, Rao et al. 1981, Bibby et al. 1992). While transect-based and quadrant-based surveys are commonly used for estimating population abundance of ungulates, some of the larger-bodied furbearing species may be surveyed with this technique. Trends in relative abundance can be compared from direct counts; absolute abundance may be estimated if correction factors are developed to account for problems with sightability or probability of detection (Samuel et al. 1987). Population estimates can also be calculated by distance methods along line transects (Burnham et al. 1980).

Aerial-based surveys for furbearing species typically are effective only for large-bodied species that occupy relatively sparsely vegetated systems to allow for maximum sightability of individuals. Aerial-based surveys have been used to estimate abundance of coyotes (e.g., Nellis and Keith 1976, Todd et al. 1981). The Interagency Grizzly Bear Study Team (2020) annually estimates the number of female grizzly bears with cubs of the year via aerial-based surveys and opportunistic ground observations. The number of animals observed can be affected by weather, vegetation structure, visibility (e.g., terrain, snow cover), type of aircraft (airplane vs. helicopter), and observer experience and fatigue. Miller and Russell (1977) compared counts of gray wolves collected during aerial-based transect-strip surveys and reported that the behavior of the animals, width of the survey strip, and visibility (e.g., lack of color contrast between wolf pelage and the background) all contributed to unreliable estimates of population abundance using aerial-based surveys in open tundra. The primary issue was that during periods when no wolves were observed during the aerial-based survey, wolves were observed during the ground-based survey.

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The use of ultraviolet, infrared, and thermal imagery has been proposed for enhancing sightability of some species during aerial-based surveys (Havens and Sharp 1998). Ground-based surveys, including coupled with FLIR cameras (T. Hiller, Wildlife Ecology Institute, unpublished data), are practical for smaller-bodied species of furbearers that can be readily viewed in sparsely vegetated areas (Fig. 9). Population trends for white-nosed coatis (*Nasua narica*) were estimated from visual counts during transects walked by observers (Kaufman 1987). In certain situations, the entire area of interest may be surveyed, and through repeated sampling and reobservation, all individuals within a population may be counted. For example, gray wolves on Isle Royale, Michigan, USA, have been observed and counted for decades, with every pack located and counted on the island each winter (Jordan et al. 1967, Wolfe and Allen 1973, Peterson et al. 1998). However, the ability to count all individuals in a defined area is a rare circumstance due to animal movements across large areas, and other factors. Alternatively, correction factors from a radio-marked sample can be used to determine a more accurate estimation of population size. Inclusion and determination of the level of precision (e.g., confidence or credible intervals) associated with a population estimate is also necessary for assessing accuracy of the survey. Population estimates with wide confidence intervals indicate a high level of variation in the counts and less ability to predict small changes in the population.

Visual Identification of Individual Animals

While the opportunity to directly observe several species of furbearers is rare, there are certain species living in national parks or reserves with sparsely vegetated landscapes that allow for direct observation and identification of all individuals in the study area. This technique has been used successfully in studies of large carnivores in Africa (e.g., Pennycuick and Rudnai 1970, Bertram 1975, Hanby and Bygott 1979). Similarly, identification of individual spotted hyenas (*Crocuta crocuta*) by distinct pelage patterns, scars, and ear notches (East and Hofer 1991) has been used to determine population size (Hofer and East 1995). Throat patches have been used to identify individual Eurasian otters (Watt 1993). Individual coyotes in Yellowstone National Park, USA, were identified through marks (e.g., radio-collars, ear tags) and phenotypic characteristics (e.g., unique individual pelage markings), which permitted determination of pack size, and hence, population size (Gese et al. 1996). The Interagency Grizzly Bear Study Team (2020) monitored population trends of grizzly bears by documenting sightings of females with cubs-of-the-year, paired with criteria (e.g., spatial and temporal characteristics, litter size) to differentiate unique family groups.

Fig. 9. Remotely operated Infrared cameras have been externally mounted to vehicles and connected to internally mounted monitors to aid with surveys of black-footed ferrets (Mustela nigripes) in Montana, USA. Image courtesy of Wildlife Ecology Institute, USA.

Monitoring through direct observation and identification of individuals generally occurs in sparsely vegetated landscapes and for a species or population that is readily observable and tolerant of human presence. Animals do not necessarily need to be marked for individual identification, as individuals may be resighted and identified indirectly. Track characteristics and location of mountain lions have been used to identify individuals, and data then combined to provide an estimate of population density (e.g., Ackerman et al. 1981, Van Dyke et al. 1986, Smallwood and Fitzhugh 1995). Alibhai et al. (2017) used data-visualization software to generate 123 measurements for each footprint from a set of standard images of 535 footprints from 35 captive mountain lions, and reported a classification accuracy of 90% for individuals and 99% for sex. The primary advantage of using characteristics of individual tracks for identification was that it entailed less field effort than a large-scale capture effort, but the accuracy of this method in relation to changes in population size remains untested. While individual identification allows for a relatively complete count of animals, the time and effort for this type of monitoring avails itself only to particular situations and is often conducted in conjunction with behavioral studies (e.g., Gese et al. 1996).

Noninvasive Genetic Sampling

NGS involves the collection of genetic material from samples that have been deposited (e.g., fecal, saliva on prey remains) in the environment, or noninvasively removed (e.g., hair from snags) from animals (Waits and Paetkau 2005, Reding et al. 2023 [Chapter 16]). Traditional approaches for monitoring furbearers (e.g., scat-deposition surveys, hair snares, scent-detection dogs) can be extended with NGS to analyze DNA within samples to confirm species and identify unique individuals, allowing for cost-effective application of occupancy modeling and capture-recapture approaches for monitoring populations (Schwartz et al. 2007). Beyond facilitating estimates of traditional parameters of population demographics, genetic information can also be used to evaluate genetic parameters associated with populations, such as genetic diversity, genetic structure, effective population size, and parentage (Schwartz et al.2007). As mentioned previously, one may be able to extract DNA of the species of interest from surfaces or substrates, or within mediums (e.g., water); these sources of DNA are commonly referred to as eDNA. For example, eDNA collected during snow-track surveys and snow collected at remote cameras was used to detect and identify 3 rare carnivores within a forested system (Franklin et al. 2019). Use of eDNA from water samples may help identify species presence for many semi-aquatic furbearing species.

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Genetic identification of sign (e.g., scats, tracks) with mitochondrial DNA (mDNA) can help resolve uncertainty in species identification, and provide more reliable estimates of population parameters. For example, misidentification of scats between sympatric species tend to be biased towards rarer species (Davison et al. 2002, Lonsinger et al. 2015*a*). Although these patterns of species misidentification can lead to biased estimates and erroneous conclusions, genetically based identification of species can help alleviate these issues due to its high level of accuracy (Lonsinger et al. 2021). Noninvasive genetic identification of individuals with nuclear DNA (nDNA) can represent natural marks and be used within CMR frameworks to estimate population demographics (e.g., population abundance, vital rates) without physically capturing or handling animals. Capture with replacement models (CAPWIRE) have been developed specifically for CMR of noninvasive genetic samples where an individual can be captured (i.e., genetically identified) more than once during a single sampling event (Miller et al. 2005, Kluever et al. 2022), but may not provide reliable estimates for wide-ranging and territorial species of furbearers (Lonsinger et al. 2019). When compared to live-capture approaches for estimating population demographics, NGS has many benefits, including lower stress to the animals, fewer risks to biologists, fewer permitting requirements, and lower cost/sample (Waits and Paetkau 2005, Schwartz et al. 2007).

Monitoring furbearer populations through NGS of scat has become common for canids (e.g., gray wolves, Stansbury et al. 2014; coyotes, Prugh et al. 2005, Kluever et al. 2022; swift foxes, Cullingham et al. 2010; kit foxes, Lonsinger et al. 2018*a*), felids (e.g., bobcats, Ruell et al. 2009; ocelots [*Leopardus pardalis*], Wultsch et al. 2015), and, to a lesser extent, mustelids (e.g., North American river otter, Brzeski et al. 2013). NGS of hair (e.g., via snares or snags) is commonly used for ursids (e.g., black bears, Gould et al. 2019; grizzly bears, Kendall et al. 2019) and mustelids (e.g., American badger, Kierpka and Latch 2016; American marten and fishers, Williams et al. 2009), and to a lesser extent for other furbearing species. Less common sampling approaches that have facilitated species identification via noninvasive genetic samples have included sampling the surface of tracks in snow (e.g., Canada lynx, fishers, wolverines; Franklin et al. 2019) and sampling water for semi-aquatic furbearing species (e.g., North American river otters; Sales et al. 2019).

The ability to collect viable DNA from the environment depends on several factors. First, the number of samples (e.g., scats) available for collection (hereafter, sample accumulation) is influenced by deposition rates of DNA and removal rates of samples (e.g., loss or destruction of scats by vehicles; Lonsinger et al. 2015*b*). Rates of sample accumulation may vary spatially (e.g., near dens or latrines) or temporally (e.g., due to seasonal variation in diets; Andelt and Andelt 1984). Sample removal can occur due to anthropogenic disturbances, interspecific interactions, or weather (e.g., substantial amounts of precipitation, high wind speeds). Detection (or acquisition) rates of samples that remain available may vary due to sampling approach (e.g., visual searches for scats versus the use of scat-detection dogs) or device (e.g., alternative snaring approaches). For samples that remain available for detection and sampling, degradation of DNA can limit sample viability for NGS. Careful consideration of rates of sample accumulation and degradation of DNA is critical for establishing reliable NGS-monitoring programs and pilot studies to aid biologists and managers with optimizing sampling designs (Waits and Paetkau 2005, Lonsinger et al. 2015*b*, Kluever et al. 2022).

Sources of noninvasive genetic samples typically provide low quantities of low-quality DNA. To maximize efficiencies and success, biologists employing an NGS approach will need to consider factors influencing degradation of DNA, potential impacts of degradation, and techniques for sample preservation to slow degradation. Patterns of degradation of DNA vary among study systems, but general patterns indicate that degradation increases with increasing ambient temperature, humidity, and precipitation; increasing level of exposure to ultraviolet light; and increasing time since deposition (Piggott 2004, Santini et al. 2007, DeMay et al. 2013, Lonsinger et al. 2015*b*, Kluever et al. 2022). The rate of degradation of DNA is typically higher for longer sequences of DNA, with smaller loci having higher rates of success for amplification (Buchan et al. 2005). The characteristic of mDNA as relatively more abundant than nDNA in a cell tends to lead to lower rates of degradation and higher rates of successful amplification for mDNA used for species identification compared to nDNA used for individual identification. Degradation of nDNA can lead to genotyping errors (i.e., false alleles or allelic dropout), generate uncertainty in identifying individuals, and generate biases in estimates of population parameters from CMR.

Although there are analytical approaches to identify and minimize the inclusion of genotyping errors (e.g., evaluating the reliability of genotypes, Miller et al. 2002; establishing consensus genotypes via replication, Broquet and Petit 2004), and to reduce biases introduced by uncertainty in individual identification (e.g., Lukacs et al. 2009, Augustine et al. 2019), minimizing factors affecting degradation of DNA during the sampling-design phase of a monitoring program can limit genotyping errors (Lonsinger et al. 2015*b*). Appropriate preservation techniques in the field can decrease the degradation rate of DNA for noninvasive samples, and biologists and managers should coordinate with a genetic laboratory to identify preferred sample-source-specific preservation approaches. Common preservation methods for fecal DNA include DET buffer (20% DMSO, 0.25 M EDTA, 100 mM Tris, pH 7.5, and NaCl to saturation) solution, 95% ethanol solution (EtOH), or storage in a freezer (Seutin et al. 1991, Frantzen et al. 1998), whereas hair samples are commonly stored with silica desiccant or frozen (Roon et al. 2003).

Telemetry

The use of telemetry allows researchers to estimate the home-range size or territory size of an animal, determine a correction factor for surveys based on the proportion of marked animals counted, monitor survival and cause-specific mortality, and assess detailed movement and spatial information for addressing a variety of questions (e.g., resource selection, step selection, immigration, dispersal). Very high frequency (VHF) telemetry units transmit a pulsed signal, and telemetry units with GPS (typically also with VHF capabilities) can store location data on the unit (i.e., for download when the unit

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is recovered or via wireless download when in close proximity to the unit) or transmit data via satellite. Global System for Mobile (GSM) communications units can transmit data through cellular phone networks. The weight, configuration, methods of attachment (e.g., collars, implanted transmitters), and battery longevity of transmitters continue to improve, as well as the large number of companies now producing VHF and GPS transmitters for use in management and research for a variety of furbearing species. High failure rates (e.g., failure of GPS-related components) of some transmitters, particularly small GPS-based units, are still prevalent, but advances in technology and battery life will presumably result in more reliable products.

Current technologies limit GPS transmitters for the smallest species of furbearers to store-on-board units, which records and stores location (and other) data in the transmitter until those data can be retrieved by either recovering the transmitter or remotely downloading via a handheld base station and ultra high frequency (UHF) signal, the latter of which typically requires being within a certain line-of-sight distance (e.g., about 200–500 m [650–1,650 ft], depending on transmitter model) of the radio-marked animal. Recent developments have produced a 50-g GPS collar capable of storing approximately 6 months of locations (with 2 GPS fixes/day), but does require recapture of the animal to download the data from the collar. Hopefully, as the amount of battery life increases, the ability to transmit locations will not necessitate recapture of the radio-marked animal, but current technology limits the amount of battery power required for uploading data.

The addition of a drop-off mechanism has the advantage of retrieving store-on-board transmitters and removal of transmitters from animals at the end of the study, or when reaching the limits of data storage. Transmitters with the ability to determine how close animals are to one another (proximity sensor) are used to examine social cohesion and pair-bond behavior. Transmitters with accelerometers provide activity data that can be used to identify behavioral states (e.g., resting) and sites where prey species have been killed, estimate energetic costs of movement, and quantify the orientation of an animal's head (Wilmers et al. 2015). Acoustic recorders have been added to radio-collars (e.g., Canada lynx; Studd et al. 2021) to further understand feeding behaviors. There is increasing interest in also adding cameras to collars, but to date, the battery and weight requirements have limited applications of animal-borne cameras to large-bodied animals (e.g., ursids; Brockman et al. 2017). With the continued use of satellite and GPS technology, intensive monitoring of furbearing species has taken tremendous leaps forward, and the volume of data collected is creating new areas of spatial analysis. Additionally, intraperitoneal transmitters are proving effective for monitoring survival of young, as well as development of new designs of harness attachment.

For some furbearing species, combining territory size and overlap between or among territories with the number of members of the social unit, plus the percentage of radio-marked transients sampled from the population, estimates of population density can be derived for the population of interest. Because canids tend to be highly social with well-defined territories, telemetry is widely accepted as a method to measure population size and density (e.g., Mech 1973, Fritts and Mech 1981, Fuller 1989, Gese et al. 1989). For more solitary species, estimates of home-range size, the extent of intersexual and intrasexual home-range overlap, and the proportion of transients in the population are used to estimate population density. This method has been used for felids (e.g., Rolley 1987, Quinn and Parker 1987), mustelids (e.g., Melquist and Hornocker 1979, Hornocker and Hash 1981, Magoun 1985, Douglas and Strickland 1987, Strickland and Douglas 1987), and procyonids (e.g., Lanning 1976, Russell 1979, Lacy 1993). While telemetry can be labor intensive and costly, this technique provides one of the most reliable and accurate estimates of population density for many species. Long-term studies using telemetry provide the most reliable annual estimates of population density for several secretive, far-ranging, low population density species of furbearers, such as Canada lynx (Quinn and Parker 1987) and wolverines (Magoun 1985).

Validation of Survey Methods

Regardless of the technique used to assess the abundance of a furbearing species, there needs to be some assurance the technique produces estimates that adequately reflects a measure of the population. In the case of a survey, the process may be as simple as repeating it several times within a brief period to determine the amount of variation among the estimates. The problem becomes more complex when indirect measures (indices) are used to assess relative abundance which may require research to determine how accurately the indirect measures relate to true abundance, and may involve testing the indirect measures with a known population size of animals.

Some form of mark-recapture approach is typically required to determine population abundance or density in one or more areas of interest (e.g., Schauster et al. 2002, Dempsey et al. 2014). This needs to be repeated several times with different population densities to determine the relationship between the index and the abundance of animals. As an example, Linhart and Knowlton (1975) described an index procedure for coyotes, with biologists identifying tracks at scent stations annually at >400 sampling points for 10 years in the western U.S. Subsequent investigation (Windberg and Knowlton 1988, Harris and Knowlton 2001) revealed that coyotes were more likely to leave tracks at scent stations away from their commonly used areas of activity. This did not negate the results of surveys if acknowledging the assumption that the proportion of animals was constant in less commonly used areas. Ultimately, when this technique was tested against a known population size, a crude relationship was established, but not nearly as strong of a relationship as determined using rates of scat deposition (Knowlton 1984). This merely confirms the necessity of validating the utility of the technique used.

Investigating whether the relationship between indices of relative abundance and absolute abundance is positively and monotonically related, or whether the relationship is nonmonotonic, can aid biologists and managers in interpreting indices (Gese 2001). Is the relationship linear with a constant slope, or linear with a variable slope? Indices that are nonmonotonic to animal abundance

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are of little use for monitoring trends of a population. Often times, indices are compared to other indices without knowing the true population size or density (Gese 2001). Comparing an inexpensive indirect method to a more expensive direct method could prove worthwhile for calibration of the less expensive technique. For example, Schauster et al. (2002) demonstrated that population size of swift foxes estimated from a mark-recapture approach was the most appropriate method (r = 0.71) for monitoring population density, but cost was US$1,427/10-km- (6.2 mi) long transect. In contrast, scat-deposition surveys were the second-most appropriate method (r = 0.70) for monitoring population density, but cost was US$160/10-km- (6.2 mi) long transect; a substantial savings but with a similar reduction in precision and accuracy.

Monitoring a population using multiple methods is superior to a single method, particularly if the basic assumptions may be violated (Gese 2001, Campbell et al. 2008, Burr et al. 2017). For example, combining track plates with other monitoring techniques, including CMR, live trapping, spotlighting surveys, or remote cameras, proved useful for identification of several species (e.g., Barrett et al. 2012, Loughry et al. 2012, Riem et al. 2012, Jordan and Lobb-Rabe 2015, Mellville et al. 2015). Studies determining which method is most applicable to the species or system need to be investigated (Gese 2001). For example, Alldredge et al. (2019) used predator calls to attract mountain lions along the urban-wildland interface to sites with remote cameras and hair snags; the combination of NGS with an auditory call was reliable for estimating population density. Schauster et al. (2002) showed the combination of mark-recapture and scent-station methods provided information that was a good predictor (r = 0.85) of population density of swift foxes. Population indices based on the combination of two non-invasive techniques (scent stations and scat-deposition surveys) were almost as accurate (r = 0.83), but cost far less than a mark-recapture approach and did not necessitate capturing and marking animals. During such a calibration, the techniques should be performed concurrently and may need to be conducted on a species-specific, habitat-specific, and seasonal basis (e.g., Schauster et al. 2002). Unfortunately, few indices of relative abundance have been properly compared with a known population size (Gese 2001).

Trend Analysis

A common goal of monitoring programs is to determine change, or sustained trends in population size over time (e.g., Gerrodette 1987, Kendall et al. 1992). Nearly all types of population measures (e.g., frequency of occurrence, relative density, absolute abundance, vital rates, several genetic measures) can be used to examine population trends. While frequency of occurrence data may often be easier to obtain, relative and absolute abundance and relative population density provide greater sensitivity to change and require smaller sample sizes to detect change than frequency of occurrence data (Vesely et al. 2006). Museum records and historical publications could prove useful in revealing large-scale and long-term changes in distribution and abundance of a furbearing species.

Population trends are most often analyzed using generalized linear regression models, in which the trend over time is related to trends in other variables. However, although a simple count seems straight-forward, counts vary annually for many reasons. An increase in estimated population size from one year to the next may not reflect a true increase in the population, but rather differences in survey technique, changes in sightability, or normal cycles in environmental and demographic stochasticity. Too few counts or counts that are too variable can prevent detection of a population trend when a trend is actually occurring. While a series of at least three counts is needed to assess a trend, in practice it is difficult to reliably detect trends with less than five counts (Elzinga et al. 2001). Thomas (1996) thoroughly reviewed four regression models used for evaluating trends, and the assumptions associated with each approach, and addressed factors that complicate analysis of trend data, including observer bias and missing data.

MONITORING POPULATION DEMOGRAPHICS

The previously described survey methods provide information about quantifying a population, but do not necessarily answer questions about why the population trajectory is increasing, decreasing, or stationary. To do this, one must quantify the demographic rate functions of survival, fecundity, immigration, and emigration, all which influence the trajectory of a population. In this section, we summarize the important parameters necessary to understand these demographic processes. Because most of the actual techniques used to measure survival, fecundity, immigration, and emigration are species specific, for the scope of this chapter, we will only provide a listing of the various measures one may want to monitor. For practitioners embarking on a study of population dynamics of furbearing species, excellent resources are available to help ensure successful collection of the proper data for demographic analyses (e.g., Caughley 1977, Royama 1992, Thompson et al. 1998, White and Garrott 1990, MacKenzie et al. 2018, Clark and Powell 2023 [Chapter 5]).

Fecundity

The fecundity rate of a female is the number of offspring produced over an interval of time (Caughley 1977). Measuring fecundity or reproduction is relatively complex and time consuming. However, there are several basic questions associated with fecundity that biologists may ask, including: 1) when does the breeding season start and how long does it last, both in terms of estrous and gestation; 2) when are the young born; 3) what proportion of the females in the population breed; 4) how many young are produced; 5) is there one (monestrous) or multiple (polyestrous) breeding seasons in a year; 6) what is the sex ratio at birth; and 7) what is the age of first reproduction? There are various techniques to answer these questions. For furbearing species, collection of carcasses, recovery of tagged animals, and observations in the field or captivity may address these questions. More specifically, examination of ovaries (counts of corpora lutea) and counts of placental scars from recovered or harvested animals, the ratio of juveniles to females in harvest data, or observation of litter size in the field will provide some measure of reproductive

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output (e.g., age-specific fecundity). Behavioral observations of animals in the field or captivity, physical examination, or tissue histology may provide information on initiation and cessation of the breeding season, and age of first breeding or sexual maturity.

Survival

Measuring the survival rates of furbearing species usually involves construction of a life table or estimation of survival from telemetry data. There are several pertinent questions that a biologist may consider when designing a study to address survival rates, including: 1) what is the number of mortalities in each age class; 2) what is the probability of mortality in each age class; 3) does the rate of mortality vary among seasons; and 4) what are the causes of mortality? Ages from animals collected from hunters and trappers can be used to construct life tables (e.g., Caughley 1977, Clark and Powell 2023 [Chapter 5]). Measuring the number of radio-days and number of mortalities during defined time intervals derived from radio-collared animals can be used to estimate daily (and other intervals) rates of survival, and assess cause-specific mortality (e.g., Trent and Rongstad 1974, Heisey and Fuller 1985, White and Garrott 1990). Age-at-harvest (or stage-at-harvest) data provide an alternative approach to estimating survival without marking animals. For example, Skelly et al. (2023) used counts of cementum annuli in teeth to estimate the age of harvested bobcats, and developed a novel age-at-harvest model to generate estimates of survival from harvest data that were both realistic and consistent with estimates generated from radio-collared animals and more established known-fate models.

Immigration and Emigration

Measuring rates of emigration and immigration within a population usually involves the capture and marking of several individuals and the subsequent recapture or monitoring (e.g., telemetry) of those individuals (Gese 2001). Genetic techniques can also be used to mark and recapture individuals if sampling an area of sufficient size for emigration and immigration to be measured. Monitoring the movements of animals out of a marked population (e.g., dispersal) is simpler than monitoring movements into the population because biologists cannot predict where immigration will occur from outside the known study population (Gese 2001). Thus, biologists typically assume the rate of movement out (egress) of their study area is equal to the rate of movement into (ingress) the study area. This assumption is usually violated, particularly if one of the populations is receiving some form of management. Whether the population being studied is maintained as a source or a sink is pertinent to understanding the system and managing the population of furbearing species of interest.

Immigration (or emigration) may also be identified or quantified using genetic data. Analyses of the genetic structure of populations can be used to identify potential migrants (i.e., animals that assign genetically to a different population from where they occur; Pritchard et al. 2000). Riley et al. (2006) identified potential migrants within populations of bobcats and coyotes based on genetic samples collected during a single capture and handling event for each species. Migrants in bobcat populations have also been detected by applying analyses of genetic structure of populations to samples from harvested animals (Reding et al. 2013). Similarly, even when potential source populations have not been sampled, the likelihood of an individual's genotype within their population can be used to identify recent immigrants and calculate immigration rates (Rannala and Mountain 1997). For example, Lonsinger et al. (2018*b*) calculated the probability that an individual kit fox within a population was an immigrant, and estimated immigration rates using noninvasive genetic samples for the contemporary population and museum specimens or the historical population.

Disease Monitoring

The role of disease in population dynamics is often overlooked when monitoring furbearing species (Gese 2001). With an increasing interface between furbearers and humans and their pets, livestock, and expanding development, the possibility of disease transmission continues to escalate. Exposure to disease agents can have dire consequences for rare or endangered species. For example, canine distemper virus caused a rapid decline in populations of black-footed ferrets, and almost caused the species to become extinct (Williams et al. 1988). Similarly, rabies has been implicated in the decline of African wild dogs (*Lycaon pictus*; Woodroffe and Ginsberg 1997). Biologists initiating a study may need to implement a disease-monitoring program and handling protocol (for animals and samples collected), especially for plans to reintroduce or translocate a species, or for a rapidly declining population. Physical examination of living animals, sample collection for serology and other assays, and post-mortem examinations of harvested animals and recovery of radio-marked animals can be used for monitoring health and disease. Wildlife veterinarians affiliated with a diagnostic laboratory or university can provide guidance about which diseases to prioritize for screening and methods of sample collection and curation when designing a monitoring program (see Gillin et al. 2024 [Chapter 7]).

Population Modeling

Demographic variables, such as survival, fecundity, and age structure can be used to model population trends of various furbearing species (e.g., Connolly 1978, Mowbray et al. 1979, Sterling et al. 1983, Clark and Powell 2023 [Chapter 5]). These models can then be used to simulate the population response when one or more demographic variables is manipulated. Population viability analysis (PVA) and population and habitat viability assessment (PHVA) are used to evaluate the impact of various management actions, environmental perturbations, and stochastic events on the population viability of a species over a predetermined period of time (e.g., Shaffer 1981, Boyce 1992, Reed et al. 1998, Kelly et al. 1999).

Biologists and managers using PVA and PHVA models should consider the realism of these models and ensure the models are adaptive in response to changes in ecological, environmental, and management factors (Williams et al. 2001). A PVA or PHVA is only a model and may not actually reflect or predict population persistence, and thus should not be the primary tool for developing a conservation plan.Prior to their use, researchers should evaluate the accuracy, sensitivity, and uncertainty of the data integrated into the model (Reed et al. 1998). Macdonald et al. (1998)

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reported that PVAs seemed most useful for guiding management actions and identifying practical monitoring methods. Some PVAs and PHVAs are most appropriately used to raise questions and formulate hypotheses for future testing (Macdonald et al. 1998, Reed et al. 1998, Gese 2001).

Population modeling can be challenging for species that are difficult to monitor, including furbearers, or where alternative forms of data may be available across different spatial or temporal scales (e.g., number of individuals harvested, age-at-harvest data, camera-based data). Analytical advancements offer approaches to integrate data from multiple approaches while accounting for uncertainty related to process and observation variability. Integrated population models (IPMs) provide a unified analysis framework linking changes in population count data with demographic data to better estimate population dynamics (Schaub and Abadi 2011). By combining multiple types of data (often from independent studies), IPMs can overcome challenges created by sparse data and may provide estimates of parameters for which direct data are not available (Zipkin and Saunders 2018). For example, Horne et al. (2019) used IPM to combine pack counts with survival estimates from GPS-collared gray wolves to estimate pack size, rates of harvest and non-harvest mortality, dispersal, and recruitment; they were also able to estimate pack sizes for periods when count data were not available.

Occupancy and Occupancy Dynamics

Estimates of abundance and associated vital rates (i.e., fecundity, survival, immigration, emigration) can be difficult to reliably estimate, particularly for species that are rare or elusive, such as many furbearing species. In these cases, occupancy can serve as alternative state parameter of interest for biologists and managers. Occupancy represents the proportion of area occupied by a species, and when an appropriate probabilistic sampling design is employed, the probability that the species of interest occupies a randomly selected (unsampled) site (MacKenzie et al. 2002).

Occupancy modeling can also be used to infer the influence of environmental factors on patterns of occurrence, while accounting for imperfect detection (i.e., detection rates <1). With appropriate considerations of sampling design, any survey technique for furbearers that can reliably detect the presence of a species can be used to estimate patterns of occupancy. For example, occupancy modeling requires survey replication within sample sites to disentangle the ecological process driving the distribution of a species from the detection process, which influences the observed encounter history of the species (MacKenzie et al. 2002).

Survey replication (i.e., multiple surveys of each site within a primary sampling session, or season, over which closure is assumed) can be achieved in many ways, including repeat visits to each site, multiple spatially replicated surveys within each site, or multiple independent observers during single site visit (MacKenzie et al. 2018). When conducted over multiple seasons, dynamic (or robust design) occupancy modeling can be used to estimate dynamic parameters of occupancy, including the probabilities of site colonization and local extinction (MacKenzie et al. 2003, Lonsinger et al. 2017).

Extensions to occupancy modeling have greatly increased the utility of occupancy for monitoring. For example, multi-state occupancy analyses permit inference on factors driving spatial variation in relative abundance for a species, relative habitat quality, and disease ecology (Nichols et al. 2007, Bailey et al. 2014, Reddell et al. 2021). Multi-scale occupancy analyses have facilitated comparisons of competing sampling approaches for furbearing species (e.g., remote cameras, hair snares, track plates; Nichols et al. 2008). Occupancy analyses formally evaluating patterns of co-occurrence between or among ≥2 species (Richmond et al. 2010, Rota et al. 2016) have been used to understand species interactions and the role of interspecific interactions on the occurrence of furbearing species (e.g., Robinson et al. 2014, Green et al. 2018). These extensions, as well as others, and the assumptions for each are detailed by MacKenzie et al. (2018).

BEYOND MONITORING

Once a monitoring program is in place, practitioners may consider whether their program could provide the opportunity for examining other basic or applied questions of interest, such as food webs, trophic cascades, and community interactions (Sauer and Knutson 2008). Often times, monitoring programs are too one dimensional and simply focus on sampling the population of interest without examination of the trophic levels and community interactions influencing the population of interest.

If limited resources prohibit a more extensive study, then maintaining a basic monitoring program is still of value. However, there are often times when a researcher would like to expand a study into other areas of scientific inquiry. For example, once a monitoring program for swift foxes had been developed and tested in Colorado (Schauster et al. 2002), research to determine the role of coyote predation on population dynamics of swift foxes could then be conducted (Karki et al. 2007). Further expansion of these monitoring activities, combined with protocols for estimating the abundance of the prey base and other predators, plus documentation of vegetation structure, allowed for an expanded investigation into the food web and community interactions influencing this population of swift foxes (Thompson and Gese 2007). In addition, Kitchen et al. (2005, 2006) used genetic samples collected from swift foxes during capture to examine the spatial, breeding, and social ecology of this relatively unstudied canid.

Collecting accurate and reliable knowledge is paramount to inform management decisions that support long-term conservation and sustainable harvest of furbearer populations. Successful monitoring programs require thorough consideration of statistical, biological, logistical, political, ethical, social, and economic factors during the planning process (Gese 2001). Noninvasive techniques are becoming more prevalent for monitoring furbearing species (e.g., Long et al. 2008), and will continue to be important for monitoring rare, threatened, and endangered species, particularly when capturing and handling efforts could jeopardize the health and welfare of a species.

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Use of multiple techniques for monitoring carnivores should always be considered to reduce erroneous conclusions based upon a single methodology (Gese 2001, Schauster et al. 2002, Campbell et al. 2008). Researchers must always be evaluating sampling procedures to minimize direct and indirect impacts on the furbearing species being studied. Careful thought and planning will help avoid problems in the future. It is our hope that this chapter provides some food for thought and questions that should be considered when designing and implementing a monitoring program.

ACKNOWLEDGMENTS

We thank T. C. Atwood and B. M. Kluever for helpful reviews of the manuscript. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government. The findings and conclusions in this publication have not been formally disseminated by the U.S. Department of Agriculture and should not be construed to represent any agency determination or policy. The findings and conclusions of this article are those of the authors and do not necessarily represent the views of the U.S. Fish and Wildlife Service. However, the findings and conclusions of this publication do represent the views of the U.S. Geological Survey.

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