Chapter 7
Inventory and Monitoring Studies

7.1 Introduction

Inventory and monitoring are probably the most frequently conducted wildlife studies. Not only are they conducted in the pursuit of new knowledge (e.g., to describe the fauna or habitats [see Sect. 1.5 for definition of habitat and related terms] of a given area, or understand trends or changes of selected parameters), but also they are cornerstones in the management of wildlife resources. In general terms, *inventories* are conducted to determine the distribution and composition of wildlife and wildlife habitats in areas where such information is lacking, and *monitoring* is typically used to understand rates of change or the effects of management practices on wildlife populations and habitats. In application to wildlife, inventory and monitoring are typically applied to species' habitats and populations. Because sampling population parameters can be costly, habitat is often monitored as a surrogate for monitoring populations directly. This is possible, however, only if a clear and direct linkage has been established between the two. By this, we mean that a close correspondence has been identified between key population parameters and one or more variables that comprise a species' habitat. Unfortunately, such clear linkages are lacking for most species.

The need for monitoring and inventory go well beyond simply a scientific pursuit. For example, requirements for monitoring are mandated by key legislation (e.g., National Forest Management Act [1976], National Environmental Policy Act [1969], Endangered Species Act [1973]), thus institutionalizing the need for conducting such studies. Even so, monitoring is embroiled in controversy. The controversy is not so much over the importance or need to conduct monitoring, but surrounds the inadequacy of many programs to implement scientifically credible monitoring programs (Morrison and Marcot 1995; White et al. 1999; Moir and Block 2001). Unfortunately, few inventory/monitoring studies are conducted at an appropriate level of rigor to precisely estimate the selected parameters. Given that inventory and monitoring are key steps in the management process and especially adaptive management (Walters 1986; Moir and Block, 2001), it is crucial to follow a credible, repeatable, and scientific process to provide reliable knowledge (cf. Romesburg 1981). The purpose of this chapter is to outline basic steps that should be followed for inventory and monitoring studies.

THEME: Mexican Spotted Owl (Strix occidentalis lucida)

Throughout this chapter, we will use a theme based on the Mexican spotted owl to illustrate inventory and monitoring concepts. The Mexican spotted owl is a less renown relative of the northern spotted owl (*S. o. caurina*). Like the northern spotted owl, the Mexican subspecies is listed as threatened under the Endangered Species Act, which prompted development of a recovery plan (USDI Fish and Wildlife Service 1995). Much of the information presented in this chapter is gleaned from that recovery plan and the deliberations underlying its development. Box 7.1 provides a brief summary of the salient points of the owl's ecology and management as they relate to points discussed in this chapter.

Box 7.1 Background on the Mexican Spotted Owl

Owl Ecology

Detailed reviews of various aspects of the owl's ecology are provided in the recovery plan (USDI Fish and Wildlife Service 1995). Our intent here is to present salient points about the owl that were key considerations in developing management recommendations. Although the Mexican spotted owl occupies a broad geographic range extending from Utah and Colorado south to the Eje Neovolcanico in Mexico, it occurs in disjunct localities that correspond to isolated mountain and canyon systems. The current distribution mimics its historical extent, with exception of its presumed extirpation from some historically occupied riparian ecosystems in Arizona and New Mexico. Of the areas occupied, the densest populations of owls are found in mixed-conifer forests, with lower numbers occupying pine-oak forests, encinal woodlands, rocky canyons, and other vegetation types. Habitat-use patterns vary throughout the range of the owl and with respect to owl activity. Much of the geographic variation in habitat use corresponds to differences in regional patterns of vegetation and prey availability. Forests used for roosting and nesting often exhibit mature or old-growth structure; specifically, they are uneven-aged, multistoried, of high canopy closure, and have large trees and snags. Little is known of foraging habitat, although it appears that large trees and decadence in the form of logs and snags are consistent components of forested foraging habitat. The quantity and distribution of potential owl habitat, specifically forests that possess relevant habitat correlates, is largely unknown. Existing data sets on forest structure are too variable in quality and in terms of coverage to permit even ballpark guesses.

With the exception of a few population demography studies, little is known of the population ecology of the Mexican spotted owl. The recovery team recognized the limitations of existing data and the inferences that could be drawn from them. Consequently, the recovery team reviewed and reanalyzed those data to estimate appropriate population parameters needed for development of the population monitoring approach that would provide more rigorous and defensible estimates of population trend.

Recovery Plan Management Recommendations

The recovery plan is cast as a three-legged stool with management recommendations as one of the three legs. Three areas of management are provided under the general recommendations: protected areas, restricted areas, and other forest and woodland types. Protected areas receive the highest level of protection. Guidelines for restricted areas are less specific and operate in conjunction with existing management guidelines. Specific guidelines are not proposed for other forest and woodland types.

Protected areas are all occupied nest or roost areas, mixed-conifer and some pine-oak forests with >40% slope where timber harvest has not occurred in the past 20 years, and all legally administered reserved lands (e.g., wilderness). Active management within protected areas should be solely to alleviate threats of catastrophic stand-replacing fires by using a combination of thinning small trees (<22 cm dbh) and prescribed fire.

Restricted areas include mixed-conifer forests, pine-oak forests, and riparian areas not included in protected areas. Management for the owl should focus on maintaining and enhancing selected restricted areas to become replacement nest and roost habitat, and abating risk of catastrophic fire in much of the restricted habitat. The amount of restricted area to be managed as replacement habitat varies with forest type and location, but ranges between 10 and 25% of the restricted area landscape. Thus, between 75 and 90% of restricted areas can be managed to address other resource objectives.

No specific guidelines are provided for other forest and woodland types – primarily ponderosa pine (*Pinus ponderosa*) and spruce-fir (*Picea* spp.-*Abies* spp.) forests, and pinyon-juniper (*Pinus* spp.-*Juniperus* spp.) and trembling aspen (*Populus tremuloides*) woodlands – outside of protected areas. However, some relevant management of these vegetation types may produce desirable results for owl recovery. Examples of extant guidelines include managing for landscape diversity, mimicking natural disturbance patterns, incorporating natural variation in stand conditions, retaining special habitat elements such as snags and large trees, and using fire appropriately.

In addition, some guidelines were proposed related to specific land use, such as grazing and recreation, and these guidelines apply to all management areas. The team recognized that effects of such activities on spotted owls are not well known, and advocated monitoring potential effects to provide a basis for more specific recommendations if warranted.

Because aspects of owl ecology, biogeography, and management practices varied geographically, the recovery team divided the range of the Mexican spotted owl into 11 recovery units: six in the United States and five in Mexico (Rinkevich et al. 1995). Recovery units were based on physiographic provinces, biotic regimes, perceived threats to owls or their habitat, administrative boundaries, and known patterns of owl distribution.

(continued)

Box 7.1 (continued)

By and large, the management recommendations allowed resource agencies considerable latitude in designing and implementing activities. The general philosophy of the team was to protect habitat where it existed, and to enhance habitat where appropriate. Whether or not the management recommendations are successful could be measured only through habitat and population monitoring, the other two legs of the stool. Without monitoring, there would be no empirical and objective basis for determining whether management guidelines led to desired outcomes and whether the owl should be delisted.

Delisting Criteria

Delisting the Mexican spotted owl will require meeting five specific criteria (USDI Fish and Wildlife Service 1995, pp. 76–77). Three of these criteria pertain to the entire US range of the owl, and two are recovery unit specific. The three range-wide delisting criteria are:

1. The populations in the Upper Gila Mountains, Basin and Range-East, and Basin and Range-West recovery units must be shown to be stable or increasing after 10 year of monitoring, using a design with power of 90% to detect a 20% decline with a Type I error rate of 0.05.
2. Scientifically valid habitat monitoring protocols are designed and implemented to assess (1) gross changes in habitat quality across the range of the Mexican spotted owl and (2) whether microhabitat modifications and trajectories within treated stands meet the intent of the Recovery Plan.
3. A long-term, US-range-wide management plan is in place to ensure appropriate management of the subspecies and adequate regulation of human activity over time.

Once these three criteria have been met, delisting may occur in any recovery unit that meets the final two criteria.

4. Threats to the Mexican spotted owl within the recovery unit are sufficiently moderated and/or regulated.
5. Habitat of a quality to sustain persistent populations is stable or increasing within the recovery unit.

Implicit to the delisting criteria is the need for reliable, defensible data to (1) assess population status, (2) habitat trends, and (3) develop long-term management guidelines. Without such information, the recovery team felt that risks to the threatened owl would be too great. As an example of the team's philosophy, we detail the population monitoring approach presented in the recovery plan, and discuss ramifications of failure to implement population monitoring (see Box 7.5).

7.2 Selection of Goals

Inventory and monitoring studies entail similar, but distinct processes. Although some steps are clearly identical, others are specific to the type of study being done (Fig. 7.1). The first step, which is universal to any study, is to clearly state the goals. For example, why conduct the study? What information is needed? How will the information be used in this or future management decisions? Clearly answering these questions will help to define a study design that addresses them adequately. That is, establishing inventory and monitoring goals is critical for defining what will be monitored (e.g., selected species or all species, population variables or habitat variables), setting target and threshold values, designing appropriate protocols for collecting data, and determining the appropriate methods for data analysis.

7.3 Basic Design Applications

Researchers and managers conduct inventory and monitoring to meet a variety of needs. These can range from basic needs such as characterizing species occurrence to monitoring effects of management activities on population status and trends of species of interest. We elaborate on basic applications of inventory and monitoring below.

7.3.1 Inventory

An inventory assesses the *state* or status of one or more resources. It should be designed to provide information on an environmental characteristic, such as the distribution, population, or habitat of a given species or suite of species. An inventory

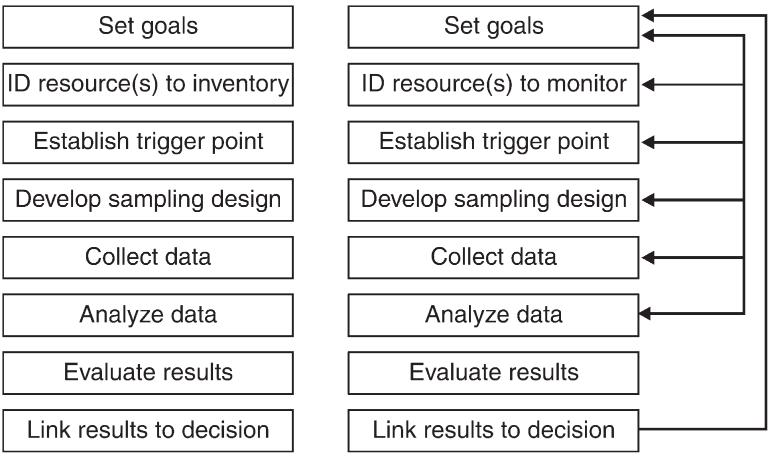


Fig. 7.1 Simplified sequence of steps involved with inventory and monitoring

provides a quantitative or qualitative description of a resource or set of resources for a given point or interval of time. Inventories are typically confined within a specific area or set of areas to determine the number and perhaps relative abundance of the species present, and they must be conducted at appropriate spatial scales and for appropriate durations depending on the resource(s) under study and the question(s) being asked. Inventories may take many years and require spatially extensive sampling to meet study goals. For example, inventories to estimate the density of rare species such as a far-ranging predator may require sampling more area than needed to estimate the density of a common species. Developing a list of breeding birds will require sampling only during the breeding season, whereas acquiring a list of all birds that use an area requires sampling year-round to record migrating and wintering birds. Further, sampling the breeding bird community will require a certain sampling effort (e.g., sampling points, duration) to provide an unbiased estimate of the species using an area. As an example, Block et al. (1994) used a bootstrap technique (Efron 1982) to estimate the number of counting stations and number of years needed to account for all species using oak woodlands during the spring breeding season (Fig. 7.2). They found that 56 counting stations sampling about 175 ha were required to record all species detected during a 2-year period, but

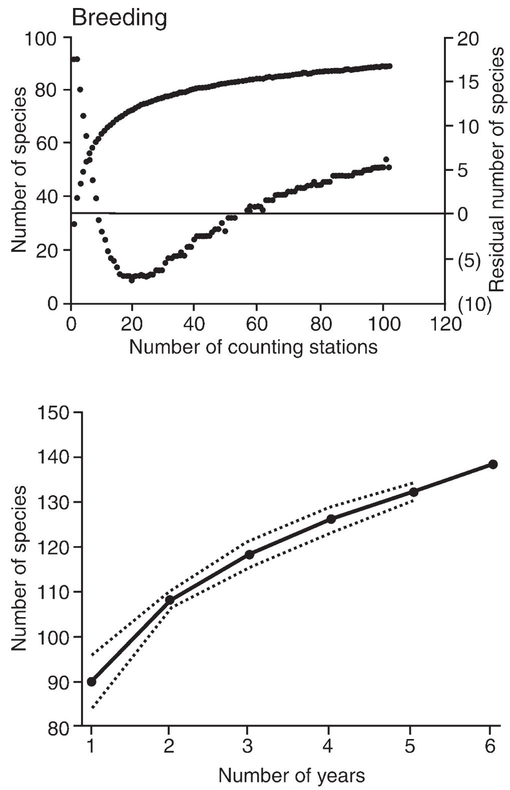


Fig. 7.2 Example of the number of (**a**) counting stations and (**b**) number of years to detect most birds using oak woodlands in California's Sierra Nevada foothills. Reproduced from Block et al. (1994), with kind permission from The Wildlife Society

that new species were being detected even after 6 years of sampling, likely because it is extremely difficult to detect all rare and incidental species.

Inventories are typically done in areas or conditions for which data are lacking, or across a range of areas or conditions to more clearly define ecological distribution of a species (e.g., define both presence and absence) (Heyer et al. 1994). A typical goal of inventories is to assess the presence or infer absence of species within an area prior to initiating a habitat-altering activity. Note that we state "infer absence." Verifying presence is straightforward; if you see or otherwise detect a species then it uses the area. However, failure to detect a species does not necessarily translate into it being absent when you sampled or that it never uses the area. This is where designing and implementing an appropriate study design is critical. The study design must be such that the probability of detecting a species or individuals using the area is high. Some important associated components are use of the proper field methodology and sampling during the appropriate period and with adequate intensity. We cannot reiterate these points enough because proper study design is the foundation of a valid wildlife study (cf. Chaps. 1 and 2).

Thus, inventories are critical tools to aid in resource planning and species conservation. Even basic information on the distribution of species and habitats in an area can then help design management to protect or enhance conditions for desired species, whether they are threatened or endangered species or those valued for consumptive or nonconsumptive reasons.

7.3.2 Monitoring

In contrast to inventory, monitoring assesses change or trend of one or more resources. The difference between change and trend is subtle in that change is evaluated by testing for differences between two points in time, whereas trend typically requires sampling for more than two occasions to evaluate the direction and consistency of change. Either way, both change and trend measure or index the dynamics as opposed to the state of a resource. Thus, monitoring requires repeated sampling of the variable(s) of interest to measure the change or trend. The variables measured and techniques used to measure them often overlap with those used for inventories. Some variables, however, may be unique to monitoring, especially those that measure rates, such as survival, and those that require repeated measures such as habitat succession. As with inventories, monitoring studies must be scaled to the variable and question being addressed. Thus, if one is assessing changes in forest structure and composition as they relate to a species' habitat, monitoring must be scaled temporally to vegetative processes. Monitoring a population to determine population trend must occur over a long enough time to be sure that the population has been subjected to an appropriate range of environmental variations. For example, populations studied during favorable weather conditions may exhibit positive trends, whereas those studied during unfavorable weather may show just the opposite. To guard against this potential bias, it is important to scale study duration long enough to include these variations.

Monitoring can include studies specifically to evaluate effects of a particular environmental treatment or impact on the resource of interest, or it could entail examining general trend without considerations of any specific activity. Impact assessment was discussed in detail in Chap. 6 so we refer you to that chapter for more detailed discussion of that particular topic. A more common monitoring study is to examine population or habitat trend, regardless of the causal factors. For example, is abundance of the Mexican spotted owl stable or increasing? Are macrohabitat and microhabitat of the owl stable or increasing? Answering these basic questions is key to the management process. If these trends are determined to be negative, one then could conduct more directed impact assessment monitoring to evaluate potential causal mechanisms.

Some broad objectives for conducting monitoring include (from Spellerberg 1991):

1. To provide guidance to wildlife management and conservation
2. To better integrate wildlife conservation and management with other land uses
3. To advance basic knowledge in addition to applied knowledge
4. To track potential problems before they become real problems

These objectives are often addressed by conducting monitoring studies (from Gray et al. 1996; Miller 1996):

1. To determine wildlife use of a particular resource (e.g., herbaceous forage) or area
2. To evaluate effects of land use on populations or habitats, measure changes in population parameters (e.g., size, density, survival, reproduction, turnover)
3. To evaluate success of predictive models
4. To assess faunal changes over time

Monitoring can be classified into four overlapping categories: implementation, effectiveness, validation, and compliance monitoring. *Implementation monitoring* is used to assess whether a directed management activity has been carried out as designed. For example, a prescribed fire is done as a habitat improvement project and the goal of the fire is to reduce fine ground fuels by 50%. Implementation monitoring would be done to evaluate whether that goal would be met. *Effectiveness monitoring* is used to evaluate whether or not the action met its stated objective. Say, for example, that the ultimate objective of the prescribed fire was to increase population numbers of the deer mouse (*Peromyscus maniculatus*). Effectiveness monitoring would involve a study to evaluate the response of the deer mouse population to the treatment. *Validation monitoring* is used to evaluate whether established management direction (e.g., National Forest Plans) provides guidance to meet its stated objectives (e.g., sustainable forest management) (Morrison and Marcot 1995). It is also used to test assumptions of models or prescriptions used to develop management decisions. This type of monitoring can be the most difficult to categorize as it often involves multiple resources and ambiguous goals. For example, forest sustainability is a laudable goal, but typically is nebulously defined. Determining exactly what to measure, and how to measure it, can be difficult indeed. On the other hand, management plans often contain specific and measurable criteria, such as the desired amount of forest in a given structural class (e.g., mature or old-growth forest), or the number of a given habitat element that should occur across the landscape. For these criteria, establishing a valid monitoring study is not nearly as challenging. *Compliance monitoring* is done when mandated by statute (see Sect. 1.3.2). An example of compliance is monitoring established within a biological opinion provided by the US Fish and Wildlife Service during interagency consultation under the Endangered Species Act. Typically, this monitoring, referred to as *take monitoring*, assesses whether an activity adversely affects the occupancy or habitat of a threatened or endangered species. If so, the action agency is charged with a "take," meaning that the activity had an adverse impact on a specified number of the species. To illustrate further the different types of monitoring, we draw upon our theme, the Mexican spotted owl (Box 7.2).

Box 7.2 Monitoring for a Threatened Species: The Mexican Spotted Owl

Different monitoring goals are illustrated in the spotted owl example. The extent to which management activities are actually applied on the ground and the degree to which those activities are in accord with recovery plan guidelines would be evaluated by implementation monitoring. For example, consider a silvicultural prescription with the ultimate objective of creating owl nesting habitat within 20 year (the criteria for nesting habitat were provided in the recovery plan). The prescription entailed decreasing tree basal area by 15% and changing the size class distribution of trees from one skewed toward smaller trees to an equal distribution of size classes. Further, the recovery plan specifies the retention of key correlates of owl habitat – trees >60 cm dbh, large snags, and large downed logs – during active management practices such as logging and prescribed burning. In this case, implementation monitoring must have two primary objectives. One is to determine if losses of key habitat elements exceeded acceptable levels, and the second is to determine if tree basal area was reduced as planned and the resultant size class distribution of trees was even. Recall that the ultimate objective of the treatment was to produce a stand in 20 year that had attributes of owl nesting habitat. Whether or not the prescription achieved this objective is the goal of effectiveness monitoring.

The owl recovery plan (USDI Fish and Wildlife Service 1995) provided five delisting criteria that must be met before the owl should be removed from the list of threatened and endangered species. One criterion was to demonstrate that the three "core populations" were stable or increasing, and another required habitat stability across the range of the subspecies. The recovery plan became official guidance for the US Fish and Wildlife Service, and then for the US Forest Service as they amended Forest Plans for all forests in the southwestern region to incorporate the recovery plan recommendations (USDA Forest Service 1996). For a little background, National Forests are mandated to develop Forest Plans by the National Forest Management Act (NFMA) of 1976, thus making

(continued)

Box 7.2 (continued)

Forest Plans a legal requirement. Guidance in Forest Plans is provided by a series of standards and guidelines, which must be followed in planning and conducting management activities. The ultimate goal of Forest Plans with respect to the Mexican spotted owl was to implement the recovery plan, and ultimately delist the owl. Whether or not implementing Forest Plans provides conditions for a viable population of owls and results in delisting is measured through validation monitoring. Two tangible measures for the owl would be to demonstrate that both owl habitat and owl populations were stable or increasing.

Compliance monitoring is done as part of the terms and conditions set forth in a biological opinion resulting from interagency consultation. For example, a form of compliance monitoring would be to monitor for "take" of owls or habitat. Take of owls could be assessed by abandonment of a territory or change in reproductive output. Take of habitat would involve reduction of key habitat components below some minimum threshold.

Monitoring can be used to measure natural or intrinsic rates of change over time or to understand effects of anthropogenic or extrinsic factors on population or habitat change or trends. By intrinsic changes, we refer to those that might occur in the absence of human impact, such as trends or changes resulting from natural processes (e.g., succession) or disturbances (fire, weather, etc.) (Franklin 1989). Anthropogenic factors are those that may alter or disrupt natural processes and disturbances and potentially affect wildlife habitats or populations. In most management situations, monitoring is conducted to understand effects of anthropogenic factors (e.g., water diversions, livestock, logging, fire suppression) on wildlife. However, recognizing trends even in the absence of anthropogenic factors is complicated by the dynamic and often chaotic behavior of ecological systems (Allen and Hoekstra 1992). Because intrinsic and extrinsic factors more often than not act synergistically to influence trend or change, the effects of either may be difficult to distinguish (Noon et al. 1999). Again, this is where application of an appropriate study design plan is critically important. A well-conceived and well-executed study may allow the investigator to partition sources of variation and narrow the list of possible factors influencing identified trends (see previous chapters).

7.4 Statistical Considerations

A premise underlying most of what we present in this volume is that study designs must permit valid treatment of the data. For inventory studies, we must be able to characterize accurately the species or habitat variables of interest. For monitoring, we must know the effort needed to show a trend over time or to document a specified effect size in a parameter from time t_1 to t_2.

In this regard, the investigator should be well aware of concepts of statistical power, effect size, and sample size, and how they interact with Type I and Type II errors (see Chaps. 2 and 3 for detailed discussion of these concepts). Typically, investigators focus on the Type I error rate or alpha. However, in the case of sensitive, threatened, endangered, or rare species, consideration of Type II error rate is equally, if not more, relevant. A Type II error would be failure to detect a difference when it indeed occurred, an error that should be kept to a minimum. With threatened, endangered, or rare species, overreaction and concluding a negative impact or negative population trend when it is not occurring (Type I error) may have no deleterious effects on the species because additional protections would be invoked to guard against any negative management actions. In contrast, failing to conclude a significant decline in abundance when it is occurring (Type II error) may allow management to proceed without change even though some practices are deleterious to the species. The potential risk to the species could be substantial.

7.4.1 Effect Size and Power

Effect size and power go hand in hand when designing a monitoring study. Simply stated, *effect size* is a measure of the difference between two groups. This difference can be quantified a number of ways using various indices that measure the magnitude of a treatment effect. Steidl et al. (1997) regarded effect size as the absolute difference between two populations in a select parameter. Typically, investigators establish effect a priori and should be the minimum level that makes biological difference. For example, a population decline of 10% for a species of concern might be biologically relevant, so you would need a study with adequate sensitivity to show that decline when it occurs.

Three common measures of effects size are Cohen's d, Hedges' g, and Cohen's f^2 (Cohen 1988, 1992; Hedges and Olkin 1985). Cohen's d measures the effect size between two means, where d is defined as the difference between two means divided by the pooled standard deviation of those means. To interpret this index, Cohen (1992) suggested that $d = 0.2$ indicates a small, 0.5 a medium, and 0.8 a large effect size. Hedges' $\hat{g}$ incorporates sample size by both computing a denominator which looks at the sample sizes of the respective standard deviations and also makes an adjustment to the overall effect size based on this sample size. Cohen's f^2 is analagous to an F test for multiple correlation or multiple regression. With this index, f^2 of 0.02 is considered a small effect size, 0.15 is medium, and 0.35 is large (Cohen 1988).

Simply stated, *statisical power* is the probability that you will correctly reject a null hypothesis (Steidl et al. 1997). Recall from Chap. 2 that failure to reject correctly the null hypothesis is termed Type II error. As power increases, Type II error decreases. Power analysis can be done before (prospective) or after (retrospective) data are collected. Preferably, a researcher conducts prospective power analysis to determine sample sizes needed to have adequate power to detect the effect size of interest. The strength of this approach is that you can evaluate the interactions among power, effect size, and sample size to evaluate what is attainable. Stedl et al. (1997) provided an example of such analysis for two common birds species – hairy woodpecker (*Picoides villosus*) and chestnut-backed chickadee (*Poecile refuscens*) – found in Oregon forests (Fig. 7.3). They generated four curves for each species corresponding population increases of 50, 100, 150, and 200% across 3–9 replicate treatment pairs (treated and untreated). They applied the general rule that power >0.80 was acceptable. That was not achieved until there were at least eight replicates for the woodpecker and, even then, there was adequate power to detect only a 150% increase in the population (effect size). Had the population increased only 50%, a study with eight replicates would have been insufficient. By comparison, the more common chickadee required fewer replicates (seven) to detect a smaller increase (100%) in its population. Unfortunately, populations rarely show this level of response to habitat change caused by management unless, of course, the change

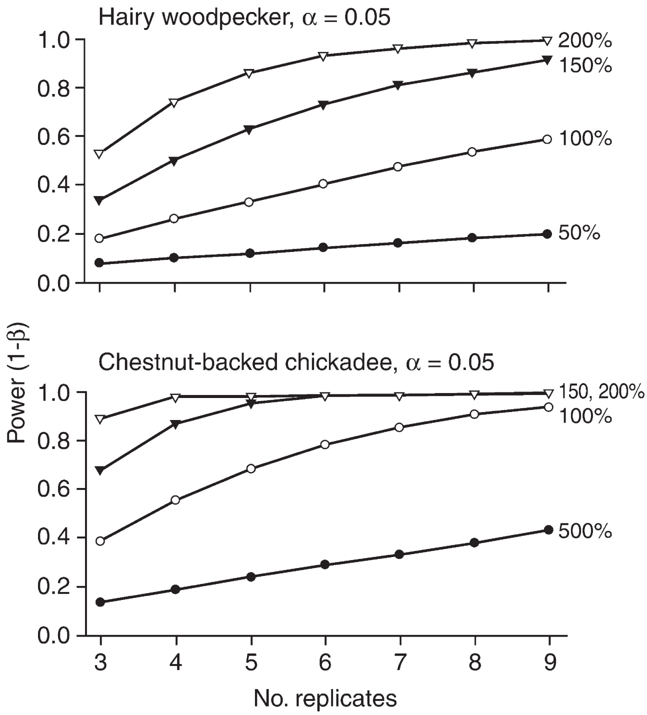


Fig. 7.3 Power analysis for hairy woodpecker and chestnut-backed chickadee to evaluate number of replicates needed to detect population increases of 50, 100, 150, and 200%. Reproduced from Steidl et al. (1997), with kind permission from The Wildlife Society

is severe. Thus, we are interested in more subtle population changes, which may go undetected given this experimental design.

It has become common practice to conduct retrospective power analysis in situations where results of a test are nonsignificant. Basically, such tests are used more as a diagnostic tool to evaluate what effects size might have been detected given a certain power, or vice versa, what power might be achieved given certain effect size or sample size. Steidl et al. (1997) caution about taking the results of retrospective power analyses too far. Effectively, their primary use is to evaluate hypothetical scenarios that may help to inform similar studies conducted sometime in the future. In some cases, they might also be used to test hypothesized effects sizes thought to be biologically relevant or to calculate confidence intervals around the observed effect size (Hayes and Steidel 1997; Thomas 1997).

7.4.2 Balancing Response Variables with Study Goals

Resources can be measured directly or indirectly. For example, if the study is to address the effects of a management activity on population trend of a species, then a direct approach would involve measuring the appropriate population attribute, such as abundance or density. However, populations of many species or other ecosystem attributes are difficult to sample because of their rarity or secretiveness that precludes obtaining enough samples even with a huge effort. In these cases, investigators often resort to indirect measures. These can include indices, indicator species, and stressors. Indirect measures should only be used if a clear and interpretable relationship has been established between the resource being studied and the surrogate measure (Landres et al. 1988).

Direct measures are variables that link clearly and directly to the question of interest. If they exist and are feasible to obtain, direct measures are preferred over indirect measures. Concerning inventories of species presence or faunal composition for community surveys, direct measures are used to assess presence or infer absence of the species of interest. Direct measures for populations can be measures of abundance, density, or of other population parameters of interest (e.g., survival, reproduction). Inventories or monitoring of habitats often focus on variables established as strong correlates of use by a species, or strong correlates to some measure of fitness.

Indirect measures are widely used for inventory and monitoring studies. *Indicator species* are used to index or represent specific environmental conditions or the population status of another ecologically similar species. They can be divided into two major categories: ecological indicators and management indicators. This concept was initially proposed by Clements (1920) to explain plant distributions based on specific environmental conditions, primarily soil and precipitation. Vertebrates are also tied to specific environmental conditions as this is the basis for describing species' habitats (Block and Brennan 1993). Many wildlife species, however, are vagile, and can adjust to variations in environmental conditions simply by moving or migrating. Thus, relationships between environmental conditions and most wildlife species may not be quite as strong as they are for many plants, and their predictive value of environmental conditions may be limited (Morrison 1986). If indicators are used, they should meet rigorous standards (Landres et al. 1988). These include (1) clearly stating what the indicator indicates about the environment or resource, (2) selection of indicators should be objective and quantitative, (3) all monitoring programs using indicators should be reviewed (a standard that should apply to all monitoring, not just indicators), and (4) indicators must be used at the appropriate spatial and temporal scales. Thus, use of indicators should not be simply a matter of convenience, but must be based on strong empirical evidence that supports their usage.

Stressors are another group of surrogate variables that can be measured in lieu of measuring a resource directly. *Stressors* are natural and anthropogenic events that affect resource distribution or abundance. Examples of stressors are loss of late seral forest due to fire; alterations of hydrologic regimes by water diversions; reduction, loss, or fragmentation of habitat; increased sediment loads following storms; or overharvesting of game or commercial species (Noon et al. 1999). Thus, rather than inventorying or monitoring a population or habitat directly, inferences are made based on some metric applied to the stressor. As with indicator species, the validity of stressors and their relationships to the variables of interest must be firmly established prior to their use.

As mentioned earlier, habitat is often monitored as a surrogate for monitoring an animal population directly. Costs of monitoring a population sufficiently to have acceptable statistical power to detect a trend can be rather high (Verner 1983). The estimated annual costs for conducting population monitoring for the Mexican spotted owl, for example, was about $1.2 to 1.5 million (USDI Fish and Wildlife Service 1995). When projected for 10–15 years, the costs could exceed $20 million for just this one subspecies! Consequently, macrohabitat or microhabitat (sensu Block and Brennan 1993) is often monitored to index population trend for a species. Unfortunately, information that documents clear and strong relationships between habitat components and population trend is lacking for most species. Thus, caution is advised when extrapolating habitat trends to populations.

If an indicator or stressor is monitored, then justification based on previous work must be provided to demonstrate that the variable studied is a good measure of ecosystem status or health. If the literature is unclear and cannot support the selection of a surrogate for study, then you should conduct a pilot study to test whether or not the variable you select measures what you intend it to, or abandon use of a surrogate and monitor the variable of interest directly. We recognize, however, that the use of surrogates such as indicators or stressors in monitoring, specifically their applicability and validity, is the subject of debate (Morrison 1986; Landres et al. 1988).

Another group of indirect measures or indices is *community metrics*. These indices provide little information about individual species, but provide quantitative values that are related to numbers, degree of association, diversity, and evenness of species (see Sect. 1.5.2). They can be applied to animals and their habitats.

Whereas *species richness* is a fairly straightforward concept in that it is simply a count of the number of species present, numerous algorithms are available for estimating degrees of association, diversity, and evenness (Hayek 1994; Pielou 1977). Measures of association include similarity coefficients, matching coefficients, and more traditional association coefficients (Hohn 1976; Hayek 1994). Similarity (e.g., Sorensen (1948) or Jaccard (1901)) and matching coefficients (e.g., Sokal and Michener (1958)) are not test statistics and are not based on a presumed sampling distribution. At best, they can be used in qualitative comparisons between different areas or comparisons of the same place but at different times. Traditional association coefficients include chi-square and contingency statistics, and can be evaluated against a probability distribution. Hayek (1994) reviewed various measures of species diversity and concluded that the concept is "variously and chaotically defined in the literature." Generally, measures include estimates of species richness and evenness. Evenness refers to the distribution of individuals among species. Differences among diversity algorithms often relate to how they weight diversity and evenness in calculation of their index value. A plethora of algorithms has been proposed; the two most often used are Shannon–Weiner and Simpson's indices. Often, it is difficult or impossible to ascribe a biological interpretation to diversity indices because nobody really knows what they measure. Thus, we recommend caution in using these indices as valid measures for inventory and monitoring studies.

7.5 Distinguishing Inventory from Monitoring

The answer to the question of what makes inventorying and monitoring different is basic. The difference between the two is largely a function of time; inventory measures the status of a resource at a point in time, whereas monitoring assesses change or trend over time in resource abundance or condition. Inventory and monitoring follow different processes to meet their goals, especially the series of feedback loops inherent to monitoring (see Fig. 7.1). Both require that you set goals, identify what to measure, and, in the case of management, state a value that when exceeded will result in a management decision. However, because inventory is to assess resource state whereas monitoring is to assess resource dynamics, they will often require different study designs. For example, the sampling design for a study to inventory Arizona to determine the distribution of spotted owls would be much different from a study to monitor population trend. Each study would be designed to estimate different parameters and would entail application of different statistical procedures, thus requiring different approaches to collect the relevant data. One basic principle common to both inventory and monitoring is that both should be scientifically valid. Thus, concepts discussed in Chaps. 1 and 2 regarding adequate sample sizes, randomization, replication, and general study rigor are critically important to any inventory or monitoring study. Failure to incorporate these considerations will result in misleading information, and potentially inappropriate conclusions and deleterious management decisions. To provide an example of how the goals of inventory and monitoring differ, consider the inventory and monitoring goals presented below in the Mexican spotted owl recovery plan (Box 7.3).

As we can see from this example, goals of inventory and monitoring can be quite different. Inventories are often done with the goal of assessing the status of a species

Box 7.3 Inventory and Monitoring Goals for the Mexican Spotted Owl

Inventories are used in two basic ways for the Mexican spotted owl. One is part of project planning and the other is to increase basic knowledge about owl distribution. The Mexican Spotted Owl Recovery Plan requires that all areas with any chance of occupancy by owls be inventoried prior to initiating any habitat-altering activity. The reason why is to determine if owls are using the area and if so, to modify the management activity if necessary to minimize impact to the bird. Thus the goal is straightforward: to determine occupancy (or infer nonoccupancy) of owls to help guide the types and severity of habitat-modifying management that might impact the owl. The second goal of inventory is to understand the distribution of the owl better. Most inventories for owls have been conducted in areas where management (typically timber harvest and prescribed fire) is planned as part of the process described above. These areas represent only a subset of the lands that the owl inhabits. Thus, to increase knowledge of owl distribution and population size, the plan calls for inventories in "holes in the distribution" or in potential habitats where no records of owls exist.

The recovery plan also requires both population and habitat monitoring. The reasons for monitoring are multifaceted. First, the owl was listed as threatened based on loss of habitat and the concern that habitat would continue to be lost given current management practices. Although not explicitly stated in listing documents, it was assumed that there was a population decline concomitant with habitat decline. Thus, a very basic reason to monitor is to evaluate whether or not these trends are indeed occurring and if they are correlated. A second objective for monitoring is to evaluate whether or not implementation of management recommendations in the recovery plan were accomplishing their intended goal, namely recovering the subspecies. This would entail (1) implementation monitoring to determine if management activities were done as designed and (2) effectiveness monitoring to evaluate whether following management recommendations is sustaining owl populations and habitats. This would be tested by examining both habitat and population trends to ensure that owl populations persist into the future. A third objective of monitoring, validation, would provide measurable, quantitative benchmarks that when met would allow the bird to be removed from the list of threatened species (i.e., delisted).

on an area planned for management activities. By status, we mean presence/absence, abundance, density, or distribution. With threatened or endangered species such as the owl, inventories are often used to permit modify, or curtail habitat-altering activities. Other goals of inventory might be to document species presence within a management unit to aid in resource planning or conservation (Hunter 1991; Scott et al. 1993), or evaluate habitat suitability of an area for a given species to determine if it has the potential for occupancy (Verner et al. 1986), or the goal might be simply for increasing scientific knowledge by inventorying new areas and documenting species that were previously undescribed. Certainly faunal inventories by early naturalists such as Wallace, Darwin, Audubon, Xantu, and others provided key baseline information for addressing many interesting and complicated ecological questions.

7.6 Selection of a Design

Monitoring and inventory projects require an adequate sampling design to ensure unbiased and precise measures of the resource(s) of interest. To do so requires a priori knowledge of the resource under study, including its behavior, distribution, biology, and abundance patterns (Thompson et al. 1998). It is also necessary to understand the statistical properties of the population from which a sample is to be taken. Once these basic properties are known, the investigator must determine the appropriate sampling methodology to meet inventory or monitoring objectives, given available funds and personnel.

A sampling design for an inventory or monitoring study consists of four interrelated components (see Morrison et al. 1998 for detailed discussion). An investigator must first decide *what* it is that he or she wants to measure, *where* to sample (the sampling universe), *when* to study (timing and length of time), and, finally, *how* to collect data. We discuss these components below.

7.6.1 Identifying the Resources to Be Measured

Selecting the variables to measure should be supported by previous knowledge or established information. Hopefully, the investigator possesses a certain expertise in the species or system being inventoried or monitored and can draw on that knowledge to select variables or specific resources to study. Often this is not the case and the investigator will need to do some background work, such as a literature review, consulting with established experts, or using results of similar studies to establish the basis for measuring a given variable(s).

When monitoring populations, it is important to determine the parameters most sensitive to change and focus on those. Typically, investigators focus on population abundance or density of breeding individuals. This might be misleading, however, if there exists a large number of nonterritorial animals (e.g., nonbreeding individuals) not easily sampled using traditional methods (e.g., auditory surveys). In this case, it is possible that you can have high mortality of territorial animals that are immediately replaced by surplus, floating individuals. The population of territorial animals may appear stable while the over all population is declining. Information on the age of initial territorial occupancy or the age class distribution might be needed to more fully understand the status of the population. Again, the point here is that you must understand the biology and population dynamics of the species being monitored to make better decisions on exactly what to monitor.

7.6.2 Selection of Sampling Areas

Once the study objective is established, the scale of resolution chosen by ecologists is perhaps the most important decision in inventory and monitoring because it predetermines procedures, observations, and results (Green 1979; Hurlbert 1984). A major step in designing an inventory or monitoring study is to establish clearly the target population and the sampling frame. Defining the target population essentially defines the area to be sampled. For example, if an area was to be inventoried to determine the presence of Mexican spotted owls on a national forest, sampling areas should include general areas that the owl uses (mature conifer forests and slickrock canyons) but not include areas that the owl presumably would not use (grasslands, desert scrub) based on previous studies. This first step establishes the sampling universe from which samples can be drawn and the extent to which inferences can be extrapolated. Thus, the results of these owl surveys apply only to the particular national forest and not to all national forests within the geographic range of the owl.

Although this seems rather straightforward, the mobility of wildlife can muddle the inferences drawn from the established area. Consider, for example, the case of the golden eagle example presented in Chap. 6. A somewhat arbitrary decision was made to define the "population" potentially affected by wind turbine mortality as the birds found within a fixed radius of 30 km of the wind farm. The basis for this decision included information on habitat use patterns, range sizes, movement patterns, and logistics of sampling a large area. The primary assumption is that birds within this radius have the greatest potential of encountering wind turbines and are the birds most likely to be affected. Not measured, however, were cascading effects that may impact eagles beyond the 30 km radius, because eagles found within this radius were not a distinct population. Thus, factors that influenced birds within this arbitrary boundary may have also affected those outside of the boundary. The point here is that even though considerable thought went into the decision of defining the sampling universe for this study, the results of the monitoring efforts may be open to question because mortality of birds within the 30 km may be affecting the larger population, including birds found beyond the 30 km radius.

7.6.3 Study Duration

A key aspect in the design of any study is to identify when to collect data. There are two parts to this aspect: the timing of data collection and the length of time over which data should be taken. The choice of timing and length of study is 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. Overarching these considerations is the need to sample adequately for precise estimates of the parameter of interest.

Timing refers to when to collect data and it depends on numerous considerations. Obviously, studies of breeding animals should be conducted during the breeding season, studies of migrating animals during the migration 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. Male passerine birds, for example, are generally more conspicuous during the early part of the breeding 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 vary 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 substantially depending on when data are collected. Once the decision is made as to when to collect data, it is crucial that data are collected during the same time in the phenology of the species during subsequent years to control for some of the within season variation.

Objectives of a study also dictate when data should be collected. If the study is an inventory to determine the presence of species breeding in an area, sampling should occur throughout the breeding season to account for asynchrony in breeding cycles and heterogeneity in detectabilities among species. Sampling spread over the course of the season would give a greater chance of recording most of the species using the area. If a monitoring study is being conducted to evaluate population trend of a species based on a demographic model, sampling should be done at the appropriate time to ensure 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 unbiased estimates for the respective measures (USDI Fish and Wildlife Service 1995).

Length of the study refers to how long a study must be done to estimate the parameter of interest. It depends on a number of factors including study objectives, field methodology, ecosystem processes, biology of the species, budget, and feasibility. A primary consideration for monitoring and inventory studies should be temporal qualities of the ecological process or state being measured (e.g., population cycles, successional patterns). Temporal qualities include frequency, magnitude, and regularity, which are influenced by both biotic and abiotic factors acting both stochastically and deterministically (Franklin 1989). Further, animals are subjected to various environmental influences during their lifetimes. 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 environmental conditions. Consider two hypothetical wildlife populations that exhibit cyclic behaviors, one that cycles on average ten times per 20 years, and the other exhibiting a complete cycle just once every 20 year (Fig. 7.4). 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 10-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. Likely, a longer timeframe would be needed to monitor the population of species with the lower frequency cycles.

However, considering only the frequency of population cycles may be inadequate as the amplitude or magnitude of population shifts may also influence the length of a study to sort out effects within year variation from between year variation. Consider two populations that exhibit ten cycles in 20 years, but now the magnitude of the change for one is twice that of the other (see Fig. 7.4). Sampling the population exhibiting greater variation would require a longer period to detect a population trend or effect size should one indeed occur.

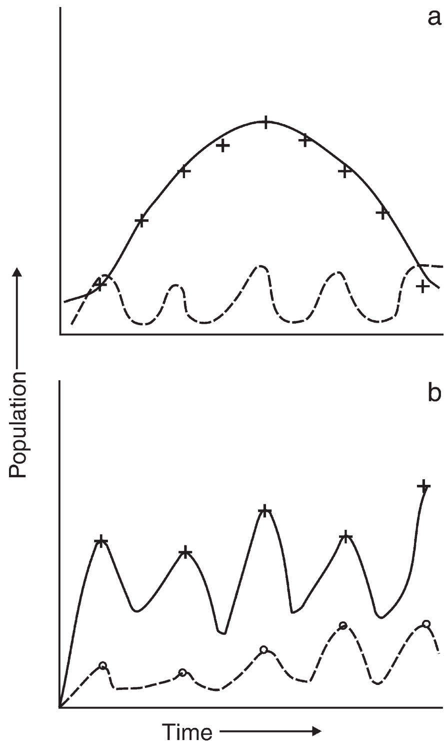


Fig. 7.4 Theoretical population cycles comparing (**a**) species with high (*dashed line*) and low (*solid line*) frequency cycles, and (**b**) species of low-amplitude (*dashed line*) and high-amplitude (*solid line*) population cycles

Typically, biologists do not have a wealth of information to draw upon prior to deciding the duration of study. In these situations, they must draw upon the best available information, and perhaps structure the study to adjust the length, as data are collected. In the case of Mexican spotted owls, for example, a wealth of information was available from both the literature and unpublished reports for developing population monitoring. Based on this information, a period of 10–15 years to delist the owl and 5 year postdelisting was chosen as the period for population monitoring (USDI Fish and Wildlife Service 1995). The basis for 10 years until delisting was that this would be ample time for 70% of the adult population to turn over, and that 10 years would allow the owl population to be subjected to variations in environmental factors that might influence its population. The additional 5 year of postdelisting monitoring would provide additional time to reaffirm the trend measured after 10 years. If the population trend is negative after 10 years of monitoring, the birds would not be delisted and monitoring should continue. The point here is that the length of study must have a biological basis. Failure to conduct a study for an adequate length of time might lead to erroneous conclusions of trends or effects.

In reality, however, costs, personnel, logistical constraints, and shifting priorities add a great deal of difficulty to first committing to and then continuing monitoring over the long term (Morrison and Marcot 1995; Moir and Block 2001; White et al. 1999). Consequently, innovative approaches are required to attempt to achieve unbiased results from suboptimal monitoring designs. The compromise typically made is to find alternatives to long-term studies. These approaches are discussed in Sect. 7.7.

7.6.4 Monitoring Occupancy vs. Abundance

Gathering abundance and demographic data can be costly, entail extensive field sampling, and require highly skilled personnel. Cost is higher largely because of the number of the samples needed for precise point estimates of the relevant parameters. Often, these costs are beyond the budget of many funding agencies, thus requiring more cost-effective approaches. Even if cost is not the primary constraint, the feasibility of obtaining enough samples to estimate abundance for rare species may be limiting.

New advances for estimating detection probabilities and using this information to adjust occupancy rates are largely responsible for the renewed interest in occupancy monitoring. In addition, one can model covariates as they relate to occupancy rates. These models can serve as descriptive tools to explain variation in occupancy rates. Although occupancy estimation is not new and is the basis for numerous indices, it has gone through a recent resurgence as a viable monitoring approach, especially for rare and elusive species (MacKenzie et al. 2004). Generally, occupancy monitoring is cost-efficient, can employ various indirect signs of occupancy, and does not always require as highly skilled personnel. Occupancy can be useful for a number of different studies including those investigating metapopulation structures, changes in geographic distribution, patch use, and species diversity patterns. However, occupancy does not convey the same information as abundance or density estimates. Hopefully, occupancy will index abundance but those relationships are likely species, time, and location specific.

Ganey et al. (2004) evaluated the feasibility of implementing the mark–recapture design for monitoring Mexican spotted owls presented in the recovery plan for this subspecies. Their evaluation included logistical aspects of implementing the study and statistical considerations of the sampling effort needed to show population decline. They concluded that the expense and personnel needs to conduct mark-recapture monitoring were daunting. More troublesome, however, was that random variation in the population was so great that it was difficult to ascribe a 20% decline in the population to anything more than chance. Given high costs and logistical hurdles of implementing this approach, the Mexican Spotted Owl Recovery Team revised their approach to population monitoring by focusing on occupancy.

7.6.5 Sampling Strategies

We focus extensively on sampling design and applications in Chaps. 4 and 5, so we will not repeat them here. Clearly, the design and execution of monitoring and inventory studies depends on the same basic considerations as other studies.

In some cases, the sampling universe is small enough to permit a complete enumeration (e.g., census) of the entire area. More typically, the entire sampling universe cannot be surveyed, thus you need to establish sample plots. Primary considerations with establishing plots are (1) their size and shape, (2) the number needed, and (3) how to place them within the sampling universe (See Chap 2). Size and shape of plots depend on numerous factors, such as the method used to collect data, biological edge effects, distribution of the species under study, biology of the species, and logistics of collecting the data. Thompson et al. (1998, pp. 44–48) summarize the primary considerations and tradeoffs in choosing a plot design. For example, long and narrow plots may allow for more precise estimates, but square plots will have less edge effect. They concluded that no single design is optimal for all situations, and they suggested trying several in a pilot study. Plot size depends largely on the biology and distribution of the species under study. Larger plot sizes are needed for species with larger home ranges and for species with clumped distributions. For example, larger plots would be needed to survey the spotted owl (home range size about 800 ha) than would be needed for the dark-eyed junco (*Junco hyemalis*) (home range about 1 ha). Further, larger plots are needed for species with clumped distributions, such as quail, than might be needed for species with more even distributions, such as the plain titmouse (Fig. 7.5). Note that the species in Fig. 7.5b will not be sampled adequately using the same plot size as used for the species in Fig. 7.5a. Larger-sized plots will be needed to sample the species with the clumped distribution (Fig. 7.5b).

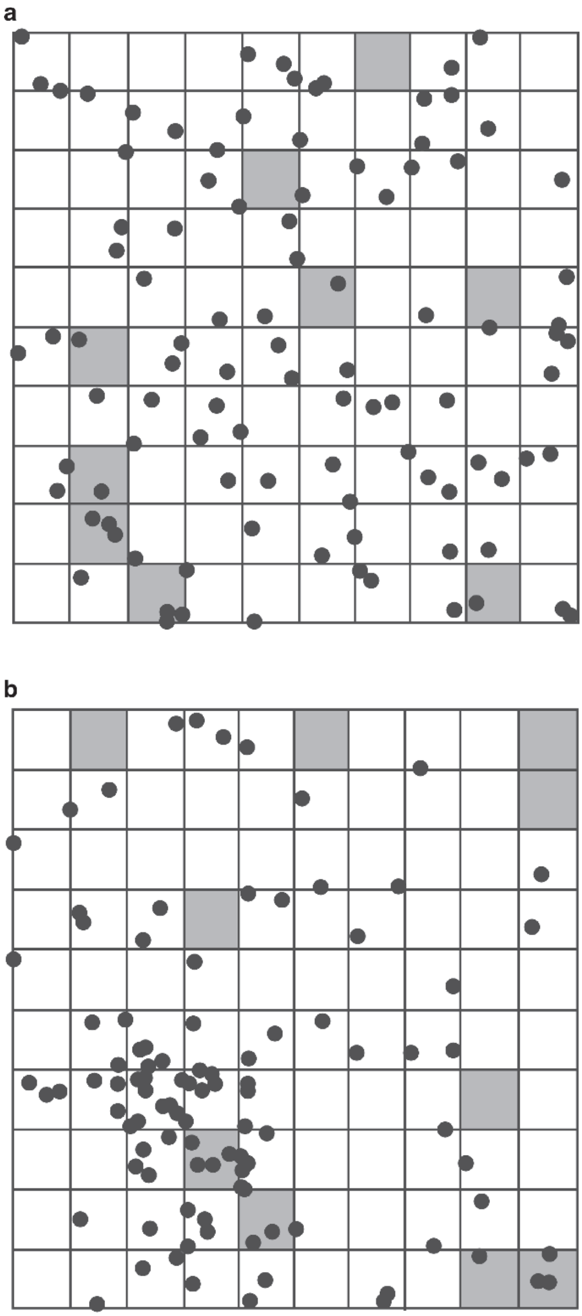


Fig. 7.5 Simple random samples of ten plots (gray plots) from sampling frames containing (**a**) a random distribution of individuals and (**b**) a clumped distribution of individuals. Reproduced from Thompson et al. (1998), with kind permission from Elsevier

The number of sample plots and placement of plots within the study area depend on a number of sampling considerations, including sampling variances and species distributions and abundances. Sample size should be defined by the number of plots to provide precise estimates of the parameter of interest. Allocation of sample plots should try to minimize sampling variances and can be done a number of ways. Survey sampling textbooks are a good source of discussion of the theoretical and practical considerations. Basic sampling designs include simple random, systematic random, stratified random, – cluster sampling, two-stage cluster sampling, and ratio estimators (Thompson 2002; Cochran 1977). Chapters 4 and 5 presented some of these basic sampling designs with examples of how they are typically applied.

7.6.6 Use of Indices

Historically, wildlife biologists have made heavy use of indices as surrogates for measuring populations. These can include raw counts, auditory counts, track surveys, pellets counts, browse sign, capture per unit of effort, and hunter success. Indices are often used to address inventory and monitoring questions (see Sect. 7.1). Implicit to indices is that they provide an unbiased estimate of the relative abundance of the species under study. This assumption, however, rests heavily on the assumption that capture probabilities are homogeneous across time, places, and observers (Anderson 2001).

Although indices are widely used, they are not widely accepted (Anderson 2001; Engeman 2003). Primary criticisms are that they fail to account for heterogeneous detection probabilities (Anderson 2001), employ convenience samples which are not probabilistic samples (Anderson 2001, 2003) typically lack measures of precision (Rosenstock et al. 2002), and when provided they have large confidence intervals (Sharp et al. 2001).

However, few investigators have enough resources to feed the data hungry analyses that permit raw counts to be adjusted by detection probabilities (Engeman 2003), thereby relegating investigators to using indices. McKelvey and Pearson (2001) noted that 98% of the small mammal studies published in a 5-year period had too few data for valid mark–recapture estimation. Verner and Ritter (1985) found that simple counts of birds were highly correlated with adjusted counts, but simple counts were possible for all species whereas adjusted counts were possible only for common species with enough detection.

Index methods are efficient and their use will likely continue (Engeman 2003). Engeman (2003) notes that the issue with indices is not so much the method as it is with selecting and executing an appropriate study design and conducting data analysis to meet the study objective. Methods exist to calibrate indices by using ratio estimation techniques (see Chap 5; Eberhardt and Simmons 1987), double sampling techniques (Bart et al. 2004), or detection probabilities (White 2005). These calibration or correction tools may reduce bias associated with indices and render indices more acceptable as inventory and monitoring tools.

7.7 Alternatives to Long-Term Studies

Four phenomena necessitate long-term studies (1) slow processes, such as forest succession or some vertebrate population cycles, (2) rare events, such as fire, floods, diseases, (3) subtle processes where short-term variation exceeds the long-term trend, and (4) complex phenomena, such as intricate ecological relationships (Strayer et al. 1986). Unfortunately, needs for timely answers, costs, changing priorities, and logistical considerations may preclude long-term studies. In such cases, alternative approaches are sought to address inventory or monitoring objectives. Various alternatives to long-term sampling have been proposed, such as retrospective sampling (Davis 1989), substitution of space for time (Pickett 1989), the use of systems with fast dynamics as analogies for those with slow dynamics (Strayer et al. 1986), modeling (Shugart 1989), and genetic approaches (Schwartz et al. 2007).

7.7.1 Retrospective Studies

Retrospective studies have been used to address many of the same questions as long-term studies. A key use of retrospective studies is to provide baseline data for comparison with modern observations. Further, they can characterize slow processes and disturbance regimes, and how they may have influenced selected ecosystem attributes (Swetnam and Bettancourt 1998). Perhaps the greatest value of retrospective studies is for characterizing changes to vegetation and wildlife habitats over time. Dendrochronological studies provide information on frequencies and severities of historical disturbance events (Swetnam 1990) (Fig. 7.6). This information can be used to reconstruct ranges of variation in vegetation structure and composition at various spatial scales. These studies can also be used to infer short- and long-term effects of various management practices on habitats, as well as effects of disruptions of disturbance regimes on habitats.

Other potential tools for retrospective studies include databases from long-term ecological research sites, forest inventory databases, pollen studies, and sediment cores. They are also used in epidemiological and epizootiological studies. With any of these studies, one must be aware of the underlying assumptions and limitations of the methodology. For example, dendrochronological methods often fail to account for the small trees because they are consumed by fire and not sampled. This limitation may result in a biased estimate of forest structure and misleading inferences about historical conditions. If the investigator understands this idiosyncrasy, then he or she can consider this during evaluation.

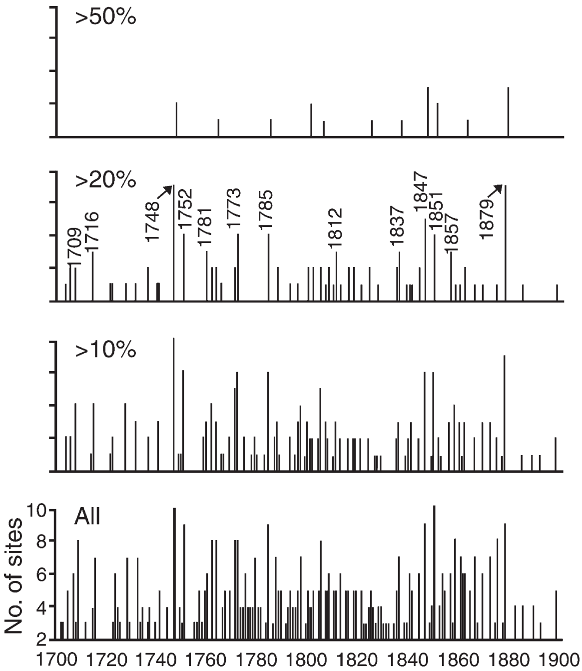


Fig. 7.6 Fire-area index computed as the number of sites recording fires per year for the period 1700–1900. Fires recorded by any tree within the sites are shown on the bottom plot, whereas fires recorded by 10, 20, or 50% of the trees are shown above (from Swetnam 1990)

7.7.2 Substitutions of Space for Time

Substituting space for time is achieved by finding samples that represent the range of variation for the variable(s) of interest in order to infer long-term trends (Pickett 1989, Morrison 1992). The assumption is that local areas are subjected to different environments and different disturbance histories that result in different conditions across the landscape. Thus, rather than following few samples over a protracted period to understand effects of slow processes, random events, or systems with high variances, more areas are sampled hoping that they represent conditions that might exist during different phases of these processes. For example, if you wanted to understand the long-term effects of forest clear-cutting on wildlife, a logical approach would be to locate a series of sites representing a *chronosequence* of conditions rather than waiting for a recent clear-cut to go through succession. By chronosequence, we mean areas that were clear-cut at various times in the past (e.g., 5, 10, 20, 30, 50, 75, and 100 years ago). By sampling enough areas representative of vegetation structure and composition at different times following clear-cuts you could draw inferences as to possible short- and long-term effects on wildlife. To provide valid results using this approach requires that many sites with somewhat similar histories and characteristics be used (Morrison 1992). If substantial sources of variation between sampling units cannot be accounted for, then substituting space for time will fail (Pickett 1989). Even if these sources can be accounted for, space-for-time substitutions may fail to take into account mesoscale events (Swetnam and Bettancourt 1998) that affect large regions and tend to mitigate or swamp local environmental conditions. Pickett (1989) cautioned that studies that rely on spatial rather than temporal sampling are best suited for providing qualitative trends or generating hypotheses rather than for providing rigorous quantitative results. Even so, spatially dispersed studies are preferred for inventory studies.

Clearly, an empirical basis is needed to support the use of space-for-time substitutions in monitoring studies. By this, we mean that you should conduct a baseline study to evaluate whether such an approach would provide unbiased estimates of the variable(s) under study. This baseline study would require comparisons of an existing long-term data set collected as part of another study with a data set collected from multiple locations over a time. If no significant differences are observed in estimates of the variables of interest, then space-for-time substitutions may be justified. If a difference is observed, then one can explore methods to calibrate results of one approach with the other. If the differences cannot be rectified by calibration, you should reconsider the use of space-for-time substitutions in your study design.

7.7.3 Substitutions of Fast for Slow Dynamics

Applying the results of a simple system with rapid generation times or accelerated rates of succession can provide insights into how systems with inherently slower processes might behave (Morrison 1992). For example, applying results of laboratory studies on rodents might provide some insight on population dynamics of larger wild mammals. Obviously, extending results of captive animals to wild populations has obvious drawbacks, as does applying results from *r*-selected species such as rodents to larger *K*-selected species such as carnivores. At best, such substitutions might provide a basis for development of hypotheses or theoretical constructs that can be subjected to empirical tests. These tests should be designed to show the correspondence between the surrogate measure (e.g., that with fast dynamics) and the variable that exhibits slow dynamics. If the relationship is strong, then it might be acceptable to use behavior of the surrogate measure as an index for the variable of interest.

7.7.4 Modeling

Use of models has gained wide application in studies of wildlife habitats (Verner et al. 1986) and populations (McCullough and Barrett 1992). Models can be conceptual or empirical (Shugart 1989). Conceptual models are generally used to structure a scientific endeavor. As an example, one might ask, “How is the population of spotted owls influenced by various environmental factors?" A conceptual model might consist of an envirogram that depicts how owls are linked to various ecological components and processes (Verner et al. 1992). This conceptual model can provide the basis for conducting specific studies to understand the effects of one or more factors on owl population trends (Fig. 7.7). One can argue, in fact, that all scientific studies are based on conceptual models of various levels of sophistication regardless of whether the researcher is explicitly aware of this fact. The example

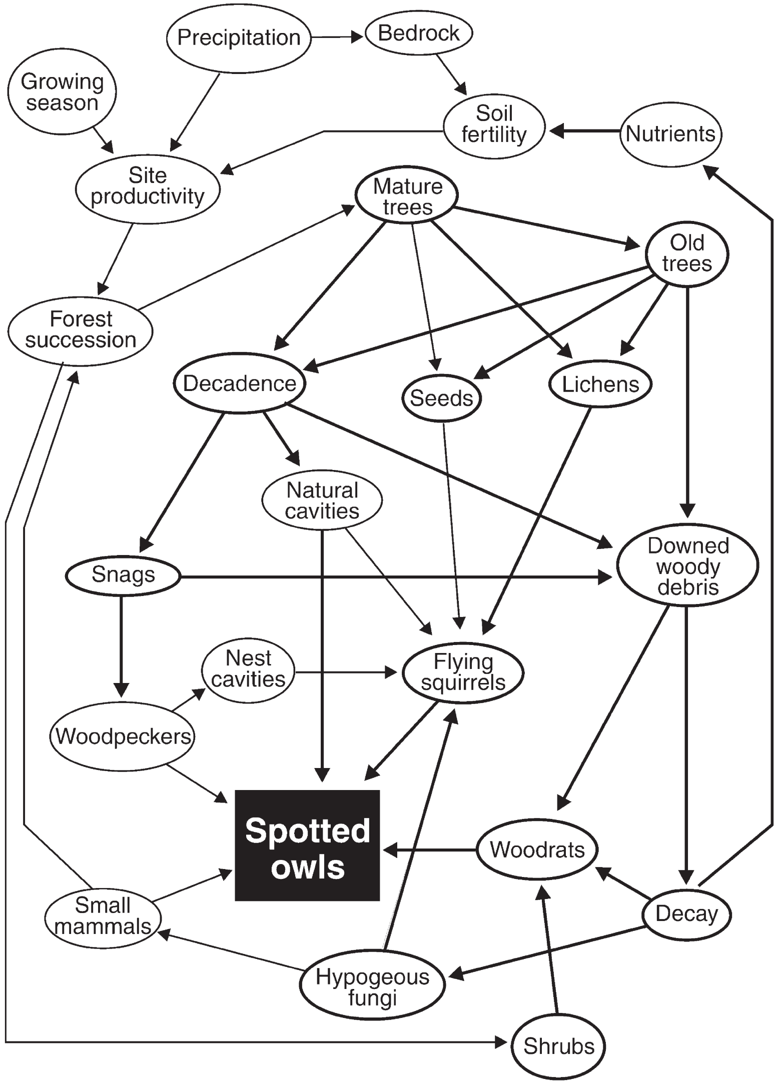


Fig. 7.7 Simplified schematic representation of some important ecological linkages associated with California spotted owls (from Verner et al. 1992)

provided in Fig. 7.7 is perhaps more detailed than most conceptual models, but it does show how a system can be characterized as interactions among many tractable and researchable components.

Quantitative forecasts from predictive models are used to provide wildlife managers with realizations of ecological processes. When structuring any modeling exercise to address population dynamics questions, an initial decision must be made concerning the proposed model's purpose (McCallum 2000). Empirical models are quantitative predictions of how natural systems behave. Models for examining population dynamics exist on a continuum from empirical models used to make predictions to abstract models that attempt to provide general insights (Holling 1966; May 1974; McCallum 2000). Predictive models require a larger number of parameters than abstract models, increasing their predictive ability for the system of interest, but reducing the generality of the model and thus its ability to expand results to other systems.

Ecological modeling in wildlife studies encompasses a broad range of topics, but most often relates to two topics, demographic (parameter estimation) and population modeling. Demographic modeling is directed toward developing a model which best explains the behavior and characteristics of empirical data, and then using that model to predict how that or similar systems will behave in the future (Burnham and Anderson 2002). The use and sophistication of demographic modeling has increased along with increases in personal computing power (White and Nichols 1992) and development of statistical programs specifically for ecological data (Sec 2.7.2).

Population modeling is directed towards development of predictive models, based on the aforementioned demographic parameters, which we use to forecast the response of wildlife populations to perturbations. Population models come in many forms: population viability analysis, matrix population models, individual based models, and so on (Caswell 2001; Boyce 1992; DeAngelis and Kross 1992) each structured with the intent of describing and predicting population dynamics over time and space (Lande et al. 2003). To be realistic, population models must include simultaneous interactions between deterministic and stochastic processes (Lande et al. 2003), which lends uncertainty to predictions of population trajectories. Because the fundamental unit in animal ecology is the individual (Dunham and Beaupre 1998), many population models incorporate individual variability (e.g., stochasticity in estimates of demographic parameters).

7.7.5 Genetics

Genetic techniques represent a new and burgeoning field providing novel approaches to monitoring. Schwartz et al. (2007) provide an insightful overview of these techniques. They separated genetic monitoring into two categories (1) markers used for traditional population monitoring and (2) those used to monitor population genetics.

Most genetic materials are obtained through noninvasive samples – hair, scat, feathers, and the like – thus, obviating the need to capture or even observe the species under study. Individual animals are identified using genetic markers, thus permitting estimates of abundance and vital rates. For rare species, abundance indices are possible, which are adjusted subsequently for small population size or detection probability (White 2005). For more abundant species, capture–recapture analyses can be applied (see Chap. 4). These samples can also be used to estimate survival and turnover rates. Survival rates are often difficult to estimate using traditional mark–capture techniques, especially when detection or capture rates vary with time. For example, male northern goshawks are detected more easily using traditional techniques during years when they breed than in years when they do not (Reynolds and Joy 2006). Survival estimates based on years when the goshawks do not breed may be underestimates given lower capture probabilities. This bias might be reduced using molted feathers and genetic markers to estimate survival.

Genetics can also be used to identify species, the presence of hybrids, and the prevalence of disease or invasive species. For example, genetics has been used to identify the historical geographical range of fisher (*Martes pennanti*) (Aubry et al. 2004; Schwartz 2007), the presence of Canada lynx (*Lynx canandensis*) (McKelvey et al. 2006), hybridization between bobcats (*Lynx rufus*) and lynx (Schwartz et al. 2004), and hybridization between northern spotted owls (*Strix occidentalis caurina*) and barred owls (*Strix varia*) (Haig et al. 2004).

Genetics can also be used to estimate effective population size and changes in allele frequencies. This information is critical to understanding patterns of gene flow and effects of habitat fragmentation on populations. The insight provided by these approaches and others has tremendous implications for present and future management of these species. Ultimately, the success of that management can only be assessed with continued monitoring in the mode of adaptive management.

7.8 Adaptive Management

The concept of *adaptive management* rests largely on monitoring the effects of implementing land management activities on key resources, and then using monitoring results as a basis for modifying those activities when warranted (Walters 1986; Moir and Block 2001). It is an iterative process whereby management practices are initiated and effects are monitored and evaluated at regular intervals. Effectively, land management activities are implemented incrementally and desired outcomes are evaluated at each step. If outcomes are consistent with or exceed predictions, the project continues as designed. If outcomes deviate negatively from predictions, then management can proceed in one of three directions: continue, terminate, or change.

This general scenario can be characterized by a seven-step process that includes a series of feedback loops that depend largely on monitoring (Moir and Block 2001) (Fig. 7.8). The primary feedback loops in Fig. 7.8 are between steps 5–6–7, 2–7, and 7–1. The 5–6–7 feedback loop is the shortest and perhaps the fastest. It implies that management prescriptions are essentially working and need only slight, if any, adjustments. Perhaps the primary obstacle in this loop is the lag time

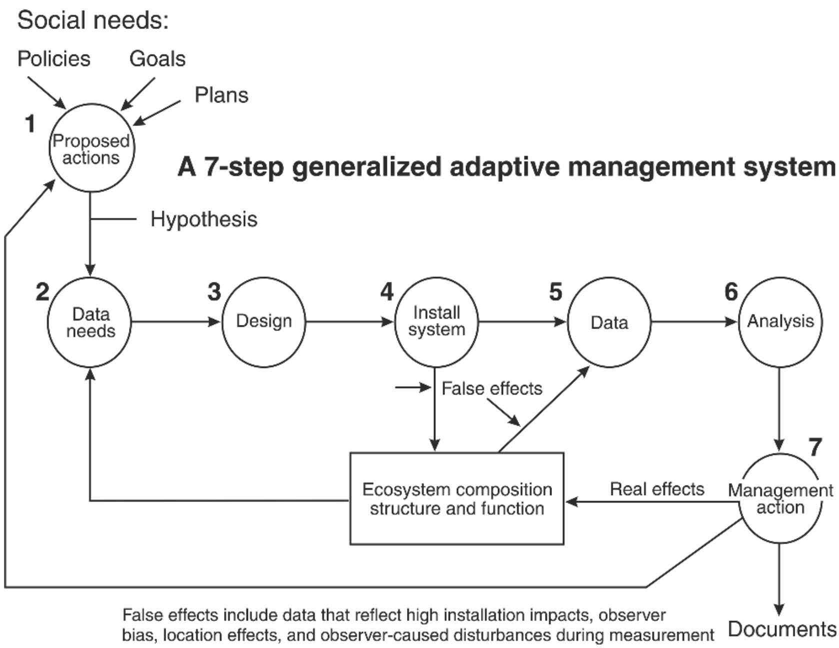


Fig. 7.8 A seven-step generalized adaptive management system illustrating the series of steps and feedback loops. Reproduced from Moir and Block (2001), with kind permission from Oxford University Press

between project implementation and completion of monitoring. Because this timeframe can be prolonged, numerous factors may complicate the ability or willingness of the organization to complete monitoring (Morrison and Marcot 1995). Consequently, the loop is often severed and feedback is never provided. The second feedback loop, 2–7, indicates that monitoring missed the mark. By this, we mean the monitoring study was poorly designed, the wrong variables were measured, or monitoring was poorly executed. Regardless of exactly what went wrong, monitoring failed to provide reliable information to permit informed conclusions on the efficacies of past management, or in making decisions for future management direction. The 7–1 feedback loop is the one typically associated with adaptive management; it is when a decision must be made regarding the course of future management and monitoring activities. If monitoring was done correctly, then informed decisions can be made for future management direction. If monitoring was not conducted or was done poorly, then another opportunity was lost to provide a scientific basis for resource management. Unfortunately, the latter is more the rule than the exception (White et al. 1999; Moir and Block 2001).

If adaptive management is to be the paradigm followed in the future as espoused by most contemporary resource management agencies, it is only possible by conducting credible monitoring. Inventory and monitoring provide critical information on resource status and trends needed to make informed management decisions. Failure to incorporate these studies will doom adaptive management to failure.

7.8.1 Thresholds and Trigger Points

In designing inventory or monitoring studies for management applications, you must establish some benchmark that signals a need for subsequent actions. Benchmarks can signify success as well as failure. Thus, actions taken in response to reaching a benchmark may range from a cessation of activities to engaging in the next step in a management plan. Regardless of exactly how the benchmark is used, it provides a measurable criterion for management actions.

Benchmarks also play a role in study design, particularly in determining sampling intensity. In monitoring, effect size establishes the amount of change that you want to detect if it indeed occurs. Thus, effect size is closely interrelated with statistical power, sample size, and Type I error, as all three of these will define the minimal size of an effect that can be detected. Figure 7.9 shows the tradeoff between power and effect size to detect a population trend. Note that as effect size increases, statistical power increases. This essentially means that it is easier to statistically show effect when the change is big than it is to statistically show effect when change is small. The tradeoff is one that must be carefully evaluated and decided upon at the onset of designing a study.

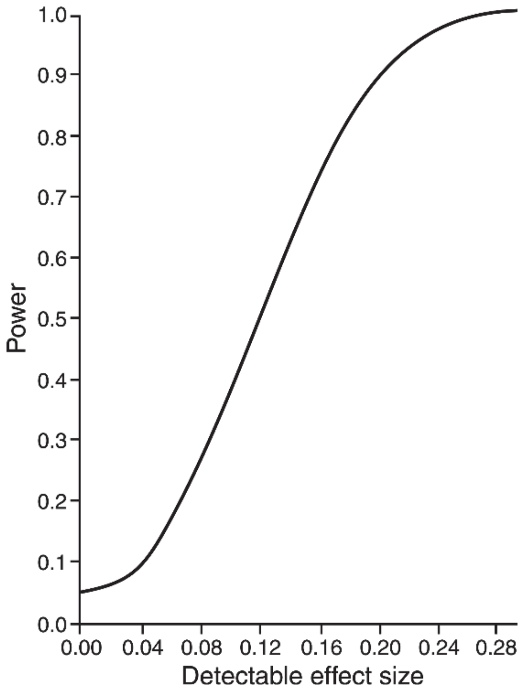


Fig. 7.9 Hypothetical curve of the statistical power needed to detect a population trend in a population (from USDI Fish and Wildlife Service 1995)

Thresholds and trigger points represent predetermined levels that when exceeded will lead to an action or response. The action or response could be termination or modification of a particular activity. For example, consider a prescribed fire project to be conducted in spotted owl habitat. The plan calls for treating 5,000 ha, spread across six separate fires. The fire plan calls for no special protection to trees, but predicts that no trees >60 cm dbh will be killed because of the fire. Two statistical tests are developed, one to test the spatial extent of loss and the other to test for the absolute magnitude of the percentage of large trees lost. A monitoring plan is developed following a standard protocol (see Box 7.4). Following postfire monitoring, it was determined that too many large trees were lost exceeding prefire predictions, resulting in feedback into the system. In this case, the loss of any trees signified a threshold that was exceeded. If actions were then developed and initiated to mitigate the loss of trees, then the threshold becomes a trigger point. In this case, future prescription may require removing litter and flammable debris from the base of large trees to minimize the probability of tree mortality.

Box 7.4 Mexican Spotted Owl Microhabitat Implementation Monitoring

The Recovery Plan for the Mexican Spotted Owl (USDI Fish and Wildlife Service 1995) allows treatments in forested landscapes. However, the extent and type of treatments are limited in mixed-conifer, part of the pine-oak, and riparian vegetation types. The Plan also calls for monitoring of macrohabitat and microhabitat as part of the recovery process. Delisting criterion 2 in the Recovery Plan specifically requires habitat monitoring to demonstrate that habitat across the range is stable or increasing (USDI Fish and Wildlife Service 1995, p. 77).

This protocol partially addresses the microhabitat monitoring requirement (referred to as implementation monitoring) by assessing the retention of Key Habitat Components (described below) in protected and restricted forest types following habitat-altering activities.

The purpose of monitoring is to index the change of key components in owl habitat in treated areas. Losses are predicted at two scales. One is the total percentage change to the component across the entire project area (project-level monitoring). Analysis of total percentage change will provide information on the magnitude of change across the project. The second scale is the percentage loss of the component on a plot-level basis. Analysis on plot-level basis will provide spatial information on treatment effects.

This protocol applies to silviculture, thinning, management-ignited fire, and other activities directed at modifying forests and woodlands (excluding prescribed natural fire) in protected and restricted areas as defined in the Recovery Plan (USDI Fish and Wildlife Service 1995, pp. 84–95).

(continued)

Box 7.4 (continued)

What, Where, and When Monitoring Should Occur

The approach described below focuses only on implementation monitoring and not on effectiveness monitoring as required in the Recovery Plan. Implementation monitoring addresses (1) whether treatments within threshold areas were successful in maintaining habitat attributes at or above the levels shown in Table 7.1 and (2) posttreatment changes in Key Habitat Components (defined below) as the direct or indirect result of management activities were roughly equivalent to predicted changes.

It is also important to know how well treatments in restricted areas (including target and threshold) and protected areas retain Key Habitat Components. Key Habitat Components of Mexican spotted owl habitat include large trees, snags, logs, and hardwoods, and must be retained in adequate quantities and distributions (USDI Fish and Wildlife Service 1995, pp. 94–95). The objectives of monitoring treatments in these areas is to evaluate whether actual losses in the Key Habitat Components exceed the losses predicted during project planning, to quantify the loss of these components, and then adjust future prescriptions as appropriate.

Table 7.1 Target/threshold conditions for restricted area mixed-conifer and pine-oak forests. Table III.B.1. from USDI (1995)

	% Stand density				
Recovery Units Forest Type	trees 30-45 cm dbh	trees 30-45 cm dbh	trees >60n dbh	Basal area (m/ha)	Tree>45 cm (number/ha)
Basin and Range - East RU					
Mixed-conifer	10	10	10	32	49
Mixed-conifer	10	10	10	39	49
All RUs, except Basin and Range - East RU					
Mixed-conifer	10	10	10	32	49
Mixed-conifer	10	10	10	39	49
Colorado Plateau, Upper Gila Mountains, Basin and Range - West RUs					
Pine-oak[a]	15	15	15	32	49

[a]For pine-oak, 20ft^2/acre of oak must be provided as a threshold/target condition
Source: table III.B.1. from USDI Fish and Wildlife Service (1995)

Variables Assessed in Microhabitat Monitoring

The variables assessed in these protocols are those identified in the Recovery Plan to be habitat correlates of Mexican spotted owls and their prey. These variables include Key Habitat Components and Fine Filter Factors that apply to all protected and restricted areas, and variables derived from Table 7.1 that apply only to target/threshold areas. Generally, all variables listed below are directly from the Recovery Plan or are based on our interpretation of the Recovery Plan.

Key Habitat Components

The variables listed in the Recovery Plan (USDI Fish and Wildlife Service 1995, pp. 94, 107) are of importance to the habitat of Mexican spotted owls and their prey. These variables include:

- Number of trees >60 cm diameter at breast height (dbh) for conifers, or diameter at root collar (drc) for hardwoods
- Number of trees 48–60 cm dbh/drc
- Number of logs >30 cm at 1.5 m from the large end and 1.3-m long
- Number of live hardwood stems >12 cm drc
- Number of snags >30 cm dbh and >1.3 tall
- Total basal area of trees >12 cm dbh/drc

Table 7.1 Variables

Additional variables must be measured in threshold areas to evaluate whether threshold values (see Table 7.1) were maintained following treatment. The variables must also be measured in target areas to evaluate how close post-treatment conditions are relative to values in Table 7.1. The variables needed in addition to the Key Habitat Components include:

- Number of live trees 30–45 cm dbh/drc
- Number of live trees 12–29.9 cm dbh/drc
- Number of live trees 2.5–11.9 cm dbh/drc

These measurements will also allow for calculations of total tree basal area, the distribution of stand density across diameter classes, and the density of large trees (i.e., those >45 cm dbh/drc).

Procedures for Monitoring Key Habitat Components

Planning/Project Design

The purpose of monitoring Key Habitat Components is to index their change in treated areas. Thus, treatment plans must state treatment objectives and quantify projected changes to each Key Habitat Component (such as the expected percentage loss of each component) as result of the treatment. Losses are considered in two ways. One is the total percentage loss of the component across the project area. The other loss is the percentage loss of the component on a plot-level basis. If the loss of Key Habitat Components during implementation exceeds those predicted during the analysis, then prescriptions should be adjusted to mitigate excessive losses in future projects.

Two criteria are considered when evaluating project implementation. One is the spatial extent of the loss of each Key Habitat Component. Thus, the number of plots in which this change occurs provides an index of how much area was affected. The other is assessing the magnitude of the loss of each component across the project area. Both should be considered simultaneously because the plots where a component (e.g., large trees, large logs) was lost

(continued)

Box 7.4 (continued)

may have been where it mostly occurred. Thus, even though only a few plots may be affected, the actual loss of a component may have been large. Considering losses both spatially and in magnitude is important when evaluating project implementation and for planning future treatments.

Analysis and Rationale for Number of Plots Needed

The minimum number of plots needed to be sampled will probably differ between the plot- and project-level analyses because different statistical analyses will be used. Plot-level analyses are based on a one-sided chi-square test, whereas project-level analyses are based on a paired-sample t test.

Plot-Level Analysis

A one-sided chi-square test (Marascuilo and McSweeny 1977, pp. 196–198) is the basis for the plot-level analysis. This is applied as a one-tailed test with the level of significance at 0.05. The test assesses if implementation of the prescription resulted in excessive loss (i.e., more than specified in the treatment plans; thus a one-tailed t test) of each Key Habitat Component on a plot-level basis.

Two proportions are considered in this analysis: null hypothesis proportion of plots (hereafter null proportion) and the observed proportion. The null proportion is the predicted proportion of plots where the loss of a Key Habitat Component will exceed a predetermined threshold value. The observed proportion of plots is that where loss of a Key Habitat Component was exceeded from posttreatment measurements.

Necessary sample size is based on the null proportion and the statistical power of detecting an increase over the null proportion (Table 7.2). Only null proportions between 0 and 10% were considered because monitoring is conducted on key components of spotted owl habitat; thus, a "light trigger" is needed to measure excessive losses of these components. Statistical power was set at $P = 0.9$ for detecting small increases for the same reason.

Table 7.2 Minimum sample sizes for plot-level analysis based on the null hypothesis of the proportion of plots affected by treatment

Type I error	Statistical power	Null proportion	Sample size
0.05	0.90	0	25
0.05	0.90	0.10	50

Table 7.2 specifies necessary minimum sample sizes for two null proportions. Application of these sample sizes will depend on the particular Key Habitat Component and the number of acres treated. See below for more specific guidelines.

This analysis involves a two-step process to evaluate whether the treatment was implemented correctly. The first is to compare the observed proportion of plots where losses exceeded predictions under the null hypothesis. If the observed proportion is less than the null proportion, then the project was successful from a spatial standpoint. If the observed proportion is greater than the null proportion, the analysis should proceed to the second step.

In the second step, $P = 0.95$ one-sided confidence limits on the observed proportion are compared to the null proportion. Figure 7.10 contains confidence limits plotted for a range of observed proportion with sample specified at $n = 25$ and $n = 50$, respectively. In Figure 7.10a, if an observed proportion is 0.05 or larger, the lower confidence limit exceeds the null proportion of 0 and the project should be judged as unsuccessful. Also, based on the upper confidence limit, the "true" proportion of plots exceeding predicted losses might be 20% or more, an unacceptable level. In other words, we estimated the proportion to be 0.05 based on a sample, but the real effect could be much higher, 20% or more.

In Fig. 7.10b, if the observed proportion is 0.18 or larger, the lower confidence limit exceeds the null proportion of 0.10 and the project should be judged unsuccessful. The upper confidence limit on the "true" proportion is 0.30, also an unacceptable level.

The lower and upper confidence bounds can be calculated empirically by (Fleiss 1981, pp. 14–15)

$$\text{Lower limit} = ((2np + c^2 - 1) - c(c^2 - (2 + 1 / n) + (4\,p(nq + 1))^{1/2}))/(2n+2c^2),$$

$$\text{Upper limit} = ((2np + c^2 - 1) + c(c^2 - (2 + 1 / n) + (4\,p(nq + 1))^{1/2}))/(2n+2c^2),$$

where n = sample size, p = observed proportion, $q = 1 - p$, c = value from the normal distribution corresponding to $1 - (\alpha/2)$.

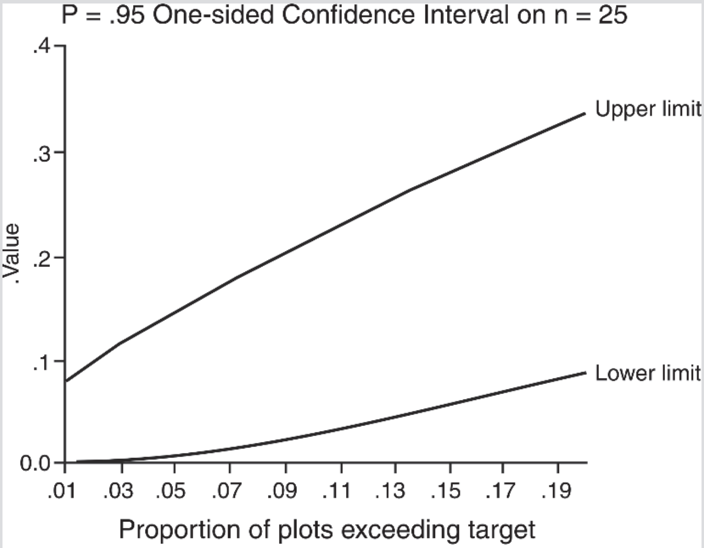
For example, authors of an environmental assessment done for the Podunk Ranger District estimate that 20% of the snags (magnitude loss) will be lost within a 603-acre project area because of prescribed fire. In this case, the null proportion would be 10 and the necessary sample 50 (see Table 7.2). Posttreatment monitoring indicated that >20% of the snags were lost on 11 of the 50 plots (22%). Since the observed proportion (0.22) was greater than 0.18, the lower confidence limit exceeds 0.10 and the project should be judged as unsuccessful. It is also worth noting that the upper confidence limit at this point is 0.34, a very high level.

Project-Level Analysis

Losses to habitat components do not occur evenly over a project area; for example, some areas in a management-ignited fire might crown out while other areas may not burn at all. Because of this, a proportion of plots should have losses that exceed what was predicted over the treatment area. Although excessive losses may occur in patches within a treatment area, it does not

(continued)

Box 7.4 (continued)



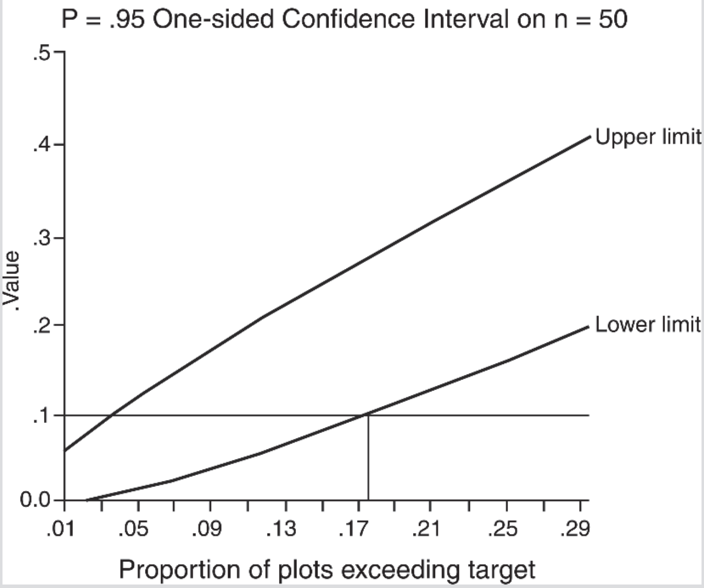


Fig. 7.10 Confidence limits plotted for a range of observed proportions with sample size specified at (**a**) n = 25 and (**b**) n = 50

mean that the treatment was implemented unsuccessfully. However, if the areas where losses occur are where most of a particular component is found, then most of that component may be lost and one may conclude statistically that the treatment was successful.

The basis for this analysis is a paired *t* test (Sokal and Rohlf 1969, p. 332). One group is the set of plots measured prior to the treatment; the other group is the same plots measured posttreatment. Sample size can be calculated empirically using the methodology presented by Sokal and Rohlf (1969, p. 247).

This approach requires that we know the standard deviation, state the difference that we want to detect in the loss of each Key Habitat Component, and the statistical power or probability that the difference will be significant. The sample size equation takes the form (Sokal and Rohlf 1969, p. 247)

$$n > 2\,(\sigma / \delta)^2\ \{t_{\alpha[v]} + t_{2(1-P)[v]}\}^2,$$

where, n is the sample size, σ is the standard deviation of the differences, δ is the smallest difference desired to be detected, α is the significance level, v is the degrees of freedom of the sample standard deviation, and $t_{\alpha[v]}$ and $t_{2(1-P)[v]}$ are the values from a two-tailed table with n degrees of freedom and corresponding to probabilities of α and $2(1 - P)$, respectively.

This test should be done as a one-tailed test with $\alpha = 0.05$. If a two-tailed table is used, then use critical values for $\alpha = 0.10$. This test also requires knowledge of the standard deviation for each variable. This can be obtained with a pilot study or from comparable existing data. A finite population correction factor should be used in calculating the standard deviation, which effectively reduces the standard deviation and needed sample sizes. Calculation of the standard deviation with a finite population correction factor takes the generalized form:

$$\sigma_c = \sigma_u\,(1 - \phi),$$

where σ_c is the finite population corrected standard deviation, σ_u is the uncorrected deviation, and ϕ is the ratio of the proportion of the area sampled out of the total area.

Box 7.5 Population Monitoring for the Mexican Spotted Owl

The Mexican Spotted Owl Recovery Plan sketched out a procedure for estimating population trend of the subspecies with the three most populated recovery units (USDI Fish and Wildlife Service 1995). Populations in the other, more sparsely populated recovery units were not monitored because of logistical difficulties in designing and implementing an unbiased sampling approach. Implicit to monitoring birds in only three recovery units was the assumption that they represented the core populations, and trends in these populations would apply to the overall trend in owl populations throughout its geographic range. We briefly describe below some of the primary components of the population monitoring sampling design.

The target population to be sampled consists of territorial owls in the three recovery units mentioned above. Thus, all potential owl habitat in these three recovery units must be included in the sampling frame. Sampling units will consist of randomly spaced quadrats 50–75 km^2 in size. The intent is to evaluate populations subjected to all factors, natural and anthropogenic, that may influence trends. Quadrats should be stratified by vegetation type and owl density within each recovery unit. A certain percentage of quadrats would be replaced each year to guard against selective management practices being practiced inside quadrats that were not being done outside of quadrats. For example, an agency may avoid cutting timber within quadrats to minimize any potential effects on owls. If timber harvest does influence owl population trends, exclusion of this practice from the sampling units may result in a biased estimate of population trends.

Within quadrats, sampling will consist of the following aspects. Survey stations will be placed to ensure adequate coverage of the quadrat. Each survey station will be sampled at night four times during the breeding season to locate territorial adult owls. Multiple visits to each survey station will allow for estimation of detection probabilities, which can be used to adjust raw counts. Adjusted counts are then transformed to density estimates for each quadrat and then aggregated for estimates within strata. Auxiliary diurnal sampling will be done to visually locate birds detected during nighttime surveys, assess reproductive status, color-band adult and juvenile owls, and provide a check on the accuracy of nighttime surveys. For example, nighttime surveys might only detect a male owl, when the territory is occupied by a pair. Furthermore, color-banding individuals may allow alternative ways to estimate population trends based on analyses of mark–recapture data.

A monitoring program of this magnitude and complexity has rarely, if ever, been conducted on a wildlife species. Thus, a pilot study was needed to evaluate sampling intensity or the number of quadrats needed for precise estimates of the relevant population parameters. Once completed, the responsible management agencies must decide whether they have the resources and commitment to implement the program and carry it through fruition (White et al. 1999).

Ganey et al. (2004) reported the results of a pilot study conducted in 1999. The study occurred with the Upper Gila Mountains Recovery Unit on 25 40–76 km^2 quadrats. Quadrats were stratified into high and low density, and field sampling followed established mark–recapture protocols for this subspecies. They concluded that the approach was possible but infeasible given costs and logistics of conducting field samples. They also found that temporal variation inherent to Mexican spotted owl populations was so large, the power to detect a population trend was relatively low. They proposed occupancy monitoring as a cost-effective alternative to mark–recapture, a proposal under serious consideration by the Mexican Spotted Owl Recovery Team.

7.9 Field Applications

7.9.1 Short-term and Small Area Applications

Many inventory and monitoring studies are short term occurring within a restricted area or both. Indeed, studies done to inventory an area for species of concern prior to implementing a habitat-altering activity do not have the luxury of a long-term study. To accommodate this situation, specific studies should be done or existing data sets should be analyzed to establish the minimum amount of time for the study to provide reliable information. For rare or elusive species, such studies would focus on the amount of time and number of sampling points needed to detect a species if it is present. For studies whose goal is to develop list of the species present, pilot studies or existing data could be used to develop species accumulation curves that can help to define the amount of effort needed to account for most species present (Fig. 7.2).

Often studies are restricted in the amount of area available for study. This may occur in an island situation either in the traditional sense or when a patch of vegetation is surrounded by a completely different vegetation type (e.g., riparian habitats in the southwest). Small areas also occur when the area of interest is restricted. An example is when development is planned for a small parcel of land and the objective is to evaluate the species potentially affected within that parcel. In these situations, you are not so much faced with a sampling problem as you are with a sample size problem. Given the small area, you should strive to detect every individual and conduct a complete census. Even so, you may have too few data to permit rigorous treatment of the data for many of the species encountered. Various tools such as rarefaction and bootstrapping can be used to compensate for small samples encountered in small areas studies.

7.9.2 Long-term and Regional Applications

Ideally, monitoring studies should occur over long periods. The objective of such studies is to document trends that can help to inform predictions of future trajectories. Many of these monitoring programs also occur over wide geographic areas such the Breeding Bird Survey, Christmas Bird Counts, and Forest Inventory and Assessment. The logistics of implementing such large-scale, long-term monitoring programs is daunting, and potentially compromises integrity of the data. Perhaps the major hurdle of long-term, regional studies is to make sure that protocols are followed consistently over time. For example, the Breeding Bird Survey is a long-term monitoring program that occurs throughout United States (Sauer et al. 2005). The survey entails conducting point counts along established road transect. Unfortunately, coverage of these transects varies from year to year, which reduces the effectiveness of the monitoring program and necessitates innovative analyses to fills in the gaps. Transects are sampled by a large number of people, with varying levels of expertise, skill, and ability. A similar situation occurs with habitat monitoring and the US Forest Service's Forest Inventory and Assessment program. This program includes vegetation plots on a 5,000-m grid with plots on lands regardless of ownership (i.e., not just on Forest Service land). For various reasons the Forest Service has altered the sampling design by changing the number of plots surveyed, revising measurement protocols, and the frequency at which they sample points. These changes effectively compromise the ability to examine long-term trends because of the difficulty of sorting out variation ascribed to changes in sampling protocols from variation resulting from vegetation change.

7.10 Summary

Inventory and monitoring are key aspects of wildlife biology and management; they can be done in pursuit of basic knowledge or as part of the management process. Inventory is used to assess the state or status of one or more resources, whereas monitoring is typically done to assess change or trend. Monitoring can be classified into four overlapping categories:

- Implementation monitoring is used to assess whether or not a directed management action was carried out as designed.
- Effectiveness monitoring is used to evaluate whether a management action met its desired objective.
- Validation monitoring is used to evaluate whether an established management plan is working.
- Compliance monitoring is used to see if management is occurring according to established law.

Selecting the appropriate variable to inventory or monitor is a key aspect of the study design; direct measures, such as population numbers, are preferred over indirect measures, such as indicator species. The length of monitoring studies depends largely on the process or variable being studied. The appropriate length often exceeds available resources, necessitating alternative approaches such as retrospective studies, modeling, genetic tools, substituting space for time, and substituting fast for slow dynamics.

Time, cost, and logistics often influence the feasibility of what can be done. Use of indices can be an effective way to address study objectives provided data are collected following an appropriate study design and data are analyzed correctly. Indices can be improved and calibrated using ratio-estimation and double-counting techniques.

Monitoring effects of management actions requires a clear and direct linkage between study results and management activities, often expressed as a feedback loop. Feedback is essential for assessing the efficacy of monitoring and for validating or changing management practices. Failure to complete the feedback process negates the intent and value of monitoring.

References

Allen, T. F. H., and T. W. Hoekstra. 1992. Towards a Unified Ecology. Columbia University Press, New York, NY.

Anderson, D. R. 2001. The need to get the basics right in wildlife field studies. Wildl. Soc. Bull. 29: 1294–1297.

Anderson, D. R. 2003. Response to Engeman: Index values rarely constitute reliable information. Wildl. Soc. Bull. 31: 288–291.

Aubry, K., S. Wisley, C. Raley, and S. Buskirk. 2004. Zoogeography, pacing patterns and dispersal in fishers: insights gained from combining field and genetic data, in D. J. Harrison, A. K. Fuller, and G. Proulx, Eds. Martins and Fishers (Martes) in Human-Altered Environments: An International Perspective, pp. 201–220. Springer Academic, New York, NY.

Bart, J., S. Droge, P. Geissler, B. Peterjohn, and C. J. Ralph. 2004. Density estimation in wildlife surveys. Wildl. Soc. Bull. 32: 1242–1247.

Block, W. M., and L. A. Brennan. 1993. The habitat concept in ornithology: Theory and applications, in D. M. Power, Ed. Current Ornithology, vol. 11, pp. 35–91. Plenum, New York, NY.

Block, W. M., M. L. Morrison, J. Verner, and P. N. Manley. 1994. Assessing wildlife–habitat-relationships models: A case study with California oak woodlands. Wildl. Soc. Bull. 22: 549–561.

Boyce, M. S. 1992. Population viability analysis. Annu. Rev. Ecol. Syst. 23: 481–506.

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach, 2nd Edition. Springer-Verlag, New York, NY.

Caswell, H. 2001. Matrix Population Models: Construction, Analysis, and Interpretation, 2nd Edition. Sinauer Associates, Inc., Sunderland, MA.

Clements, E. E. 1920. Plant Indicators. Carnegie Institute, Washington, DC.

Cochran, W. G. 1977. Sampling Techniques, 3rd Edition. Wiley, New York, NY.

Cohen, J. 1988. Statistical Power Analysis for the Behavioral Sciences, 2nd Edition. Erlbaum, Hillsdale, NJ.

Cohen, J. 1992. A power primer. Psychol. Bull. 112: 155–159.

Davis, M. B. 1989. Retrospective studies, in G. E. Likens, Ed. Long-Term Studies in Ecology: Approaches and Alternatives, pp. 71–89. Springer-Verlag, New York, NY.

DeAngelis, D. L., and L. J. Gross. 1992. Individual-Based Models and Approaches in Ecology. Chapman and Hall, London.

Dunham, A. E., and S. J. Beaupre. 1998. Ecological experiments: Scale, phenomenology, mechanism, and the illusion of generality, in J. Bernardo and W. J. Resetarits Jr., Eds. Experimental Ecology: Issues and Perspectives, pp. 27–49. Oxford University Press, Oxford.
Eberhardt, L. L., and M. A. Simmons. 1987. Calibrating population indices by double sampling. J. Wildl. Manage. 51: 665–675.
Efron, B. 1982. The Jackknife, The Bootstrap, and Other Resampling Plans. Society for Industrial and Applied Mathematics, Philadelphia, PA, 92 pp.
Engeman, R. M. 2003. More on the need to get the basics right: Population indices. Wildl. Soc. Bull. 31: 286–287.
Fleiss, J. L. 1981. Statistical Methods for Rates and Proportions, 2nd Edition. Wiley, New York, NY.
Franklin, J. F. 1989. Importance and justification of long-term studies in ecology, in G. E. Likens, Ed. Long-Term Studies in Ecology: Approaches and Alternatives, pp. 3–19. Springer-Verlag, New York, NY.
Ganey, J. L., G. C. White, D. C. Bowden, and A. B. Franklin. 2004. Evaluating methods for monitoring populations of Mexican spotted owls: A case study, in W. L. Thompson, Ed. Sampling Rare and Elusive Species: Concepts, Designs, and Techniques for Estimating Population Parameters, pp. 337–385. Island Press, Washington, DC.
Gray, P. A., D. Cameron, and I. Kirkham. 1996. Wildlife habitat evaluation in forested ecosystems: Some examples from Canada and the United States, in R. M. DeGraaf and R. I. Miller, Eds. Conservation of Faunal Diversity in Forested Landscapes, pp. 407–536. Chapman and Hall, London.
Green, R. H. 1979. Sampling Design and Statistical Methods for Environmental Biologists. Wiley, New York, NY.
Haig, S. M., T. D. Mullins, E. D. Forsman, P. W. Trail, and L. Wennerberg. 2004. Genetic identification of spotted owls, barred owls, and their hybrids: Legal implications of hybrid identity. Conserv. Biol. 18: 1347–1357.
Hayek, L. C. 1994. Analysis of amphibian biodiversity data, in W. R. Heyer, M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster, Eds. Measuring and Monitoring Biological Diversity: Standard Methods for Amphibians, pp. 207–269. Smithsonian Institution Press, Washington, DC.
Hayes, J. P., and R. J. Steidel. 1997. Statistical power analysis and amphibian population trends. Conserv. Biol. 11: 273–275.
Hedges, L. V., and Olkin, I. (1985). Statistical Methods for Meta-Analysis. Academic, San Diego, CA.
Heyer, W. R., M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster. 1994. Measuring and Monitoring Biological Diversity: Standard Methods for Amphibians. Smithsonian Institution Press, Washington, DC.
Hohn, M. E. 1976. Binary coefficients: A theoretical and empirical study. J. Int. Assoc. Math. Geol. 8: 137–150.
Holling, C. S. 1966. The functional response of invertebrate predators to prey density. Mem. Entomol. Soc. Can. 48: 1–86.
Hunter Jr., M. L. 1991. Coping with ignorance: The coarse filter strategy for maintaining biodiversity, in K. A. Kohm, Ed. Balancing on the Brink of Extinction: The Endangered Species Act and Lessons for the Future, pp. 266–281. Island Press, Washington, DC.
Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54: 187–211.
Jaccard, P. 1901. The distribution of the flora in the alpine zone. New Phytol. 11: 37–50.
Lande, R., S. Engen, and B. -E. Sæther. 2003. Stochastic Population Dynamics in Ecology and Conservation. Oxford University Press, Oxford.
Landres, P. B., J. Verner, and J. W. Thomas, 1988. Ecological uses of vertebrate indicator species: A critique. Conserv. Biol. 2: 316–328.
MacKenzie, D. I., J. A. Royle, J. A. Brown, and J. D. Nichols. 2004. Occupancy estimation and modeling for rare and elusive populations, in W.L. Thompson, Ed. Sampling Rare or Elusive Species. pp. 142–172. Island Press, Covelo, CA.
Marascuilo, L. A., and M. McSweeny. 1977. Non-Parametric and Distribution-Free Methods for the Social Sciences. Brooks/Cole, Monterey, CA.

May, R. M. 1974. Stability and Complexity in Model Ecosystems, 2nd Edition. Princeton University Press, Princeton.

McCallum, H. 2000. Population Parameters: Estimation for Ecological Models. Blackwell, Malden, MA.

McCullough, D. R., and R. H. Barrett. 1992. Wildlife 2001: Populations. Elsevier, London.

McKelvey, K. S., and D. E. Pearson. 2001. Population estimation with sparse data: The role of estimators versus indices revisited. Can. J. Zool. 79: 1754–1765.

McKelvey, K. S., J. von Kienast, K. B. Aubry, G. M. Koehler, B. T. Maletzke, J. R. Squires, E. L. Lindquist, S. Loch, M. K. Schwartz. 2006. DNA Analysis of hair and scat collected along snow tracks to document the presence of Canada lynx. Wildl. Soc. Bull. 34(2): 451–455.

Miller, R. I. 1996. Modern approaches to monitoring changes in forests using maps, in R. M. DeGraaf and R. I. Miller, Eds. Conservation of Faunal Diversity in Forested Landscapes, pp. 595–614. Chapman and Hall, London.

Moir, W. H., and W. M. Block. 2001. Adaptive management on public lands in the United States: Commitment or rhetoric? Environ. Manage. 28: 141–148.

Morrison, M. L. 1986. Birds as indicators of environmental change. Curr. Ornithol. 3: 429–451.

Morrison, M. L. 1992. The design and importance of long-term ecological studies: Analysis of vertebrates in the Inyo-White Mountains, California, in R. C. Szaro, K. E. Severson, and D. R. Patton, Tech. Coords. Management of Amphibians, Reptiles, and Small Mammals in North America, pp. 267–275. USDA Forest Service, Gen. Tech. Rpt. RM-166. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Morrison, M. L., and B. G. Marcot. 1995. An evaluation of resource inventory and monitoring programs used in national forest planning. Environ. Manage. 19: 147–156.

Morrison, M. L., B. G. Marcot, and R. W. Mannan. 1998. Wildlife–Habitat Relationships: Concepts and Application, 2nd Edition. University of Wisconsin Press, Madison, WI.

Noon, B. R., T. A. Spies, and M. G. Raphael. 1999. Conceptual basis for designing an effectiveness monitoring program, in B. S. Mulder, B. R. Noon, T. A. Spies, M. G. Raphael, C. J. Palmer, A. R. Olsen, G. H. Reeves, and H. H. Welsh, Tech. Coords. The Strategy and Design of the Effectiveness Monitoring Program in the Northwest Forest Plan, pp. 21–48. USDA Forest Service, Gen. Tech. Rpt. PNW-GTR-437. Pacific Northwest Research Station, Portland, OR.

Pickett, S. T. A. 1989. Space-For-Time Substitutions as an Alternative to Long-Term Studies, in G. E. Likens, Ed. Long-Term Studies in Ecology: Approaches and Alternatives, pp. 110–135. Springer-Verlag, New York, NY.

Pielou, E. C. 1977. Mathematical Ecology, 2nd Edition. Wiley, New York, NY.

Reynolds, R. T., and S. M. Joy. 2006. Demography of Northern Goshawks in Northern ARIZONA, 1991–1996. Stud. Avian Biol. 31: 63–74.

Rinkevich, S. E., J. L. Ganey, W. H. Moir, F. P. Howe, F. Clemente, and J. F. Martinez-Montoya. 1995. Recovery units, in Recovey Plan for the Mexican Spotted Owl (*Strix occidentalis lucida*), vol. I, pp. 36–51. USDI Fish and Wildlife Service, Southwestern Region, Albuquerque, NM.

Romesburg, H. C. 1981. Wildlife science: Gaining reliable knowledge. J. Wild. Manage. 45: 293–313.

Rosenstock, S. S., D. R. Anderson, K. M. Giesen, T. Leukering, and M. E. Carter. 2002. Landbird counting techniques: Current practices and alternatives. Auk 119: 46–53.

Sauer, J. R., J. E. Hines, and J. Fallon. 2005. The North American Breeding Bird Survey, Results and Analysis 1966–2005. Version 6.2.2006. USGS Patuxent Wildlife Research Center, Laurel, MD

Schwartz, M. K. 2007. Ancient DNA confirms native rocky mountain fisher *Martes pennanti* avoided early 20th century extinction. J. Mam. 87: 921–92.

Schwartz, M. K., K. L. Pilgrim, K. S. McKelvey, E. L. Lindquist, J. J. Claar, S. Loch, and L. F. Ruggiero. 2004. Hybridization between Canada lynx and bobcats: Genetic results and management implications. Conserv. Genet. 5: 349–355.

Schwartz, M. K., G. Luikart, and R. S. Waples. 2007. Genetic monitoring as a promising tool for conservation and management. Trends Ecol. Evol. 22: 25–33.

Scott, J. M., F. Davis, B. Csuti, R. Noss, B. Butterfield, C. Grives, H. Anderson, S. Caicco, F. D'Erchia, T. Edwards Jr., J. Ulliman, and R. G. Wright. 1993. Gap analysis: A geographical approach to protection of biodiversity. Wildl. Monogr. 123.

Sharp, A., M. Norton, A. Marks, and K. Holmes. 2001. An evaluation of two indices of red fox (Vulpes vulpes) abundance in an arid environment. Wildl. Res. 28: 419–424.

Shugart, H. H. 1989. The role of ecological models in long-term ecological studies, in G. E. Likens, Ed. Long-Term Studies in Ecology: Approaches and Alternatives, pp. 90–109. Springer-Verlag, New York, NY.

Sokal R. R., and C. D. Michener. 1958. A statistical method for evaluating systematic relationships. Univ. Kansas Sci. Bull. 38: 1409–1438.

Sokal, R. R., and F. J. Rohlf. 1969. Biometry: The Principles and Practice of Statistics in Biological Research. Freeman, San Francisco, CA.

Sorensen, T. 1948. A method for establishing groups of equal amplitude in plant sociology based on similarity of species content, and its applications to analyses of the vegetation on Danish commons. Det Kongelige Danske Viden-skkabernes Selskab, Biloogiske Skrifter 5: 1–34.

Spellerberg, I. F. 1991. Monitoring Ecological Change. Cambridge University Press, New York, NY.

Steidl, R. J., J. P. Hayes, and E. Schauber. 1997. Statistical power analaysis in wildlife research. J. Wildl. Manage. 61: 270–279.

Strayer, D., J. S. Glitzenstein, C. G. Jones, J. Kolasa, G. Likens, M. J. McDonnell, G. G. Parker, and S. T. A. Pickett. 1986. Long-Term Ecological Studies: An Illustrated Account of Their Design, Operation, and Importance to Ecology. Occas Pap 2. Institute for Ecosystem Studies, Millbrook, New York, NY.

Swetnam, T. W. 1990. Fire history and climate in the southwestern United States, in J. S. Krammes, Tech. Coord. Effects of Fire Management of Southwestern Natural Resources, pp. 6–17. USDA Forest Service, Gen. Tech. Rpt. RM-191. Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Swetnam, T. W., and J. L. Bettancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. J. Climate 11: 3128–3147.

Thomas, L. 1997. Retorspective power analysis. Conserv. Biol. 11: 276–280.

Thompson, S. K. 2002. Sampling, 2nd Edition. Wiley, New York, NY.

Thompson, W. L., G. C. White, and C. Gowan, 1998. Monitoring Vertebrate Populations. Academic, San Diego, CA.

USDA Forest Service. 1996. Record of Decision for Amendment of Forest Plans: Arizona and New Mexico. USDA Forest Service, Southwestern Region, Albuquerque, NM.

USDI Fish and Wildlife Service. 1995. Recovery Plan for the Mexican Spotted Owl (*Strix occidentalis lucida*), vol. I. USDI Fish and Wildlife Service, Albuquerque, NM.

Verner, J. 1983. An integrated system for monitoring wildlife on the Sierra Nevada Forest. Trans. North Am. Wildl. Nat. Resour. Conf. 48: 355–366.

Verner, J., and L. V. Ritter. 1985. A comparison of transects and point counts in oak-pine woodlands of California. Condor 87: 47–68.

Verner, J., M. L. Morrison, and C. J. Ralph. 1986. Wildlife 2000: Modeling Habitat Relationships of Terrestrial Vertebrates. University of Wisconsin Press, Madison, WI.

Verner, J., R. J. Gutiérrez, and G. I. Gould Jr. 1992. The California spotted owl: General biology and ecological relations, in J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould Jr., and T. W. Beck, Tech. Coords. The California Spotted Owl: A Technical Assessment of Its Current Status, pp. 55–77. USDA Forest Service, Gen. Tech. Rpt. PSW-GTR-133. Pacific Southwest Research Station, Berkeley, CA.

Walters, C. J. 1986. Adaptive Management of Renewable Resources. Macmillan, New York, NY.

White, G. C. 2005. Correcting wildlife counts using detection probabilities. Wildl. Res. 32: 211–216.

White, G. C., and J. D. Nichols, 1992. Introduction to the methods section, in D. R. McCullough and R. H. Barrett, Eds. Wildlife 2001: Populations, pp. 13–16. Elsevier, London.

White, G. C., W. M. Block, J. L. Ganey, W. H. Moir, J. P. Ward Jr., A. B. Franklin, S. L. Spangle, S. E. Rinkevich, R. Vahle, F. P. Howe, and J. L. Dick Jr. 1999. Science versus reality in delisting criteria for a threatened species: The Mexican spotted owl experience. Trans. North Am. Wildl. Nat. Resour. Conf. 64: 292–306.