10 Manage Habitat, Monitor Species

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Monitoring is the collection of data over time. We monitor many things: temperatures at local weather stations, daily changes in sea level along the coastline, annual prevalence of specific diseases, sunspot cycles, unemployment rates, inflation, commodity futures—the list is virtually endless. In wildlife biology, we also conduct a lot of monitoring, most commonly measuring state variables that either directly or indirectly relate to the size and areal extent of wildlife populations. Most of these activities, both in wildlife biology and more generally, are not strongly targeted at answering specific questions. Rather, they reflect measurements of things we consider to be of general interest, but which, as data streams temporally extend, often produce a variety of novel insights. For instance, no one could have predicted the benefits of long-term, fixed local weather stations or atmospheric CO_2 measurements in evaluating climate change, a purpose for which neither data stream was initially intended (Lovett et al. 2007; Harris 2010).

The general value of monitoring is not debated: without a time series of previously collected data we are limited in our ability to evaluate current conditions or make projections into the future. However, the collection and maintenance of data across extensive time and space that characterize an effective monitoring program are both expensive and difficult. Maintaining both resources and infrastructure to regularly collect and archive information with established methods and quality controls is challenging. Thus, it is reasonable to ask not whether these data are useful (they invariably are) but rather how useful they are when compared with the associated opportunity costs and whether monitoring data could be collected more efficiently.

Choosing exactly what to monitor in wildlife biology is no trivial matter. For example, should we monitor populations, habitats, or both? If populations are monitored, what is the appropriate state variable(s) to measure: abundance, density, presence, age-class structure, genetic variation or structure, survival, reproduction, movement? At what scale and resolution should managers monitor these variables? Can we use surrogate measures for monitoring variables of interest that are either impossible or extremely expensive to obtain on their own?

In this review, we focus on recent thinking concerning the role of monitoring in wildlife science, choice of state variables, and the degree to which monitoring can and should be designed and targeted to answer specific questions of management significance. Lastly, we discuss the paradigm of linking monitoring to risk assessment, decision analysis, and adaptive management.

Why Monitor?

We start this review by asking a simple question: why monitor at all? Although this may seem like a simple question to address, it is not. First, monitoring must be differentiated from assessments and research. We define monitoring as the measuring of a parameter of management interest (e.g., abundance of a species), or a surrogate of this parameter, over time (Schwartz et al. 2007). The critical part of this definition is the temporal element. Without considering time, we are not monitoring but rather are conducting an assessment. In essence by our definition, when multiple assessments are combined it becomes a monitoring effort. Assessments can be very valuable in setting management objectives. In fact, an assessment is one of the first steps for initiating a monitoring program. Assessments can be particularly useful if we ask if a state variable of interest is meeting a certain threshold. For example, instead of asking if estimates of elk (*Cervus canadensis*) abundance (abundance being the state variable of interest) are changing over time, we can ask: do we have a minimum of number of elk in a particular area after a given management action? The latter question is much simpler and less expensive to accomplish. We may even tie these assessments to triggers to make the situation more powerful. For instance, we could state (a priori to the assessment) that if we do not have a minimum number of elk in a particular area, post a management action, then we will take particular and precise follow-up actions. These assessments often are not representative measures of the state variable, nor do they need to be. For example, surveys can concentrate on those areas and populations most likely to change or where detecting change is of the greatest interest. The point here is that although monitoring is very useful, it can be expensive and may not be needed in all circumstances. An assessment can help us decide if monitoring is needed, provide some baseline data for the monitoring effort, and can help us choose which variables may be most precise and powerful for monitoring.

Once we decide that we need to accomplish more than a simple assessment and instead need trend, we are in the realm of monitoring. If we are interested in how the abundance of elk changes over time, we need to consider a suite of technical questions including those relating to statistical power, such as the length of time that we need to monitor a species before we would reasonably expect to see an increase or decline, or how we plan to estimate abundance given budget constraints. Yet, before we consider technical details of a monitoring effort, we first need to clearly define our study objectives and ensure that the product of our monitoring will satisfy our original goals. That is, monitoring abundance of elk over a ten-year time horizon and having 80% certainty that the population has declined by greater than 25% still does not provide any explanation of the mechanism(s) behind the decline. If our initial goal was to identify what causes elk declines in an area, this monitoring effort alone, even if successful in detecting a decline, will have failed as our effort didn't collect data in a framework that could determine why the decline occurred (see adaptive management in the following discussion). Thus, monitoring may be necessary to track trends in wildlife populations of interest, but it may not be sufficient for us to have the information necessary to change a population trajectory (e.g., a decline).

We could argue that all "important" species should be monitored, and if we see declines, we could subsequently address the cause. However, this is much like the monitoring approach that the US Forest Service (USFS), one of the world's largest land owners, adopted from their 1982 planning regulations. Some of the biggest problems associated with the monitoring requirements from the 1982 planning regulations were that monitoring a large number of species was too expensive and thus often not conducted, leading to litigation. Furthermore, these monitoring efforts were seldom followed by any attempt to understand mechanisms behind the changes that were observed. Lastly, the USFS often relied on monitoring vegetation (habitat) as a proxy for presence or abundance of wildlife. Overall, this approach was untenable and was replaced with a new planning rule as of 2012 (see the following discussion), where monitoring is conducted at broad scales, focused on ecological conditions, and used only sparingly on focal species.

Should Habitat Surrogates Be Used?

Monitoring some state variables can be very expensive. One potential alternative to monitoring the state variable of interest directly is monitoring a surrogate. There is much debate in the literature on the appropriateness of using surrogates for monitoring (Carignan and Villard 2002). However, we need to differentiate and clearly define what we mean by surrogates. Surrogates are meant to indicate a variable of interest but are either less expensive or less complicated to measure. In science, surrogates should have a strong statistical link or causal connection to the state variable of interest (e.g., as one variable increases, the other does by a predictable amount). Many surrogates are used for monitoring, with different levels of success, depending on what is being monitored. For example, there is evidence that species accumulation indices, an index relating the percentage of an area surveyed to the percentage of the target species represented scaled by random and maximum possible curves, can act as weak indicators of biodiversity (Rodrigues and Brooks 2007). In wildlife management, we are often interested in surrogate measures for abundance, although there are other state variables that may be more meaningful than abundance.

Unfortunately, too often the surrogate for abundance that is used in practice is vegetation or habitat (Hunter et al. 1988). While the idea behind using habitat or vegetation communities as a coarse filter is very tempting, and has been used extensively in the conservation planning literature, the justification behind it is extremely weak (Oster et al. 2008; Seddon and Leech 2008; Cushman et al. 2010; Lindenmayer and Likens 2011). For example, Cushman et al. (2008, 2010) examined abundance patterns of birds in the Oregon Coast Range and found little support for the idea that vegetation measures could predict abundance patterns at both a plot and subbasin level. They noted that for this concept to work, several criteria must be met. The first is that habitat is a proxy or surrogate for population abundance. This requires that abundance of a species is tightly linked with the current and future environmental conditions such that changes in the environment are predictably reflected in abundance (Cushman et al. 2008). Second, vegetation maps at multiple scales must be reliable as a surrogate for habitat. One problem is that even if there are high correlations between vegetation and habitat, and between habitat and abundance, the compounding of relationships makes the use of vegetation maps a poor proxy of abundance. Furthermore, many wildlife-habitat models that relate species occurrence to broad vegetation types suffer from large rates of commission and omission errors (Block et al. 1994). This is not to say that there are not cases where there are tight statistical links between vegetation and abundance, but this may be the exception, not the rule.

Thus, unless a clear statistical or causal relationship between a surrogate measure such as habitat and species abundance, occurrence, or another state variable is established, one should not assume it exists (Lindenmayer and Likens 2011). Based on theoretical and empirical concerns, we recommend that if one decides to monitor, it needs to be done on focal species, unless adequate data has been collected to determine the causal or statistical relationship between a surrogate and population metric of interest. Thus, we recommend that agencies should manage habitat and monitor species in order to meet wildlife monitoring mandates.

Adaptive Management

Once a decision has been made to monitor a focal species, there is still the issue that monitoring only provides information about trend or trajectory but does not provide any mechanism behind the trend (Nichols and Williams 2006). In recognition of this fact, the study of mechanism behind change in population trajectory has been combined with monitoring to create an "adaptive management" framework (Walters 1986; Johnson et al. 1993; Williams 1996; Block et al. 2001; Yoccoz et al. 2001; Westgate et al. 2013). Adaptive management is the structured, iterative process of making a management decision when there is considerable uncertainty. The adaptive cycle relies on monitoring the system after a management action by collecting additional data and revising models to reduce uncertainty and ultimately inform future decisions. Ideally, studies are structured as experiments to assess cause and effect; in reality, studies are typically pseudoexperiments or observational (Block et al. 2001). In essence, systematic, monitoring data are collected in a hypothesis framework (Nichols and Williams 2006).

Resource managers are often quick to assert that they practice adaptive management. In a general sense, adaptive management is a structured process to learn from what was done in the past. The key here is that the process is structured with a series of steps and feedback loops that facilitate learning. Moir and Block (2001) provided a conceptual model for adaptive management on public lands. It consisted of a seven-step process with the steps linked by a flow of information and feedback loops to inform whether or not management actions were meeting their intent. Information provided in feedback loops is largely the result of monitoring, and without monitoring, adaptive management fails (Biber 2011). These feedback loops can evaluate whether: (1) the monitoring design is effective and the correct state variable(s) are being measured; (2) monitoring is being done as designed; and (3) whether or not the management action is achieving its goal. Clearly, adaptive management hinges on successful monitoring. The failure to conduct valid monitoring is considered the "Achilles' heel" to adaptive management (Moir and Block 2001).

While the adaptive management approach can work well for species in which data are relatively easy to collect and abundant, such as game species, many monitoring efforts are focused on rare, endangered, and sensitive species. Threatened, endangered, and sensitive species often occur in low numbers and can be secretive and difficult to sample (Neel et al. 2013). Data will inherently be limited compared to more abundant game species. Thus, the adaptive management framework is likely to fail for rare species where experiments are unethical and data is sparse, unless adequate sampling effort, sometimes approaching a census, is allotted.

Articulating the Monitoring Goal

One failure of many monitoring programs is the lack of articulation of clear and measurable objectives. Clearly, one of the advantages of the adaptive management framework is that objectives must be stated if data are to inform competing management hypotheses of how the system functions (Nichols and Williams 2006). However, adaptive management also has been criticized for not articulating and including triggers to change management actions, leading to a lack of accountability (Nie and Schultz 2012). Outside of the adaptive management framework, there are hundreds of ongoing monitoring efforts that occur on public lands that have no clearly stated objectives or thresholds and are not tied back into decision-making processes.

The first monitoring goal that needs to be clearly stated is the purpose of the monitoring effort. Are the data being collected for general context monitoring (*sensu* Holthausen et al. 2005), which is defined as monitoring a broad array of ecosystem components at multiple scales? Breeding Bird Survey data (e.g., Boulinier et al. 1998) are an example of these types of data. Inherent in context monitoring is the idea that monitoring will detect emerging threats to wildlife populations and unanticipated changes in species or multispecies metrics (e.g., abundance, distribution, species richness, changes in local extinction and local colonization). At the other end of the spectrum from context monitoring is targeted monitoring. This is the monitoring of how a particular species is responding to a management action or general threat (Holthausen et al. 2005). Targeted monitoring also can be the monitoring of sensitive, threatened, or endangered species that may not be suffering from ongoing management actions but are threatened by small population processes (Caughley 1994). Context and targeted monitoring have very different objectives and costs, and likely involve collection of different types of data. Therefore, the first step in designing any monitoring program will be stating this broad objective.

For the remainder of this review, we will largely focus on targeted monitoring. We do not mean to imply that large, broad-scale context monitoring efforts have been unsuccessful or are not useful. Clearly, efforts like the Breeding Bird Survey, which relies on volunteers across the United States to observe birds during the breeding season, has led to many useful insights into changes in bird distributions and abundances (Link and Sauer 1998; Sauer et al. 2003; Sauer and Link 2011; Stegen et al. 2013). Whereas we see the value in context monitoring and advocate the need for national-level monitoring programs to accomplish this, success requires well-coordinated, well-funded, broad-scale data collection on many species or assemblages. It therefore represents the minority of monitoring efforts designed each year worldwide (e.g., Greenwood 2003; Kéry and Schmid 2006; Kéry and Plattner 2007; Biodiversity Monitoring Switzerland [http://www.biodiversitymonitoring.ch]). Thus, in the subsequent sections, we discuss issues of targeted monitoring related to metrics to monitor, sampling area, scale, cost, power, effect size, and duration.

Metrics to Measure and Monitor

An investigator must first decide *what* it is that he or she wants to measure (see table 10.1 adapted from Holthausen et al. 2005). As noted earlier, the ultimate piece of information sought is how management affects the population(s) under consideration. The most common metric to monitor is abundance. Yet, abundance may not be the most biologically meaningful metric

Table 10.1 Population and community measures that may be employed in a monitoring program. This table is adapted from Holthausen et al.'s (2005) table 4.

Population Measures	
Abundance/density	Abundance and density are the most commonly used monitoring metrics. Abundance estimates require capturing, marking, and recapturing individuals multiple times. Note that individuals do not need to be physically captured (e.g., noninvasive DNA capture-recapture) to provide an estimate with reasonable variance. Because these measures are often expensive, indices of abundance are frequently used. Density is abundance per area, although the estimation of the area being considered is not always clear. Abundance change over a unit of time (λ, finite rate of increase) also can be monitored.
Occupancy	Occupancy assesses presence of an organism at sample locations. Showing presence at a site necessitates only proof of the species' existence. Demonstrating absence is more complex and requires information on the probability of detecting the organism, given it is present. These data are often placed in an occupancy framework where multiple site visits at multiple stations can be used to infer changes in occupancy, which can be linked to abundance (Royle and Nichols 2003) or monitored as its own metric (MacKenzie et al. 2006). Occupancy models (MacKenzie et al. 2006) also can be used to detect changes in local colonization and local extinction of a species.
Vital rates (birth/death; immigration/emigration)	Vital rates are age-specific birth and death rates, and emigration and immigration rates. Monitoring vital rates sometimes provides a better measure of trend than measures of abundance. Collecting vital rate data allows the use of population viability analyses, which can estimate probability of persistence and allow modeling of the impact of management actions on persistence.
Range distribution measures	Geographic range is the range of a given species at a particular point in time. Monitoring range over time provides information on if a species is expanding, contracting, or remaining constant. Groups like the IUCN often use range distribution to monitor change in species status (Mace et al. 2008).
Genetic measures	Genetic monitoring has been defined as quantifying temporal changes in population genetic metrics or other population data generated using molecular markers. In general, three categories of genetic monitoring are recognized. Category I is simply the use of diagnostic molecular markers for traditional monitoring such as estimating changes in population abundance, vital rates, hybridization, or geographic range. Category II used the population genetic data itself to monitor changes in genetic variation, effective population size, dispersal, or population substructure. Category III addresses questions of evolutionary potential, such as change in frequencies of adaptive genes (see Hansen et al. 2012).
Community Measures	
Diversity measures	Species diversity measures used in community and ecosystem ecology include species richness and evenness. Species richness can be based on repeated estimates of the presence or absence of taxa, whereas species evenness requires abundance data. With recent advances in analytical techniques (Dorazio et al. 2006; Royle and Kéry 2007; Kéry and Royle 2008), we can estimate species richness and community dynamics from local colonization and local extinction (e.g., Sauer et al. 2013) while accounting for detection.
Integrity measures	Biological integrity is a comparative approach that examines if conditions (which can be measured using other approaches mentioned previously) are similar today to historical times or to other locations that are considered "intact."

to choose (Van Horne 1983) because it only provides one piece of the puzzle, and other state variables may be the driving factors for population change. For example, imagine a stream that has two adult fish in the early spring that produces between five hundred to one thousand fry in the summer, which are subject to heavy mortality. Imagine this dynamic continuing for a decade. An estimate of abundance in the fall would be less useful than a measure of connectivity or effective population size. Even if abundance is the desired goal, as it is easily articulated and fits well into management models (Fancy et al. 2009), estimates of abundance may not be obtainable, especially across an entire geographic range. In lieu of collecting data to estimate population abundance, investigators consider various proxy measures (table 10.1) and often consider these at small spatial scales, sometimes distributed across a species geographic range.

It is important here to note that while we did not advocate the use of habitat as a surrogate measure, there are often many surrogates that can be used for formal estimates of abundance (Slade and Blair 2000; McKelvey and Pearson 2001; Tallmon et al. 2010; Tallmon et al. 2012; Noon et al. 2012). The key difference is that modeling efforts that have tested the use of these surrogates for abundance often find them to be good predictors of abundance and, in some cases, more robust for detecting trends than estimates of abundance. In other words, the causal connection has been studied and validated. For example, Tallmon et al. (2010) showed that declines in effective population size can be a more sensitive measure of declining populations than monitoring estimates of abundance under many conditions. McKelvey and Pearson (2001) demonstrated that M_{t+1} (otherwise known as the number of unique individuals captured at time t+1) showed lower variance and less sensitivity to model violations than formal abundance estimates when sample sizes were small (but characteristic of most field studies). Similarly, there has been a recent explosion of the use of occupancy estimates as a way to monitor trends in wildlife populations (Gaston et al. 2000; MacKenzie and Nichols 2004; MacKenzie et al. 2006; Estrada and Arroyo 2012).

When monitoring populations, it is beneficial to determine which parameters are most sensitive to change and focus on those. Typically, focus is on population abundance or density of breeding individuals, which could be misleading, if the population includes a large number of non-territorial animals (e.g., nonbreeding individuals) not easily sampled using traditional methods. In a situation where there is high mortality of territorial animals that are immediately replaced by surplus, the population of territorial animals may appear stable while the overall population is declining.

We recommend conducting sensitivity analyses to determine which state variables contribute most to population change (e.g., Caswell 1978; Williams et al. 2002). If demographic data are not available, the investigator will need to conduct literature reviews, consult with established experts (useful for Bayesian priors [Gelman et al. 2004]), or use results of similar studies to establish the basis for using specific state variables. All of these sources can be used in a sensitivity analysis, although some sources will be more valuable than others. For example, there will be more variation in parameter estimates from a literature review of studies in different geographic locations or resulting from different study methods compared to parameter estimates from a study conducted in the same area (Mills 2012).

Overall, we recommend that selecting the variables to measure should be based on strong statistically based information. One must understand the biology and population dynamics of the species being monitored to make better decisions on exactly what to monitor.

Selection of Sampling Areas

Once the study objective is established, the scale of resolution chosen by ecologists is perhaps the most important decision in monitoring because it predetermines procedures, observations, costs, and results (Green 1979; Hurlbert 1984). A major step in designing a monitoring program is to define the target population. Defining the target population essentially defines the area to be sampled. This first step establishes the sampling universe from which samples can be drawn and the extent to which inferences can be extrapolated.

Although this seems rather straightforward, the mobility of wildlife can muddle the inferences drawn from the established area. The basis for establishing the sampling frame includes information on habitat-use patterns, range sizes, movement patterns, and logistics of sampling a large area. Ideally, the bounds of the sampling area would correspond to some natural barrier that would keep a population closed, but this is often not the case. Without such bounds, open forms of estimators will need to be used (Marucco et al. 2011; Wilson et al. 2011; Ivan et al. 2013), or a combination of open and closed estimators, where sampling occurs during a short interval when the population is closed and over multiple seasons or years when the population is open (i.e., robust design [Pollock 1982; Kendall and Nichols 1995; Kendall et al. 1995]).

To reduce this issue associated with population closure, there have been developments in using an approach to estimating density (abundance per area) instead of abundance, using spatially explicit capture-recapture models (e.g., Efford 2004; Royle and Young 2008; Gardner et al. 2009; Russell et al. 2012). The distribution of individuals and their movements relative to a trapping array are explicitly modeled using this framework. For instance, the use of telemetry data can be used to evaluate geographic closure (White and Shenk 2001; Ivan et al. 2012) and to explicitly model animal movement in spatially explicit capture-recapture models (Sollmann et al. 2013). To evaluate geographic closure, animals are marked and time spent in the study area is estimated and used to scale abundance estimates and provide density (Ivan et al. 2013). Similarly, genetic data can be used to evaluate violations of closure assumptions by using approaches such as spatial autocorrelation among genotypes (Peakall and Smouse 2006). These methods provide indices of average animal gene flow in an area. Unfortunately, there has been little attention paid to the integration of genetic approaches to gene flow and abundance estimation, although this is beginning to change (Luikart et al. 2010).

When to Collect Data and for How Long?

A key aspect of monitoring is to know when to collect data, including the timing of data collection and the length of time over which data should be collected. Both are influenced by the biology of the organism, the objectives of the study, intrinsic and extrinsic factors that influence the parameter(s) to be estimated, and resources available to conduct the study.

Obviously, studies of breeding animals should be conducted during the breeding season, studies of nonbreeding animals should be conducted during the nonbreeding period, and so on. Within a season, timing can be critically important because detectability of individuals can change for different activities or during different phenological phases (Bailey et al. 2004; MacKenzie 2005). Male passerine birds, for example, are generally more conspicuous during the early part of the breeding season when they are displaying as part of courtship and territorial defense activities. Detection probabilities for many species will be greater during this period than at other times. Another consideration is that the very population under study can change within a season. For example, age-class structures and numbers of individuals change during the course of the breeding season as juveniles fledge from nests and become a more entrenched part of the population.

Population estimates for a species therefore may differ depending on the timing of data collection. Once the timing decision is made, data must be collected during the same time in subsequent years to control for some of the within-season variation. Objectives of a monitoring effort also dictate when data should be collected. If monitoring is conducted to evaluate population trends of a species based on a demographic model, sampling should be done at the appropriate time to ensure precise and unbiased estimates of the relevant population parameters. Demographic models typically require fecundity and survival data to estimate the finite rate of population increase. Sampling for each of these parameters may be necessary during distinct times to ensure accurate estimates for the respective measures. Timing also is essential in the field of genetic monitoring (Goudet et al. 2002; Schwartz et al. 2007). For example, different measures of gene flow and genetic variation have been found to be more sensitive to the timing (e.g., pre- or postdispersal) of sample collection (Goudet et al. 2002; Hammond et al. 2006). This makes intuitive sense as populations should be most differentiated if sampling occurs postbreeding, but predispersal, and most similar after a pulse of dispersal. Consistency in timing and methods is essential.

Length of the study refers to how long a study should be conducted for reliable and accurate parameter estimates. A number of factors including study objectives, field methodology, ecosystem processes, biology of the species, budget, feasibility, and statistical power issues influence how long a monitoring study should occur. Monitoring studies also must consider temporal qualities of the ecological process or state being measured (e.g., population cycles, successional patterns including their frequency, magnitude, and regularity; Franklin 1989). Furthermore, a study should engage in data collection over a sufficiently long period to allow the population(s) under study to be subjected to a reasonable range of extrinsic environmental conditions. Consider two hypothetical wildlife populations that exhibit cyclic behaviors: one that cycles on average ten times per fifty years and the other exhibiting a complete cycle just once every twenty-five years. The population cycles are the results of various intrinsic and extrinsic factors that influence population growth and decline. A monitoring program established to sample both populations over a twenty-five-year period may be adequate to understand population trends in the species with frequent cycles but may be misleading for the species with the long population cycle. A longer timeframe would be needed to monitor the population of species with lower frequency cycles.

Considering only the frequency of population cycles may be inadequate, as the amplitude or magnitude of population shifts also may influence the length of a study. Sampling the population of a species exhibiting greater amplitude in population increases and declines would require a longer period to detect a population trend or effect size than a species exhibiting lower amplitude in population shifts.

Power to Detect Change

A major question that arises in any study is: "Am I sampling enough to detect a trend, should one exist?" "Enough" can be enough area, enough individuals per area, enough time, and so forth. The best way to address this issue is through simulations or calculations of statistical power (Zielinski and Stauffer 1996). That is, it is important to ask a question like: "If I am interested in changes in abundance and I mark one hundred individuals per year for ten years, will I have a 95% chance of detecting a 20% decline?" Or ask: "How many samples do I need to have 80% certainty of detecting a 20% decline given a ten-year monitoring program?" Too many monitoring efforts fail to quantify what effect size they can detect with a specific effort—only to find at the end of the monitoring period that they never had power to detect change.

There are several computer programs available to examine interactions between sample size, effect size, type I error, and statistical power to detect trends in abundance, count, or occupancy under different sampling schemes (Gerrodette 1993; Gibbs 1995; Bailey et al. 2007). For example, program MONITOR estimates statistical power of a population-monitoring program when supplied with the number of plots, counts per plot and their variation, duration of the monitoring program, interval of monitoring, magnitude and direction of ongoing population trends, and significance level associated with trend detection. Similarly, program TRENDS conducts a power analysis of linear regression for monitoring wildlife populations (Gerrodette 1993). TRENDS estimates statistical power of a monitoring program after providing data related to the rate of change, duration of study, precision of the estimates, and the type I error rate. Alternatively, power can be set to a desired level and program TRENDS estimates one of the other input parameters. For example, if 80% power is desired, one can ask how many years a monitoring effort must be conducted given a precision of the estimate and type I error rate.

Recently, there has been an increase in reliance on occupancy-based approaches to estimate population trends. A new spatially based simulation program called SPACE has been created to evaluate power for detecting population trends over time based on species detections (Ellis et al. 2013). SPACE has been used to evaluate the power of monitoring trend occupancy for territorial animals while accounting for natural history (e.g., territorial overlap, different space use of different life stages), habitat use, and sampling scheme (e.g., size of the grid; Ellis et al. 2013).

We suggest that an analysis of power to detect trend is initiated before any monitoring effort is conducted. If specific programs do not exist to evaluate statistical power for a given method, consulting papers that have conducted simulations on power (and type I error rates) with that method will be insightful. Overall, it would be very unrewarding, if not embarrassing, to come to the end of a long-term monitoring program where no trend was found, only to realize that power to detect trend with the given protocol was no greater than a coin flip.

Resources to Detect Change

All of the components of monitoring that we have discussed so far are important but rely on sufficient monetary resources to complete these efforts. Agencies tasked with developing monitoring programs often work with limited budgets; therefore, cost-efficient sampling designs are paramount. Thus, we advocate using cost as an additional constraint for study design optimization (e.g., Sanderlin et al. 2009; Sanderin et al. 2014). We can determine the power to detect an effect size and the amount of sampling needed for accurate parameter estimates, but if there are not enough resources available to complete a study with these design specifications, there is no point in continuing with the monitoring effort.

Case Study: Lessons from USFS Monitoring under the Planning Rule

We can learn a lot about monitoring programs from the successes and failures of the USFS's efforts to monitor species under the 1982 "planning rule." The US Forest Service is a United States land management agency responsible for facilitating multiple-use activities on seventy-eight million hectares of land. These activities include the maintenance of healthy wildlife populations, clean water, recreational opportunities, and timber production. Guidance on how to balance competing demands on the landscape is given in the National Forest Management Act (NFMA) of 1976. One of the most controversial aspects of the NFMA has been the tiered planning approach and how planning rules accomplish NFMA's requirement to "provide for a diversity of plant and animal communities based on the suitability and capability of the specific land area in order to meet overall multiple-use objectives." Direction on how to achieve NFMA's mandate to provide for biodiversity was given in promulgated planning rules, which until 2012 called for the maintenance of viable populations of existing native and desired nonnative vertebrate species in a planning area. Part of that maintenance of viable populations included identifying management indicator species and noting that "population trends of the management indicator species will be monitored" (section 219.19 of the 1982 planning regulations). Interestingly, these management indicator species, which were proxies for other species, were often monitored through the use of habitat availability. This was considered a "proxy-by-proxy" approach and was deemed legally sound (*Inland Empire Public Lands Council v. USFS* 1995; *Lands Council v. McNair* 2008), although biologically questionable.

Monitoring under the 1982 planning regulations was seen as essential to the success of the multiple-use mandate and maintenance of viability clause. Unfortunately, the regulations relied on focused monitoring of a multitude of management indicator species, which proved too expensive and difficult to accomplish (Schultz 2010; Schultz et al. 2013). Thus, monitoring under the 1982 planning regulations often devolved to assessing the status and trend of the amount of habitat the species required without strong statistical links between the habitat and the species. We are currently in a new era where as of April 2012 the USFS is acting under a new planning rule (which has been in preparation for roughly fifteen years) with extensive public and scientific involvement. This new direction provides a great opportunity to improve management of wildlife on public lands (Schultz et al. 2013). Our goals in the remainder of this section are to review the new rule as it relates to monitoring and then provide advice as to how to choose species to monitor and points to consider.

Monitoring under the 2012 planning rule requires monitoring indicators that relate to the status of ecological conditions and focal species, which are defined as a narrow group of species whose status allows inference to the integrity of the larger ecosystem. These focal species are to "provide meaningful information regarding the effectiveness of the plan in maintaining or restoring the ecological conditions to maintain the diversity of plant and animals communities" (USFS 2012). Interestingly, the new monitoring rule has no special monitoring requirement for "species of conservation concern" (e.g., Threatened and Endangered Species), although it does have a viability requirement for this category. What remains to be tested in this new planning rule is how individual forests choose focal species to monitor and then implement a monitoring program.

In the following list, we discuss the characteristics of a focal species that we think will be effective for making inference to the larger ecosystem. An ideal focal species would have many of the following characteristics:

1. The species must index something greater than itself: A species that only reflects its own success in an ecosystem will likely be too narrow for capturing ecological integrity. While we would like to have data on all species in a planning area, common sense and the rule itself dictate that focal species are only limited subsets of species in the ecosystem. Species such as ecosystem engineers (e.g., beavers [*Castor canadensis*]) or keystone species (e.g., pileated woodpeckers [*Dryocopus pileatus*]) are good examples of animal species that index something greater than themselves. Ovenbirds (*Seiurus aurocapillus*) are another example, as they appear to be sensitive to fragmentation of mature forests and may index other species and processes of these habitats.
2. Low range of natural variability: If a species has large natural fluctuations that are not linked to climate or other environmental processes, statistical trend will be difficult to obtain. In this case, the temporal variation swamps out process variation, hence obviating the ability to identify trend. Similarly, a species known to cycle, like snowshoe hare (*Lepus americanus*), would make a poor choice as a focal species, as statistically detecting trend will be complicated. However, there are times when a species with a large range of natural variability will be exquisitely sensitive to environmental change, and this relationship (the signal) will override concerns about the large range of natural variability (the noise).
3. Species need to be statistically linked to habitat at the appropriate scale: If the purpose of the focal species is to provide inference to or monitor the integrity of the ecosystem, then there needs to be a strong statistical link to the vegetation. Likewise, selected focal species must be sensitive to particular land management actions. As many ecological processes are scale dependent, this statistical evaluation should be conducted at a multiple array of scales from plot to landscape levels.
4. Species can be monitored: A species may have a low range of variability and be a great reflection of land-use changes, but if it is too difficult or expensive to monitor then it should not be chosen. Some species' life history traits make them easier to monitor than others (e.g., small mammals that have smaller home ranges, are less vagile, and are abundant may be preferable to a carnivore that is secretive, rare, and vagile).
5. Background knowledge about national/global trend is needed for context: It is important to tie the local dynamics of a species into a broader monitoring program. Imagine picking boreal toads (*Bufo boreas*) or other amphibians for monitoring thirty years ago. The population's numbers would have likely declined in a forest, and this may have been attributed to habitat modifications or other management actions. This would be the wrong conclusion in the face of global amphibian decline. Knowing the magnitude of global decline compared to local declines would have provided relevant context.

Conclusions, Creative Approaches to Monitoring, and a Need for Large Scale National Efforts

Wildlife monitoring means many things to many different people. For some it is the broad-scale, long-term efforts to provide background information on species, while for others monitoring is very focused on a specific species for a specific reason (e.g., to detect a change in abundance after a wildfire or other natural event), and still for others monitoring is meant to provide data to better understand mechanisms underlying ecosystem change (i.e., adaptive management). Regardless of the reason to enter a monitoring effort, effective monitoring is a challenging endeavor as it can be prohibitively expensive, take a long time frame to achieve results, and may require coordination among multiple groups, especially if the effort is broad in scope. This will clearly require creative approaches to monitoring, which include the development of new field approaches and statistical techniques that can make monitoring more feasible. We have already seen many of these developments make monitoring efforts that were once inconceivable, such as the monitoring of abundance of wolves (*Canis lupus*) across the Alps (Marucco et al. 2009), now possible with the advent of DNA technology. Similarly, other technologies, such as trail cameras, have greatly improved our ability to obtain reliable data in an inexpensive manner. Developments in occupancy modeling and abundance estimation also are improving our ability to monitor. And combining the two approaches—where we use occupancy estimation with environmental DNA sampling (eDNA), which detects presence/absence of species by sampling the environment (e.g., water or air samples)—will certainly reduce costs and allow for more monitoring data to be collected.

Another recent trend to improve monitoring is to involve local citizens in monitoring efforts as part of "citizen-science" projects (e.g., Schmeller et al. 2008; Silvertown 2009; Miller-Rushing et al. 2012). This can reduce costs, allow for more monitoring data to be collected, and foster communications between the public and governmental agencies. These are unique opportunities to educate the public on research that is conducted on public lands and give a local community a sense of ownership with these public lands.

We wish to revisit several critical points that we covered earlier in this chapter, which are essential when designing more effective and efficient monitoring programs. First is the concept of the use of habitat as a proxy or surrogate measure for wildlife populations. Although it is very tempting to use habitat as a proxy for abundance of a species, there is very little evidence that this is a defensible position. The burden of proof is therefore on the group wishing to use habitat as a proxy to statistically show the tight relationship between the two variables before using them in a monitoring program. This does not mean that we are against surrogates or indices in general, but we urge caution in their use without first establishing a causal or statistical relationship, which is often not established when monitoring habitat as a surrogate for species' abundance. In fact, there is strong statistical evidence that under some circumstances indices of abundance are as good as collecting abundance, and some of these indices and surrogates may reflect change better than estimated abundance (McKelvey and Pearson 2001; Engemann 2003).

Our next summary point is that once a decision is made to monitor, we implore groups to be very specific in their objectives. This starts with the statement of intent of whether this is a broad-scale, context monitoring effort or a focal/targeted monitoring effort. The goal should be stated in context of an overarching objective. For instance, it is not enough to simply claim that we want the USFS to do focal monitoring of a species, but it is essential to understand that the intent of this focal monitoring may be to measure a broader ecosystem process. After these goals are established, we urge those responsible for monitoring efforts to consider all methodological approaches and conduct rigorous power analyses that include cost constraints to ensure the data collected will ultimately be able to answer the question posed. We also urge consistency in methods and timing of data collection. The world is filled with too many partial datasets, where someone changed methods without proper calibration linking the approaches.

Penultimately, we encourage the use of the adaptive management framework when appropriate. While this approach may not always work, we think that, when possible, setting monitoring in a framework that helps us also understand mechanisms behind trends is essential. We hardly have enough fiscal resources available now to conduct the initial monitoring, let alone funds to go back and try to conduct studies to understand trends once they are detected. Adaptive management and structured decision making allow us to kill the proverbial "two birds with one stone." However, we caution that adaptive management is best accomplished when thresholds or triggers are in place to force certain conservation actions, should these triggers be tripped.

Lastly, in this chapter we focused on the monitoring of focal species, which included species that reflect broader ecosystem processes and species that are of special management or conservation concern. While we believe that most monitoring efforts will be targeted, this does not mean that we do not see the value in context monitoring approaches. We can only imagine the valuable data that could be produced if we had a national wildlife monitoring program that was on the scale of our Forest Inventory and Analysis program (cf. Manley et al. 2004, 2005). Perhaps such an effort would have allowed the detection of emerging threats, such as invasive species, faster than our *ad hoc* detection methods currently in place could, ultimately saving billions of dollars. A commitment to large-scale monitoring also might enable us to detect patterns of faunal change resulting from overarching effects of climate change. Furthermore, the background information such a program would provide would help us interpret local trends in local monitoring datasets.

Overall, monitoring when done well provides important data on changes occurring on the landscape, yet when done poorly or in an *ad hoc* manner is simply a waste of valuable resources. We hope our framework here helps those intent on designing new monitoring efforts in the future.

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