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A DECISION TREE FOR MONITORING WILDLIFE TO ASSESS THE EFFECTIVENESS OF CONSERVATION INTERVENTIONS

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By Samantha Strindberg & Tim O'Brien

Wildlife Conservation Society
Global Conservation Program
2300 Southern Boulevard
Bronx, NY 10460

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A DECISION TREE FOR MONITORING WILDLIFE TO ASSESS THE EFFECTIVENESS OF CONSERVATION INTERVENTIONS

Basic Monitoring Considerations

Worldwide, biodiversity is being lost at a rate comparable in magnitude only to a handful of cataclysmic mass extinction events in the Earth's geological history. Loss of biodiversity has major implications for ecosystem health and function, provision of goods and services, and the impoverishment of quality of life. Biodiversity loss can be thought of as the sum of decline and loss of many individual species. Stemming biodiversity loss, therefore, requires that we reduce the decline and loss of individual species and communities through effective interventions and management. This can be accomplished through better conservation management of species, habitats and ecosystems.

Monitoring is a crucial component of good conservation management (Salafsky *et al.*, 2001; Open Standards for the Practice of Conservation, 2007). It allows us to assess whether or not threats are decreasing, and/or wildlife populations are increasing or remaining stable. It requires that we identify the most important threats, where they occur within the landscape or seascape of interest, and how they change over time. Through monitoring we can test our assumptions as to whether our interventions actually lead to what we want to achieve, or whether they are wasted efforts.

Monitoring tracks progress over time towards a clearly defined objective. We can only monitor if we have a clear idea of what we hope to achieve, thus setting explicit objectives lies at the core of effective monitoring. Monitoring assumes sufficient knowledge of the system of interest to allow us to set explicit objectives in contrast to research that gathers information about the unknown.

Ideally we would want to monitor the conservation strategies (also referred to as interventions or activities), the threats and the conservation targets themselves to get the most information about the effectiveness of our actions. We would monitor our strategies to make sure that they are being implemented as we planned (e.g., Are trained guards getting out on patrol?). Since our strategies are chosen to reduce levels of threat to wildlife and their habitat, we monitor our success in reducing threats to assess whether or not our interventions were worthwhile (e.g., Is there a reduction in the number of arms and cartridge shells in the area being patrolled?). Lastly, we look at the status of the wildlife species or habitat that form our conservation targets to see whether it improves when our interventions are implemented successfully, and threats are reduced (e.g., Are elephant populations doing better due to the reduction of poaching with firearms?).

The improved state of our conservation targets is the ultimate indicator of success. Knowing that state gives us the greatest level of confidence in our interventions. Yet, this level of monitoring is often the most difficult to implement, costs the most, and may have longer lag-times (see Figure 1). If we monitor the strategies and threat reductions as proxies for our progress there are definite tradeoffs. The time frame for seeing results and the costs of monitoring generally declines as we move from directly monitoring changes in wildlife and their habitats, to monitoring reductions in threats, to monitoring whether or not our strategies were implemented as planned. However, using these proxies that change within a shorter time frame also lowers our level of confidence in our actual conservation success (Wilkie *et al.*, 2002 and 2006).

As we will see in a subsequent section, even if we decide to monitor the conservation target directly, different types of indicators can be chosen for this and can give different results with varying associated levels of confidence in those results. For the remainder of this document we focus on monitoring our conservation targets, specifically wildlife, rather than monitoring the threats or interventions, although the techniques that will be covered can readily be applied to some forms of threat monitoring.

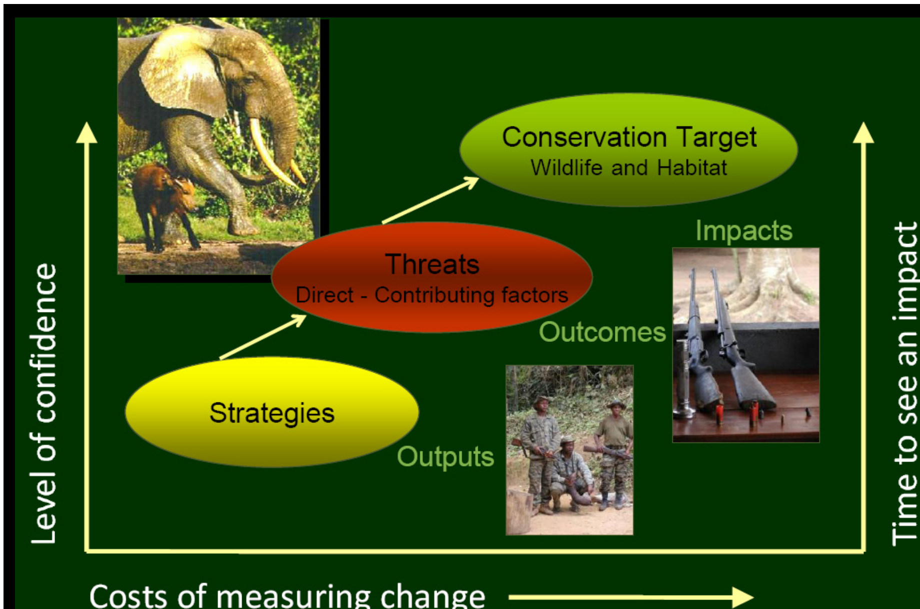


Figure 1: The relationship between confidence in monitoring results, monitoring cost and time to see impact of management activities for the different components that could be monitored over time. Monitoring strategies, threats or conservation targets are frequently referred to as measuring our outputs, outcomes and impacts, respectively.

Monitoring tracks changes over time and/or space and this distinguishes it from a sample survey, which estimates conditions at a single point in time or space. Thus monitoring uses survey results at many instances in time/space. The next section considers a general sampling framework upon which the monitoring results are built.

The General Sampling Framework

Usually, the geographic areas of interest (study sites, landscapes) for monitoring wildlife are large and difficult to access. Thus when designing a survey we will seldom be able to cover the entire area of interest, but instead select a manageable sub-region. Within that sub-region referred to as the survey area, we usually attempt to cover the entire area or we select sampling units.

If $E(C)$ is the expected value of the count statistic C (number of animals counted or number of occupied sampling units observed) and p is the detection probability, then the relationship between the count statistic and the true population size or occupancy N is given by:

$$E(C) = pN \tag{1}$$

When detection is 100% (p =1), the count statistic provides an accurate estimate of N. However, when $p < 1$ the count statistic provides a biased estimate of N. For example, if 10 animals were observed and in fact p = 1/2 then half of the 20 animals in the survey area were missed. Once the detection probability has been estimated, then the estimate of abundance or occupancy can be obtained from count statistics as follows:

$$\hat{N} = \frac{C}{\hat{p}} \tag{2}$$

Note that the hats (^) indicate estimated parameters. The equation is generalized as follows to incorporate the proportion of the survey area covered α:

$$\hat{N} = \frac{C}{\hat{p}\alpha} \tag{3}$$

This formulation is known as the canonical estimator. The various methods used to estimate abundance, density, occupancy, and species richness can be expressed in terms of the canonical estimator (Williams *et al.*, 2002).

Designing Monitoring Programs

Yoccoz *et al.* (2001) emphasize the need to pay attention to three basic questions when developing monitoring programs: (1) <u>Why</u> monitor? (2) <u>What</u> should be monitored? and (3) <u>How</u> should monitoring be carried out? With respect to 'why monitor,' programs to monitor species arise for a number of reasons and at a number of spatial scales. Species conservation can occur at the site (population), landscape/seascape (metapopulation) or global range. Once the key threats to the species have been identified and the conservation activities planned, then the monitoring program to assess the effectiveness of the interventions can be put in place. The important part of planning a species monitoring program is to have a clear idea *a priori* of the objectives of the monitoring program. Before formulating a monitoring program, or in parallel to improve its formulation over time, research may be needed to gain better scientific understanding of the ecological and human-influence factors that affect state variables (density, occupancy), vital rates, or some combination. When direct interventions are applied to address threats to a species' persistence, it is possible to gain insights from monitoring data when *a priori* hypotheses are used to make comparisons among alternatives. Combining monitoring with management interventions may yield information about the current population status and the impact of management activities.

'What to monitor' follows from the monitoring program objectives. Objectives should focus on state variables and rate parameters that characterize the system dynamics (Williams *et al.,* 2002). In species monitoring, the state variable may include abundance, density or occupancy. In biodiversity monitoring, the state variable can be a measure of species richness, or some combination of 'abundance and diversity' (Magurran, 2004). The rate parameters may be birth, death, immigration, emigration, extinction and colonization. Abundance can be measured directly (an estimate of numbers of animals or the biomass of the species), or indirectly (a measure of occupancy for a species). In addition, it can be measured by means of information collected on the animals themselves or on their sign. For communities, often it is desirable to include some measure of abundance/biomass/occupancy in the monitoring metric. This increases the complexity of the monitoring program but provides better information on the tradeoffs between species richness, species abundance and species evenness, and a better understanding of system function. In general, a monitoring program's design and field implementation details will depend on the choice of conservation target and the selected monitoring metric.

'How to monitor' should follow best practices for sampling. There is a large literature on species and community-level monitoring. Much of this literature is devoted to the 'How' question and the merits of indices requiring calibration versus estimators of absolute abundance. The ideal monitoring program would account for variation in detectability across individuals, over time, and across space (Pollock *et al.,* 2002; Moore and Kendall, 2004; Buckland *et al.,* 2005). It would also account for spatial variation and survey error. Accounting for variation in detection is normally done by estimating the detection probability (may also be referred to as a sighting or capture probability) for a population of individuals at a time and at a site, and correcting the count C (number of observed individuals, number of observed occupied sites, number of observed species) by the estimate of detection probability, p, as described above.

The ease with which counts can be obtained and $\hat{p}$ estimated varies widely for state variables of abundance, biomass, occupancy, and species richness. Usually, it will be easier to collect data on occupancy and species richness than on abundance and biomass when working with mammals, birds, herptiles and fish. There is a temptation to use the counts directly as indices of the variable of interest under the assumption that detection probabilities are either equal or are constant over space and time (Conroy, 1996). This is usually not a good idea. For example, when monitoring abundance over time, let $\hat{\lambda}_{ij}$ measure the rate of change in population size between time (or space) i and time j. $\hat{\lambda}_{ij}$ is calculated as the ratio of abundance, N_j / N_i. The counts C_i and C_j, at times i and j, are used as indices of abundance and $\hat{\lambda}_{ij}$ is estimated as:

$$\hat{\lambda}_{ij} = C_j / C_i \qquad (4)$$

The expected value of $\hat{\lambda}_{ij}$ is estimated as:

$$E(\hat{\lambda}_{ij}) = \frac{E(C_j)}{E(C_i)} = \frac{N_j p_j}{N_i p_i} \qquad (5)$$

where the expected value of the counts is equal to the product of abundance and detection probability. If detection probabilities remained constant across space and time then the use of a count statistic is justifiable as a proxy for changes in the parameter being monitored, because the count would be expected to track changes in that parameter. For example if abundance increases, then the count also increases and similarly a decline in abundance is reflected by a decline in the count. Unfortunately, detection probabilities are seldom constant in space and time and thus need to be estimated to enable reliable trend estimation from the raw counts. Without an estimate of the detection probability, it is usually impossible to interpret $\hat{\lambda}_{ij}$ due to the unpredictable and unknown fluctuations in the relationship between *C* and *N*. An index based on counts only may have a smaller variance than the corresponding unbiased abundance estimate incorporating detectability, which is desirable as this makes it easier to detect a trend (see next section on Power Analysis for other factors that impact one's ability to detect a trend). However, the gain in precision is offset by the unpredictable loss of accuracy. It is best to avoid precise metrics with unknown bias. Thus when designing a monitoring program we recommend first selecting unbiased metrics facilitating a reliable interpretation of trends and then focusing on improving precision.

Power Analysis

The ability of a monitoring program to detect a real effect (or a response to the management strategy) when it exists is called the power of the sampling program and analysis. Power increases with increasing sample size, and increasing size of the effect or response. Power decreases as the variance and standard error increases. Power analysis is most useful when planning a study or monitoring program. Power analysis can be used to explore the relationship between the range of possible sample sizes, response sizes that are important, levels of variance that are expected to occur (usually from literature or pilot data), and the desired level of statistical power. The goal is to be able to design a monitoring program (the sampling) that will detect the effect or response with sufficient sensitivity to be used as a basis for management decisions. Low power in a monitoring program means high uncertainty in interpreting the data. Unfortunately, power analyses are rarely conducted prior to setting up a monitoring design.

There are ongoing debates regarding appropriate methods for analysis of trends. In addition, there are discussions about whether or not conservation management questions should be posed in a hypothesis testing framework, which most power analyses assume. Many argue that decision making in the face of uncertainty should at least rely on multiple hypotheses and that the associated models be used to help make these decisions (Kendall, 2001; Williams *et al.*, 2002; Nichols and Williams, 2006; Gerrodette, 2011). These methods work best in data-rich environments, but are increasingly being used in situations of data paucity and limited technical capacity (Yoccoz *et al.*, 2001). Although power analyses placed in a hypothesis testing framework are perhaps not ideal, they do promote more careful thought about the data requirements for a monitoring program and are very informative in terms of illustrating how difficult it may be to show that our conservation actions are effective.

Making decisions for conservation management within a Bayesian framework is a different increasingly popular approach (Wade, 2001; Hoyle and Maunder, 2004; Wade *et al.*, 2007). Bayesian methods are well-suited to problems involving the interpretation of monitoring data. Proponents of the methods argue that they provide a much more intuitive approach to decision making in the face of uncertainty. Bayesian analysis permits the integration of information and data from a variety of sources in a single framework and explicitly considers uncertainty in the decision–making process. Just as decision making can be done in a Bayesian framework, similarly the monitoring techniques themselves can be used or the trend analysis can be done taking either a frequentist or Bayesian analysis approach (Williams *et al.*, 2002; McCarthy, 2007; Royle and Dorazio, 2008; Barker and Link, 2010).

Decision Tree for Wildlife Monitoring

Elephant monitoring, Mpala Ranch, Kenya. © M. Kinnaird

A decision tree for selecting a method for wildlife monitoring can quickly become very complex. At the highest level, we need to determine whether we want to monitor wildlife or habitat. Within each of these, we can monitor at different spatial scales: a site, landscape/seascape level or range level. We can monitor at a number of levels of organization and increasing complexity. At the species level, we can monitor a single population (site), a group of populations (metapopulation) or the entire range (single population to meta-metapopulation depending on the species distribution). At the community level we have a population of species occurring at a site (community) or at a group of sites (metacommunity). In addition, we need to estimate density/abundance, occupancy, demographic rates or a combination of these. As a demonstration of concept, we restrict the decision tree to density/ abundance estimation for a single population of a single wildlife species.

We view the decision tree as composed of several nodes where branching decisions occur. Having decided to restrict the example to estimate density/abundance for a population of a wildlife species we are already four decisions into the decision tree (Decision 1: wildlife; Decision 2: species; Decision 3: population; Decision 4: density/abundance).

Decisions

Decision 1: Wildlife vs. Habitat

This is a basic division in conservation management. In WCS, many programs focus on wildlife recovery and the prevention of future reductions to the population at a site, and the management interventions are focused on that species (e.g. reducing poaching). Alternatively, a wildlife program may focus on species habitat management (e.g. reducing deforestation). The decision here usually is based on the nature of the threat. Although some programs may be single species programs and focus mainly on wildlife monitoring, and some may be community level programs, it is also the case that programs focus on both wildlife and habitats.

Decision 2: Species vs. Community

Species programs are most widespread in WCS (Landscape Species, Global Priority Species, and endangered species). However, increasingly, we are managing and monitoring at the community level (e.g., bushmeat trade, ornamental fish trade, carnivore community conservation).

Decision 3: Population vs. Metapopulation

Most WCS programs are site-based and deal with single populations. Global Priority Species programs deal with species recovery or prevention of future declines over a set of sites, usually semi-isolated and characterized as a metapopulation.

Decision 4: State/rate variable - Abundance vs. Occupancy vs. Demographic rates

The choice of state or rate variable(s) to monitor depends on the nature of the monitoring program, spatial scale, and funding. Demographic monitoring usually occurs at a very small spatial scale (10's of km^2), and at one or a few sites. Abundance/density monitoring usually occurs at a moderate spatial scale (100's of km^2) at one or few sites. Occupancy monitoring often occurs at large scales (1000's of km^2) and at many sites (note that certain abundance/density estimation techniques also provide estimates of some demographic rates). Generally the cost per unit effort is highest for demographic data, followed by abundance/density and then occupancy.

Camera trap photo of chimpanzee with tool.
© P. Boundja

Decision 5: Detection Probability = 1 vs. <1

When we assume that detection probability is one, we are obligated to verify this assumption. It is often approximately true when surveying large animals in open habitats in relatively small areas, and when conducting demographic monitoring at local scales (i.e. cohort-based primate demography). The assumption can be verified through pilot studies that correct for imperfect detection, use of 2-stage sampling, and use of double observer methods. If we assume that detection probability is less than one, we confront three alternatives: (1) detection probability is known and fixed. In this situation we would apply a correction factor to get an unbiased estimate of the state or rate variable; (2) detection probability is unknown but fixed. In this case, we can monitor changes in the state or rate variable using biased indices under the assumption that bias is constant; and (3) detection probability is unknown and not fixed. This is the most likely case and requires correction for detection bias.

Decision 6: Complete coverage vs. sample survey

We can either have complete spatial coverage of the area of interest or we conduct a sample survey over a portion of the area. A population closure assumption is required to bind the study in space and time. For large spatial scale monitoring, we usually conduct sample surveys, but there are examples of complete coverage surveys.

Decision 7: Recognize individuals vs. sub-population vs. no discrimination

The ability to identify individual animals or distinguish between sub-groups in the population will dictate the options for monitoring methods. If it is not possible to identify individuals, distance sampling, occupancy or sign survey methods may be appropriate. If sub-groups within the population are recognizable (e.g. marked vs. unmarked, males vs. females, adults vs .juveniles), then mark-resight or change-in-ratio methods may work. If individuals can be recognized, then Spatially Explicit Capture-Recapture (SECR), capture-recapture or mark-resight methods may be applied.

Decision 8: Animal vs. sign

A basic question for designing monitoring programs is whether the target species is directly or indirectly observable. If the animal is directly observable, we can use observation-based counting techniques. If the animal is indirectly observable (cryptic, nocturnal) we may use passive detectors (camera traps, hair snares) or active trapping. Alternatively, we may decide to use sign (dung, nests, tracks, feeding, hair, acoustic cues) to indicate presence and/or to estimate abundance. Sign is usually directly observable, but often requires ancillary information on deposition and decay rates and age in order to interpret the meaning of estimates based on sign. It may make sense to choose the survey target based on the comparative detection probabilities and costs.

Note that not all decisions need to be made in all cases. Sometimes making certain choices eliminates the need for other decisions (see figures 2a and b for details). In the next two sections we briefly describe some of the more widely used methods other than those relevant for density/abundance estimation of a wildlife population, namely occupancy and species richness.

Occupancy

Often, estimation of abundance or density is logistically or financially prohibitive. In these cases, an alternative state variable to consider is the proportion of area occupied (PAO). PAO is an estimate of the species distribution or the area of use, based on three possible states: the site is occupied and a species is detected, the site is unoccupied and the species is not detected, and the site is occupied but the species is not detected. The concept is similar to abundance estimation, but instead of estimating number of animals, we estimate number of occupied sampling units (MacKenzie *et al.,* 2002, 2006). Because we need to account for animals that are present but not detected, estimation of a detection probability is a key feature of occupancy analysis. Estimating PAO has a number of practical advantages. The data are relatively easy to collect, we can use multiple sampling methods, and the interpretation is straightforward. PAO may be the most reliable metric for large landscapes because it is likely to be more robust to local and stochastic effects than estimates of abundance or density. PAO can also be easily related to covariates of interest such as habitat and exploitation.

Occupancy analysis offers great flexibility in design and analysis. The basic feature of the method is the need for replicate visits to the sampling unit. The replicates however, may be spatial or temporal, and may be carried out by repeat visits by the same investigator or simultaneous visits by several independant investigators. We can consider multiple occupancy states (classes of relative abundance for example), habitat suitability, and other covariates that affect detection and occupancy. We can also use open and closed models that allow us to monitor occupancy over time in a single analytical framework. Open models also allow for estimation of extinction and colonization rates and may be used to track community and meta-population dynamics (MacKenzie *et al.,* 2003, 2006).

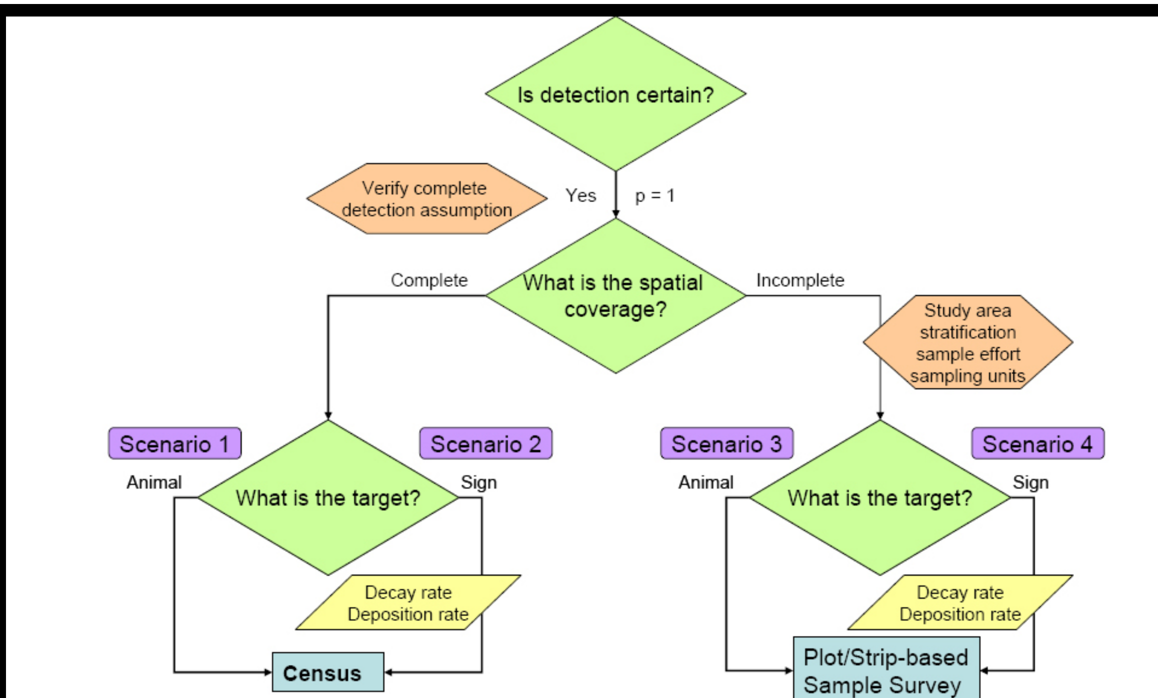


Figure 2a: If it is assumed that detection of the target is certain (p = 1), then this requires verification. If spatial coverage is complete and all individuals in a population are counted, then this is referred to as a census. A census is rarely possible, but if everything is observable and counted in an area of interest, then no statistical analysis is required, as the result is simply a single number with no associated variance. The target may be individual animals (scenario 1) or their sign (scenario 2). An example of the former is the 2008 Ewaso Nyiro elephant survey that attempted to count all elephants in the Ewaso Nyiro watershed (30,000 km^2) of northern Kenya. Examples of the latter might include bird call cue counts or fixed width elephant dung counts in a small area. In this situation, ancillary information is required to interpret the sign (estimates of calling/deposition rates and also decay rates for dung).
If complete spatial coverage is not possible, then a sample survey is conducted in a set of sampling units and the results extrapolated to the entire area of interest to obtain the population size. It is assumed that all animals (scenario 3) or sign (scenario 4) within a sampling unit are detected and counted without error and again requires deposition and decay rate estimates for the latter. Thus the variance associated with the density or abundance estimates is solely due to spatial distribution of individual animals or sign. A sample survey requires careful definition of the study area, and may benefit from stratification to improve precision, needs decisions to be made about sampling effort to obtain an acceptable balance between precision and costs, and finally the sampling units should be defined and ideally be located by means of a random design or systematic design with a random start. Aerial surveys in open habitats that follow strip transects or surveys of dung in sampling plots are some examples.

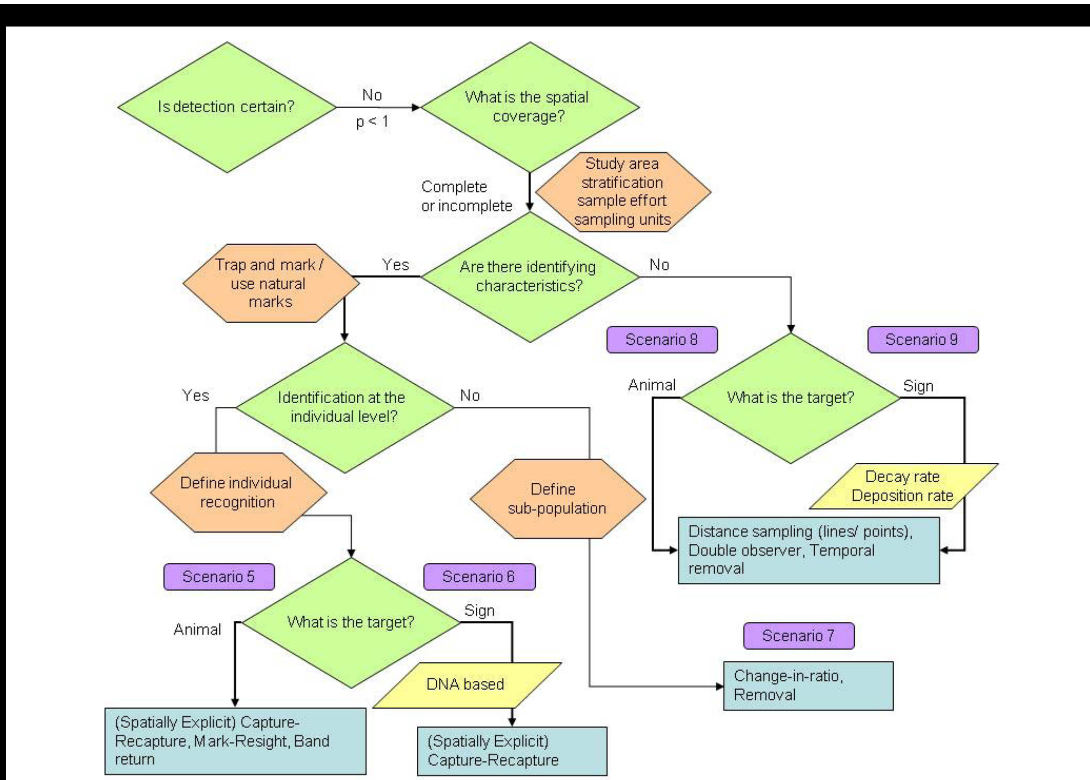


Figure 2b: When detection is less than one and varies over time and space, we must know or estimate the detection probability to obtain unbiased estimates. The potential methods available in this case are independent of whether or not spatial coverage is complete. In the former case the variance of the estimate will be solely due to variance in detection and in the latter case the variance of the estimate will be composed of spatial variance and variance in detection (and decisions will need to be made about the definition of the study area, potential stratification, amount of sampling effort, and definition and placement of the sampling units).
If the target has characteristics that can be used for identification (either due to trapping and marking of individuals or natural markings), then it may be possible to identify individuals (spotted cats, ringed birds). When dealing with animals (scenario 5), capture-recapture (spatially explicit or not), mark-resight or band return techniques might be used, for example. When dealing with sign (scenario 6), DNA analysis and identification of individuals permits the use of capture-recapture techniques, for example. If it is only possible to identify an individual as belonging to a sub-population (scenario 7), then once the sub-population has been defined it is possible to apply change-in-ratio or removal techniques, for example. When identifying characteristics are not available or not made use of, then for both animals (scenario 8) and sign (scenario 9) methods such as distance sampling, double observer or temporal removal may be used (again for sign deposition and decay rates are ideally required).

Species Richness

Species richness, defined as the number of species occupying a delineated area, is an increasingly important state variable for conservation of communities and biodiversity. There are many approaches to estimating species richness including the extrapolation of species-area or species-effort curves, the use of parametric models of species abundance based on count statistics, use of taxon ratios, and estimation of species richness based on sampling. For reviews of these methods see Magurran (1988, 2003). Burnham and Overton (1979) suggested the use of population estimation models to incorporate heterogeneity in species-specific detection probabilities into estimates of species richness. They recommend a model similar to the M_h estimator in closed population estimation permitting heterogeneous capture probabilities among individuals (Otis *et al.*, 1978; Williams *et al.*, 2002). An extension of this approach to multi-year studies allows estimation of the rate parameters of local extinction and colonization and can be implemented using Pollock's (1982) robust design and the software package COMDYN.

Cam *et al.* (2002) present a probabilistic, non-parametric estimator of species richness for use with species accumulation data. They make the connection between species richness estimation and abundance estimation using capture-removal models in which the detection probability changes after the first detection (Model M_b: Otis *et al.,* 1978). Removal models are appropriate for species accumulation data because a species is removed from the population after its first detection so the only statistics are the number of new species detected at each sample period.

Species richness also can be estimated from detection/non-detection data (MacKenzie *et al.,* 2006). Occupancy models are especially useful for estimation involving site level species richness when a list of potential species occurring at the site or in the region is available. Species richness at a particular site will be determined by local environmental conditions (i.e., habitat) and by the regional species pool that contains all possible species for the area. When the regional species pool is known, each species may serve as a "site" in the context of occupancy sampling. The estimate of occupancy is interpreted as the proportion of species from the regional pool that occur at the site. The "number of sites occupied" is the estimate of species richness. Occupancy modeling allows tracking of changes in species richness over time and the modeling of covariates that might affect detectability, extinction or colonization (MacKenzie *et al.,* 2006). The use of covariates is not possible in the capture-recapture models of Cam *et al.* (2000) because undetected species are not used in the estimation, providing no ability to use covariate information of such species.

Standardization of Monitoring Methods

Although, progress has been made on standardizing methods across landscapes or for a particular species within WCS (e.g. tiger and prey monitoring, forest elephants and apes) a great deal of work remains in this regard. In addition, a key element is tying this standardization to strategic planning and the development of monitoring frameworks that clearly detail our goals and objectives in terms of wildlife or habitat status and the desired reduction of threats. Without explicit goals and objectives it is impossible to know whether or not we are successful in our work and which activities tend to be successful.

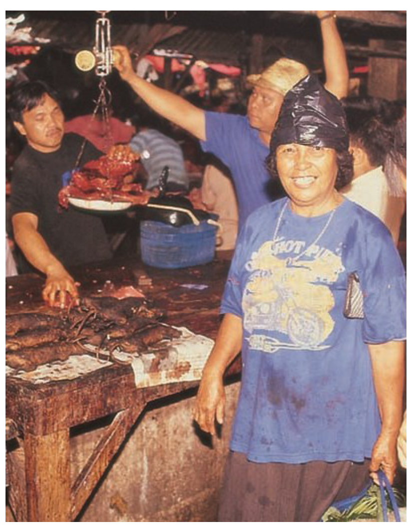
As an example, a 2009 workshop on "Strategic Planning for Conservation Management Across Landscapes" that included managers from a number of WCS Latin America and Caribbean Program landscapes, as well as species specialists focused on monitoring in different landscapes, provided some instructive ideas and insights. After a review of monitoring techniques, many discussions, and consideration of past experience, workshop participants thought it made sense to standardize sampling designs and monitoring protocols for species, communities (human and wildlife) and potentially also indicators that are monitored on a regional level (multiple country programs, multiple sites within countries). This may include technical aspects of sample design, analytical methods, and development of aggregate or headline indicators.

To form a picture of the most commonly used wildlife monitoring methods and the associated target species, we conducted a survey of the workshop participants. The results lend weight to the idea that the set of methods being used (or available) is reasonably small (see Table 1), which should make standardization a reasonable possibility. The results of this survey also gives some indication where resources might initially be invested to have the largest impact.

A few key techniques, such as capture-recapture (with camera trapping or DNA-based), distance sampling, catch per unit effort (preferably with associated model-based analysis to account for imperfect detectability that is unknown and not fixed across time or space), questionnaire surveys, could be the focus of this standardization. A first step would be the collection of existing monitoring protocols from the field sites or other sources and then to standardize and improve these where necessary. Collation and development of protocols and implementation manuals to guide development of sampling designs and analysis of species and communities of interest would be made available more broadly to WCS staff and others via a website, which could include links to already existing, good protocols available on other websites, as well as other resource materials (list servers, papers).

Aside from the further development of these protocols and implementation manuals, workshop participants thought it would be useful to put together an overview paper describing the various techniques and their applications (e.g., the variety of applications of presence surveys) with a synthesis of best practices across WCS that could be used as an overview working paper for reference. For all protocols workshop participants asked that we consider options for pooling data across studies in order to improve accuracy and precision.

Conclusions



Market scene in Sulawesi.
© M. Kinnaird

It certainly seems to be the case that methods available for monitoring wildlife are a fairly specialized and small set. In addition, in most cases the options in terms of choosing a method will be further limited by the characteristics of the conservation target and the habitat. Finding a balance between the costs of implementing the method, the available technical capacity and the required monitoring information that can appropriately inform management will also be part of the decision making process.

This relatively small set of methods available for monitoring wildlife and the desire to monitor the effectiveness of our conservation actions argues for an appropriate and comparable application of methods across WCS. Although methods will always be implemented in a manner that takes into account the characteristics of a particular species, there are still measures that can be taken to ensure that methods are correctly implemented and standardized where appropriate.

Some of the more commonly used methods shown in the nodes of figures 2a and b and mentioned previously are briefly detailed in Appendix 1. A literature review with some of the key references and internet resources is listed in Appendix 2. A small set of examples of monitoring applications at WCS sites that make use of these methods is given in Appendix 3.

TABLE 1. APPLICATION OF MONITORING METHODS AT 10 LANDSCAPE SITES IN LATIN AMERICA BASED ON AN ANALYSIS OF 120 COMBINATIONS OF TAXONOMIC GROUP, SPECIES, PROJECT GOALS AND INDICATORS. NUMBERS INDICATE FREQUENCY OF USE.

	Capture-Recapture		Harvest			Plot/Transect/Point Surveys		Other Methods						Total by group
Taxonomic Group	Animal	DNA	CPUE	Interview	Self-report	Terrestrial	Aerial	Occupancy	Census	Relative Abundance	Nest monitoring	Telemetry	Interviews	
Birds	1			1		9			3		2	1		17
Fish			9		1					3				13
Crocodiles			1	1		9		1			2			14
Turtles	2			1		4			4		2			13
Reptiles					1	1								2
Arboreal primates						10								10
River mammals	2								2	5		1		10
Terrestrial mammals/birds	14	5	1	4	8	12	1		3	11		2	2	63
Total by method	**19**	**5**	**11**	**7**	**10**	**45**	**1**	**1**	**12**	**19**	**6**	**4**	**2**	**142**

APPENDIX 1. OVERVIEW OF FREQUENTLY USED WILDLIFE MONITORING SURVEY TECHNIQUES FOR ESTIMATING ABUNDANCE, OCCUPANCY, OR DEMOGRAPHIC RATES

Occupancy Methods

Occupancy methods estimate the proportion of a habitat or number of patches occupied when detection is incomplete (Mackenzie *et al.*, 2002, 2003, 2006). The analysis recognizes three states: occupied and detected, occupied and not detected, and not occupied. It provides estimates of the probability that a sampling unit is occupied and the probability that an individual animal (or sign, if sign surveys are used) is detected. It requires replicated observations on each sampling unit and it allows for covariates that might affect occupancy or detection to be incorporated into the analysis. The basic method assumes demographic and spatial closure during a sampling period (referred to as a season) such that the occupancy status does not change and that sampling units states are independent. Additional assumptions include no errors in identifying species and that observations are independent. There are analysis options that relax most of these assumptions should this be needed. The methods are continually evolving and some of the analysis options currently available include:

- **Single Season** – estimates the proportion of occupied sampling units, detection probability, and estimates of covariate effects.
- **Multiple Seasons** – estimates include above plus estimates of colonization and extinction rates of sampling units, and estimates of covariate effects on rates.
- **Species Interactions** – allows the estimation of co-occurrence of species.
- **Spatial Autocorrelation** – relaxes the assumption that sampling units are spatially independent.
- **Multi-Method** – allows detection probabilities to be different for different methods of observation.
- **Multi-State** – allows the estimation of the probability that animals are in a given state, given that they are present, which is especially useful for relative abundance data. Multi-state models allow a species to occupy a site at different levels of abundance and to evaluate factors affecting the occurrence and abundance of a species on the landscape.
- **Point Count** – estimates population size from point-count data.
- **Habitat Suitability** – estimates occupancy as a function of site suitability.
- **Simultaneous Modeling of Habitat Suitability, Occupancy and Relative Abundance** – allows for estimation of transition probabilities between habitats and abundance.

The Presence software facilitates analysis of occupancy data and can be used for single species studies, community level studies and estimation of species richness. It is available as a free download from Patuxent Software Archive (http://www.mbr.pwrc.usgs.gov/software.html). Occupancy analysis can also be carried out in R using the Unmarked package (http://github.com/rbchan/unmarked).

Distance Sampling

Distance sampling is one of a number of survey methods that can be used to estimate animal density or abundance (Buckland *et al.*, 2001). The key to distance sampling is recording perpendicular distance to each observation (or radial distances for points) and fitting a detection function to these data that can be used to estimate both the proportion of animals detected and counted and the proportion of the survey area covered. Thus, the canonical estimator (Eq. 3) can be applied to the raw counts to obtain an unbiased estimate of abundance. Ideally transect lines or points are located randomly with respect to the distribution of the animals, which helps ensure valid statistical inference. Additional assumptions when using the standard method include that objects of interest on the line or point are detected with certainty, animals are detected at their initial location, measurements are exact, and that detections are independent events.

For distance sampling to be successfully applied it is essential that detectability decreases as distance from the transect line or point increases and that the distance between the observer and each target can be obtained accurately. Distance sampling works well for populations in well-defined groups or detected through a flushing response and can be very efficient and cost-effective for large populations, populations at low or medium individual or group density, and populations sparsely distributed over large geographic regions. In particular, point transects might be most appropriate for populations at high density, for multi-species surveys (e.g. songbirds), or when habitat is patchy or terrain is difficult, making it problematic to walk along predetermined lines. Advantages are that the detection function is robust to unmodeled heterogeneity and that repeated surveys are not required unlike occupancy or capture-recapture surveys. Distance sampling methods are continually evolving with innovations in spatial modeling using distance sampling data, incorporating covariates into the detection function, combining distance sampling with mark-recapture methods, automated survey design, for example (Buckland *et al.*, 2004)

Fortunately, the freely available Distance software exists to help with distance sampling design and analysis (Thomas *et al.*, 2010). It comes with a comprehensive online users' guide and can be downloaded from the Distance website (www.ruwpa.st-and.ac.uk).

Capture-Recapture

Capture-recapture techniques comprise a continually evolving set of methods to estimate state and rate parameters. The methods require recaptures (active or passive) of animals that can be individually identified or sub-populations that can be recognized either through tags or natural marking (or through DNA). A key assumption is that marked animals are representative of the entire population of interest and that marks are not lost (or do not change in the case of natural markings). Unmodeled heterogeneity in capture probabilities create biases in the estimates and every attempt must be made to account for this heterogeneity that may be due, for example, to reactions to physical trapping, differences in the natural behavior of individuals or changes in behavior over time. Some of the most well known capture-recapture methods include known fate models, Cormack-Jolly-Seber models, closed models, band recovery/exploitation models, multi-state models or combinations of these.

The Mark software that offers an astonishing list of analysis options is the state of the art software for the analysis of capture-recapture data (www.cnr.colostate.edu/~gwhite/ software.html). Mark's online help is comprehensive and in addition the e-book compiled by Evan Cooch, Program Mark: A Gentle Introduction, provides a wealth of a information(http://www.phidot.org/software/mark/docs/book/).

Spatially Explicit Capture-Recapture

Spatially explicit capture-recapture (SECR) uses the locations where each animal is detected to fit a spatial model of the detection process, and hence to obtain estimates of population density unbiased by edge effects and incomplete detection. Previously, the conventional approach to the analysis of animal density from trap surveys was to apply closed capture-recapture model analyses, and, then convert resulting estimates of abundances to densities using a wide range of essentially ad hoc methods. While these approaches appear to work adequately in practice, little had been known about the range of conditions under which they work well. This is because most real world study situations involve study areas of odd shapes and sizes and difficult terrain that makes setting traps challenging and conditions assumed by ad hoc approaches may not apply. Detections may take place by means of live-capture traps, with animals uniquely marked; they also may be sticky traps or snags that passively sample hair, from which individuals are distinguished by their DNA microsatellites, or cameras that take photographs from which individuals are recognized by their natural marks.

The Density software uses maximum likelihood to estimate the density of animal populations from spatially explicit capture-recapture data (www.otago.ac.nz/density). The SECR library developed for the R statistical software implements an even wider range of spatially explicit capture-recapture analysis options using maximum likelihood methods. The SPACECAP library for R implements a set of Bayesian spatially explicit capture-recapture models. It was developed specifically for tiger camera trap data.

Mark-Resight

Mark-resight methods rely on resightings rather than recaptures of individuals or recognition of marked sub-populations. They are a variation on the mark-recapture theme in that they account for imperfect detection during the estimation process and in addition utilize information on the sightings of unmarked individuals. Previously the main focus of mark-resight methods was on abundance estimation (Neal *et al*., 1993; Bowden and Kufeld 1995); however, recent developments in mark-resight models now permit the use of the robust design and an integrated approach to estimate survival and transition rates between observable and unobservable states, as well as allowing for individual heterogeneity in sightability (McClintock *et al*., 2006; McClintock and White 2009). Fortunately, these recently developed analysis options are available in the Mark software package.

Demography

Some studies are interested in monitoring demographic performance as a function of individual performance. In these studies identified individuals are followed as a cohort over time. Either a census can be conducted where all members of the cohort are identified and their presence in the population verified. New individuals are added (births and immigrations) and disappearances are noted (death, emigration). The assumption is that all individuals in the population are accounted for in each census. Examples include primate and elephant demography monitoring which typically assumes the cohort is defined by the area occupied, or an area visited.

Demographic analysis can be implemented in mathematical software packages such as MATHEMATICA or MATLAB. The Demography library for R implements functions for demographic analysis including lifetable calculations, Lee-Carter modeling, functional data analysis of mortality rates, fertility rates, net migration numbers, and stochastic population forecasting. Survival analysis is featured in many statistical software packages such as SPSS and SAS. University of Vermont Cooperative Fish and Wildlife Research Unit Spreadsheet Project offers Excel spreadsheets for age- and stage-structured life table analysis as well as instructions in the use of life tables. The specialized software Mayfield provides simple analysis options for nest survival.

An alternative to life table analysis is to use open population capture-recapture analysis in conjunction with Pollock's robust design to estimate demographic parameters (survival, mortality, immigration, emigration) for populations that include a marked sub-population. Mark is the most complete software for estimating demographic parameters from data that include marked individuals. In addition to standard models, Mark includes the ability to incorporate covariates that might affect parameters, e.g. analysis options for nest survival data. Analysis options previously available in other specialized software have been mainly incorporated into Mark. For example, Jolly-Seber-type models for open population mark-recapture data available in POPAN (<u>POP</u>ulation <u>An</u>alysis) can now be accessed in Mark.

APPENDIX 2. LITERATURE AND INTERNET RESOURCES

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INTERNET RESOURCES

Patuxent Wildlife Research Center

http://www.mbr-pwrc.usgs.gov/software

CAPTURE

PRESENCE

MAYFIELD

COMDYN

Many others

Colorado State University Department of Fishery and Wildlife Biology
http://www.cnr.colostate.edu/~gwhite/software.html

MARK

University of Otago

http://www.otago.ac.nz/density

DENSITY

Evan Cooch's software page

http://www.phidot.org/software

Links to other population analysis software

Research Unit for Wildlife Population Assessment

http://www.ruwpa.st-and.ac.uk/

DISTANCE

University of Vermont, Vermont Cooperative Fish and Wildlife Research Unit Spreadsheet Project

http://www.uvm.edu/rsenr/vtcfwru/spreadsheets/

Spreadsheet exercises for population analysis

APPENDIX 3. EXAMPLES OF WCS WILDLIFE MONITORING APPLICATIONS

Multiple Landscapes Monitoring Using Standardized Methods

Tigers Forever

Tiger prey - line transect density estimation (DISTANCE), occupancy and point abundance estimation (PRESENCE)

Tigers - density (camera trapping but different analytical methods at different sites)

Tigers - occupancy (PRESENCE)

Humpback Whales

Population estimation using DNA or fluke identification - capture recapture (CAPTURE, MARK)

Tropical Ecology Assessment and Monitoring

Terrestrial wildlife - camera-trapping and the Wildlife Picture Index (PRESENCE)

Tree/Liana - plot-based cohorts

Climate

Albertine Rift

Terrestrial wildlife camera traps

Birds - point transects (DISTANCE)

Primates - line transects (DISTANCE)

Climate

Landscape-Scale Monitoring Single Sites

Congo Africa

Great apes - line transect sign surveys (DISTANCE)

Forest Elephants - line transect sign surveys (DISTANCE)

Sudan

Large mammals - line transect aerial surveys (assume detectability is certain)

Zambia

Large mammals - line transect aerial surveys (assume detectability is certain)

Kenya

Large mammals, terrestrial birds - line-transect surveys (DISTANCE), camera traps

Elephants - cohort-based, line transect surveys (DISTANCE), aerial surveys

Livestock-wildlife interactions - camera traps

Savanna and forest birds - point count surveys

Site-Scale Monitoring (Small Spatial Scale)

Indonesia

Siamang/gibbon demography - cohort

Vegetation dynamics - plot-based cohort

Primates, hornbills, ungulates, birds - line/point transect (DISTANCE), camera trapping (PRESENCE)

Bangladesh

Bottle-nosed dolphins - photo-identification mark-resight (MARK)

Belize

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