Production of this book was made possible through the generosity of the following organizations: the Wildlife Conservation Society, the U.S. Fish & Wildlife Service's Asian Elephant Conservation Fund and the International Elephant Foundation.

The Wildlife Conservation Society (WCS) was established at the Bronx Zoo in the USA in 1895. WCS strives to develop, implement, provide and promote on-site, long-term, science-based field approaches to the conservation of wildlife, wildlife habitats and biodiversity more generally. WCS works on over 500 conservation projects in 56 countries. WCS has many decades of experience working on elephant ecology and conservation, population monitoring, law enforcement and human–elephant conflict in Africa and Asia.

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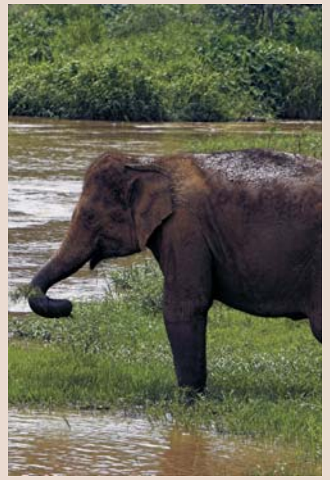

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Simon Hedges (Ed.)

MONITORING ELEPHANT POPULATIONS AND ASSESSING THREATS

Simon Hedges (Ed.)

Monitoring Elephant Populations and Assessing Threats

a manual for researchers, managers and conservationists

Universities Press

This peer-reviewed manual presents a conceptually-unified and statistically rigorous approach to monitoring elephant populations. The authors, who between them have many decades of experience in statistics, wildlife monitoring and elephant conservation work in Asia and Africa, present an array of methods for estimating elephant population size and distribution and for monitoring threats. The manual contains a pair of chapters for each of the major methods covered, with the first of the pair covering the underlying theory and the second covering practical field methods and recommendations. However, the practical chapters have been written so as to be as 'standalone' as possible; in other words, it should be possible to read a practical chapter and gain a good idea of how to use a particular method in the field without necessarily reading the entire theoretical chapter. This manual represents, therefore, a practical tool that will help address current elephant population monitoring needs and which will be of use to wildlife managers, conservationists and elephant researchers.

Monitoring Elephant Populations and Assessing Threats

a manual for researchers, managers and conservationists

Simon Hedges (Ed.)

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Front cover photograph:
African savannah elephants in Namibia, photograph © Simon Hedges

Back cover photographs clockwise from top left:
African forest elephant in the Congo Basin, photograph © Stephen Blake

Asian Elephant in Sri Lanka, photograph © Simon Hedges

Collecting fecal DNA samples for a capture–recapture based survey in Myanmar, photograph © Simon Hedges / WCS

Measuring the circumference of an elephant's dung bolus to estimate the age of the elephant, photograph © Riza Marlon / WCS Indonesia Program

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CONTENTS

PREFACE

Elephants still occur in isolated populations across much of their historical range, but unfortunately their numbers are rapidly declining. The major threats to the continued survival of these species in many places are habitat loss, degradation and fragmentation, as well as poaching for ivory and other forms of illegal killing or capture—usually as a result of conflicts with humans. Effective monitoring programs, which involve systematic collection of data on the distribution, size and trend of elephant populations, as well as threats such as illegal killing, are needed to provide a rational basis for the management of elephant populations. For example, a major component of the CITES Monitoring the Illegal Killing of Elephants (MIKE) program is the estimation of elephant population size and trend, with a commitment to long-term population monitoring. Unfortunately, it is often difficult to evaluate the efficacy of current elephant conservation interventions and policies, because rigorous monitoring is absent. As a consequence, in spite of many decades of elephant research and conservation efforts, there are still significant gaps in what we know about the distribution, status and trend of elephant populations, especially in the forests of Southeast Asia and Central Africa. In light of this situation, and considering recent major advances in animal population sampling and monitoring techniques, it was clear that a comprehensive manual describing appropriate methods for monitoring populations of elephants was needed. The Wildlife Conservation Society (WCS) therefore sought funding from the USFWS for the production and publication of an easy-to-use but comprehensive manual specifically focussing on the methods applicable for monitoring elephant populations across the range of ecological conditions in which they occur. The intention was to have a manual similar in style to the well-received and widely-used book *Monitoring tigers and their prey: A manual for wildlife managers, researchers, and conservationists in tropical Asia,* edited and written (in very large part) by K. Ullas Karanth and Jim Nichols, while also ensuring maximum compatibility with the *Dung Survey Standards for the MIKE Programme* compiled and edited by Simon Hedges and David Lawson.

As with the *Monitoring tigers and their prey* manual, we have chosen to present in this manual only those monitoring approaches that are adequately supported by peer-reviewed research and development. Please note, too, that the extent of historical coverage and discussion of the theoretical background varies significantly between chapters. For example, chapters covering subjects such as capture–recapture methods—for which there are many other easily available texts describing the development of the methods and underlying theory—do not cover that development. By contrast, we provide significantly more detail for methods for which there are few if any reviews describing the underlying theory and developments thereof (e.g., methods based on dung density). We adopted this approach to avoid making the book unnecessarily long, and to avoid duplicating material available elsewhere (e.g., for occupancy methods, there is the very thorough review provided by the *Occupancy Estimation and Modeling* book by Mackenzie et al. plus the additional discussion and practical examples in *Quantitative Conservation of*

Vertebrates by Michael Conroy and John Carroll). In addition, we have concentrated on describing the applications of the methods described to elephants, rather than providing a detailed review of all aspects of those methods.

The basic structure of this manual is to have a pair of chapters covering each major method, with the first of the pair covering the underlying theory and the second covering practical field methods and recommendations. Chapter 2 provides guidance on which chapters cover the methods likely to be appropriate for addressing a reader's needs (see especially Tables 2.1 and 2.2). The practical chapters have been written so as to be as 'stand-alone' as possible—in other words, it should be possible to read a practical chapter and gain a good idea of how to use a particular method in the field without necessarily reading all of the theoretical chapter (this necessitated some repetition among chapters, but we have tried to keep such repetition to a minimum).

Finally, we recognise that the methods described in this manual will be improved through the process of testing and scientific review, and some will eventually be replaced altogether by better methods. Indeed, we look forward to seeing such improvements and we encourage readers to experiment with these methods and communicate their findings. In the meantime, however, we hope that this manual will provide a practical tool that helps address current elephant population monitoring needs.

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The editor and the majority of the authors who worked on this manual were supported by the Wildlife Conservation Society (WCS). The U.S. Fish & Wildlife Service and the International Elephant Foundation also provided financial support for the preparation and distribution of the manual through grants to WCS. We are very grateful to all these institutions.

We would also like to acknowledge the generous financial support provided by the International Elephant Foundation, the Centre for Wildlife Studies, Saving Wild Tigers and WCS that facilitated an International Workshop on *Monitoring Asian Elephant Populations and Assessing Threats* that helped sow some of the seeds that grew eventually into this manual. The participants at that workshop were R. M. Ray, S. S. Bist, A. K. Garg, R. Raju, D. Yatish Kumar, K. Ullas Karanth, Simon Hedges, Richard Barnes, Jim Nichols, Samantha Strindberg, Fiona Maisels, A. Christy Williams, Peter Leimgruber, R. Sukumar, Charles Santiapillai, Devaka Weerakoon, Arun Venkataraman, Ajay Desai, M. D. Madhusudan, N. Samba Kumar, G. Viswanatha Reddy, Arjun Gopalaswamy and K. M. Chinnappa.

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We are particularly grateful to the following people who kindly peer reviewed part or all of this manual: Richard Barnes, Steve Buckland, Rene Beyers, Ken Burnham, Bob Burn, Tony Lynam, Fiona Maisels, Jim Nichols, Tim O'Brien and Martin Tyson. Tim O'Brien and Liz Bennett also provided useful comments on the content and style of the entire manual and we thank them both for that valuable service.

Finally, a significant proportion of this manual (the parts covering dung density and fecal DNA based survey methods) builds on the work of the *CITES MIKE Dung Survey Task Force* and the resulting *Dung Survey Standards for the MIKE Programme*. As such, we acknowledge the earlier work of the *Dung Survey Task Force*: Richard Barnes, Stephen Blake, Ken Burnham, Bob Burn, Holly Dublin (Facilitator), Lori Eggert, Simon Hedges, Richard Ruggiero, Karl Stromayer, Martin Tyson and Arun Venkataraman (Rapporteur) as well those additional people who also provided inputs to the Task Force's work: Nigel Hunter, David Lawson, Fiona Maisels, Andy Plumptre and Samantha Strindberg. Of course, several of these people also wrote and/or reviewed parts of this manual, thus proving their dedication to the cause of reliable elephant monitoring (and especially the intricacies of counting or collecting elephant dung!).

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CHAPTER 1

Wildlife Population Monitoring: A Conceptual Framework

James D. Nichols and K. Ullas Karanth

1.1 Introduction

This manual is not intended to be an exhaustive review of all established scientific methods of animal population estimation: there are several excellent books that deal with these issues [Seber 1982; Thompson 1992; Bookhout 1994; Wilson et al. 1996; Thompson et al. 1998; Williams et al. 2002; Conroy and Carroll 2009]. Rather, we expect the users of this manual to have at least a working knowledge of basic statistical methods and the standard literature on population estimation. However, for the sake of completeness, we provide brief overviews of those current sampling-based population estimation methods that are relevant to elephants and we also provide a glossary of technical terms [Appendix 5]. Most importantly, we emphasise recognition that the seemingly disparate approaches to counting animals are conceptually unified under a rigorous theoretical framework established in the standard works of reference [Seber 1982; Thompson 1992; Lancia et al. 1994; Thompson et al. 1998; Williams et al. 2002; Section 1.2].

Monitoring of animal populations can be defined as the estimation of absolute or relative abundance for the purpose of drawing inferences about the variation in abundance of animals over space and/or time. For example, we might focus on a single time period and try to find out whether animal abundance varies among different locations at that time. Such inferences are useful for assessing the distribution patterns of animals as well as for addressing questions about the relationship between animal abundance and factors such as habitat quality or the effectiveness of management actions. We might also focus on a single location and analyse whether animal abundance varies over time. Rates of temporal change are sometimes referred to as 'trends', and some workers view monitoring as restricted to such assessments of temporal change. In this manual, we discuss different approaches for use in abundance estimation and monitoring of populations of elephants.

Although we present various monitoring methods in this manual, we believe that it is useful to begin by emphasising the common conceptual basis that underlies all such methods. This conceptual framework clarifies the relationship among the different methods and provides a basis for considering and developing modifications and new methods.

1.2 THE STATISTICAL FRAMEWORK

1.2.1 Basic problems in counting animals

Virtually all inferences about animal populations are based on count statistics. In many cases, count statistics are provided by direct counts of animals themselves. For example, we might count the number of elephants observed while walking along a survey route (e.g., a line transect) or the number of elephants identified from DNA samples obtained from their dung. In other situations, count statistics might be based on animal signs such as dung piles. Finally, the count statistic of interest might be patches of habitat in which elephants are likely to occur.

We typically attempt to count individual elephants (or groups of elephants) when our principal goal is to assess their population size or abundance (sometimes expressed as density) or to measure the increase or decrease in elephant abundance over time. To get a full picture of an elephant population's dynamics, however, other parameters (vital rates) that drive changes in elephant numbers, such as survival, mortality, recruitment, and emigration and immigration rates may need to be assessed. This is likely to be the case if assessing the effects of poaching or habitat degradation on elephant populations are study goals.

Two basic problems confront biologists and managers who would like to use such count statistics to estimate and draw inferences about animal population size: detectability and spatial sampling [Nichols 1992;

Thompson 1992; Lancia et al. 1994; Skalski 1994; Nichols and Conroy 1996; Williams et al. 2002]. With respect to direct counts of animals, detectability refers to the usual inability to detect and enumerate all animals, regardless of the sampling or survey method being used. With respect to indirect evidence of animal presence such as dung piles, detectability refers more generally to an inequality between the count statistic and the true number of animals. Spatial sampling, on the other hand, refers to the fact that we are frequently interested in areas so large that we are unable to obtain count statistics over the entire area. Instead, we must select smaller areas thought to be representative of the entire area, with the idea that we will try to use counts on these sampled areas to draw inferences about the number of animals in the entire area.

1.2.2 Detectability

We consider detectability by first defining the following quantities obtained from a sample area (or from the entire area, if we assume that there is no need to sub-sample areas):

C = count statistic, i.e., the number of animals (or indirect sign) counted;
N = abundance, i.e., the true number of animals;
p = proportionality constant relating the count statistic and abundance.

In the case of direct counts of animals, p reflects the probability that an animal in the sampled area is counted (i.e., the probability that a member of N appears in the count statistic; p can also be viewed as the expected proportion of animals appearing in the count statistic). The following expression shows the relationship between the count statistic and abundance:

$$E(C) = Np \tag{1.1}$$

where $E(C)$ denotes the expected value (or expectation) of C.

C is a random variable and can assume different values each time a count is made. $E(C)$ can be viewed as the average value of C that would be obtained if the count statistic could be collected a large number of times in the same exact sampling situation with the same N. In cases where C represents some count of animal signs (or of something other than the animals themselves), equation (1.1) still provides a reasonable model. However, in the case of animal signs, p should not be thought of as a detection probability but simply as a coefficient relating N and C.

Estimation of abundance is based on the estimation of p in equation 1.1. If we are able to estimate the p associated with a particular count statistic (denoted by $\hat{p}$, where the 'hat' denotes an estimate), then abundance can be estimated as:

$$\hat{N} = C/\hat{p} \tag{1.2}$$

The estimator in equation (1.2) is very general, as virtually all population estimation methods [Seber 1982] for a single location can be written in this general form. For example, the count statistic under distance sampling [Buckland et al. 2001; Buckland et al. 2004; Chapter 4] is the number of animals observed and counted (e.g., along a line transect), and the perpendicular distances of these observations to the transect line are used to estimate the detectability function and, hence, p. The count statistic under capture–recapture sampling [Otis et al. 1978; Seber 1982; White et al. 1982; Pollock et al. 1990; Chapter 6] is the number of different animals caught, and the patterns of capture and recapture for individual animals are modelled in order to estimate p. While the count statistic, C, under patch occupancy sampling [MacKenzie et al. 2006], is the number of patches in which elephant sign is detected, p is the average probability of detecting elephant sign if they are present in a patch and $\hat{N}$ is the estimated number of patches that contain elephant sign. The probability $\hat{p}$ of detecting elephant sign, given that a habitat patch is really occupied by elephants, is estimated from replicated field visits that are analogous to the different sampling occasions of a capture–recapture survey.

As a numerical example of the rationale underlying equation (1.2), assume that we count 20 elephants in an area and estimate a corresponding detection probability of $\hat{p} = 0.25$; that is, we estimate that we detected about 25% of the animals when we conducted our counts. Our abundance estimate is then obtained as $\hat{N} = 20/0.25 = 80$. This estimate is intuitively reasonable in that we estimate that we detect approximately one of every four elephants, so our estimated abundance is four times the number of animals counted. Perhaps the most important consideration resulting from equations 1.1 and 1.2 is that the count statistic itself does not permit unambiguous inference about abundance. Instead, such inference requires information about the detection probability associated with the count statistic.

1.2.3 Spatial sampling

Typically, we cannot survey an entire area of interest, so we must select sample areas thought to be representative of the entire area. These sample areas will represent some fraction, α, of the total area. Unlike the situation where the fraction of animals present in a sampled area that is counted (p) must be estimated, the spatial sampling fraction (α) is typically known and requires no estimation. If we define $\hat{N}'$ to be the estimated abundance of animals in sampled areas representing fraction α of the total area of interest (one means of achieving representativeness is through simple random sampling), then abundance for the entire area of interest can be estimated as:

$$\hat{N} = \hat{N}'/\alpha \tag{1.3}$$

i.e., we simply divide the estimated abundance for the sampled locations by the fraction of the entire area represented by those locations.

As a numerical example, assume that we have randomly, or at least representatively, sampled several locations representing 10% of the entire area of interest ($\alpha = 0.10$), and that we have obtained an estimate of 80 elephants on these locations. Then the population estimate for the entire area is computed as $\hat{N} = 80 / 0.10 = 800$ elephants.

1.2.4 Canonical estimator

We can combine the above solutions to the problems of detectability and spatial sampling (equations 1.2 and 1.3) into a single, general estimator:

$$\hat{N} = C'/\hat{p}\,\alpha \tag{1.4}$$

where C' is the count statistic of sampled areas and $\hat{p}$ is the estimated detection probability, assumed equal for all samples in this expression [this need not be assumed in general; see Thompson (1992) and Skalski (1994)]. We thus estimate population size by dividing the count statistic by both the estimated fraction of the animals in the sampled area(s) that were detected ($\hat{p}$), and the proportion of the total area from which the count statistic was taken (α). We refer to expression 1.4 as the canonical estimator for abundance.

For example, assume that we count 20 elephants ($C' = 20$) in sample areas representing 10% of an area of interest, so $\alpha = 0.10$. Further assume that we are able to estimate a detection probability of 0.25 ($\hat{p} = 0.25$). Then we estimate abundance as:

$$\hat{N} = 20/(0.10)(0.25) = 800 \text{ elephants.}$$

The exact form of the variance of the canonical estimator depends on the spatial sampling design and on variation in detection probability over the different sampled locations [Thompson 1992; Skalski 1994]. However, it is useful to consider the general components of the variance estimator and their general effects on the magnitude of the variance. One component is the variance of the count statistic among the different sampled areas, *var*(C). This component is estimated using replication and will depend on the spatial distribution of animals over the study area and the selected sampling design. This component is small when animals are evenly distributed over space and large when animals are clumped. There is also a variance component associated with the estimation of p, $var(\hat{p})$. In general, smaller variances for abundance estimates, $var(\hat{N})$, result from smaller variance components, $var(C)$ and $var(\hat{p})$, and from larger values of detection probability, p, and proportion of the area that is sampled, α.

1.2.5 Discussion

Finally, we emphasise that equations (1.1–1.4) do not represent a specific estimation method for animal abundance. Instead, they provide a conceptual framework for thinking about abundance estimation problems. Indeed, all of the specific abundance estimation methods presented in the reviews by Seber (1982), Lancia et al. (1994), Williams et al. (2002) and Conroy and Carroll (2009) can be viewed as special cases that fall within this general framework. We also believe that this framework provides a reference basis to generate new methods tailored to the logistical and biological specifics of new estimation problems including the challenge presented by the need to monitor often low-density elephant populations across very large areas of inaccessible terrain.

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CHAPTER 2

Monitoring Needs, Resources and Constraints: Deciding Which Methods to Use

Simon Hedges

2.1 Introduction

We often hear the question: 'why count elephants?' The need for scientific monitoring of elephant populations arises from two broad considerations. First, information about elephant distribution and abundance and the trends in these parameters is needed to set appropriate management goals, to monitor the effectiveness of management interventions and policy-makers' decisions (e.g., whether ivory can be traded legally), to assess the impact of threats such as habitat loss and degradation and to inform local people and other stakeholders [Lindsey 1993; Blanc et al. 2003; Sutherland et al. 2004]. Clearly, gathering this type of information is the primary goal of wildlife managers and conservation agencies. A second, broader goal of elephant monitoring is to develop a body of empirical and theoretical knowledge that can potentially improve our predictive capacity to deal with new situations and thus increase the effectiveness of our conservation strategies. Such basic science also contributes to the general advancement of human knowledge. This task falls largely in the domain of academic studies carried out chiefly by wildlife biologists.

Given the threatened status of many elephant populations—and the substantial investments being made in elephant conservation—wildlife managers and conservation agencies need clear and reliable answers to some basic questions. Without these answers, they cannot begin to evaluate the success or failure of conservation efforts. Some of these basic questions are:

1. What are the geographic ranges and distributions of elephant populations?
2. Where are individual elephant populations increasing their ranges and where are these ranges fragmenting or shrinking?
3. For selected elephant populations, what are the population trends? In other words, are these selected elephant populations stable, declining or increasing?
4. What are the threats to elephants and their habitat in a site or landscape and how effective are law enforcement and other management interventions at reducing those threats?

The traditional approaches to answering these questions have too often been based on: (1) 'total direct counts' (censuses), counts at waterholes and 'block counts' [see, e.g., Bist (2003) for descriptions of these methods], all of which fail to address the critical issues of detectability and spatial sampling; consequently, the relationship between the count statistic and the true number of elephants is not known for those sites where these methods were used [Williams et al. 2002; Elphick 2008; Chapter 1]; (2) encounter rates of elephant sign and/or reliance on untested assumptions about sign (dung) production and persistence [see Buckland et al. (2001; 2004), Laing et al. (2003) and Chapter 4]; or (3) imprecise aerial surveys that do not always pay attention to detectability and thus return population estimates of questionable accuracy and of limited utility for monitoring population trends [Caughley 1974; Jachmann 2001; Whitehouse et al. 2001; Jachmann 2002; Chapter 8; see also Msoffe et al. 2010]. Use of uncorrected counts can result in erroneous conclusions about elephant population status and trend, leading to the misdirection of funds and overlooked conservation opportunities [Duckworth and Hedges 1998; Blake and Hedges 2004; Hedges 2006].

Fortunately, during the past three decades, there has been phenomenal progress in the methods used for wildlife population estimation. This progress is evidenced by the development and deployment of both new statistical models and new technologies [Burnham 2004]. Three important conceptual approaches to population sampling—distance sampling, capture–recapture sampling and occupancy sampling (which is related to capture–recapture sampling)—have all advanced particularly rapidly [Buckland et al. 2001; Williams et al. 2002; Buckland et al. 2004; MacKenzie et al.

2006; Bohning 2008]. In addition, hierarchical modelling methods have received a lot of attention and now provide a powerful framework for the analysis of data from capture–recapture and other sampling of populations, metapopulations and communities [Royle and Dorazio 2008; Link and Barker 2010].

Over the last 20 years, the authors of this manual, together with other collaborators, have been actively involved in developing or refining several sampling-based approaches to estimating population parameters of elephants and other species [Pollock et al. 1990; Karanth 1995; Karanth and Nichols 1998; Nichols et al. 2000; Powell et al. 2000; Walsh et al. 2001; MacKenzie et al. 2002; Pollock et al. 2002; Eggert et al. 2003; MacKenzie et al. 2003; Royle and Nichols 2003; Strindberg and Buckland 2004; MacKenzie et al. 2005; Royle et al. 2005; Stanley and Royle 2005; Hedges and Lawson 2006; Karanth et al. 2006; MacKenzie et al. 2006; Bailey et al. 2007; Goswami et al. 2007; Nichols et al. 2007; Runge et al. 2007; Nichols et al. 2008; Mackenzie et al. 2009; Royle et al. 2009; Thomas et al. 2010; Hedges et al. in review]. Furthermore, work by the authors and their range State government partners, as well as by others, has shown that high-quality population surveys using standardised peer-reviewed methods are possible in the forests and other elephant habitats of Asia [e.g., Hedges et al. (2005) in Indonesia; Hedges et al. (2007a; in review) in the Lao PDR; Hedges et al. (2007b) in Cambodia; Goswami et al. (2007) in India; Vidya et al. (2007) in Vietnam; Hedges et al. (2008) and Gumal et al. (2009) in Malaysia; Manopawitr et al. (2008) in Thailand]; in the forests of West and Central Africa [Barnes et al. 1991; Jachmann 1991; Barnes et al. 1994; Michelmore et al. 1994; Barnes et al. 1995; Barnes et al. 1997; Barnes and Dunn 2002; Eggert et al. 2003; Blake et al. 2007; Buij et al. 2007; Blake et al. 2008; Stokes et al. 2010; Yackulic et al. 2011]; and in the more open elephant habitat types of East and Southern Africa [e.g., Blanc et al. 2007: see various surveys listed within; Morley and van Aarde 2007; Martin et al. 2010].

As a result, it is now recognised that the methods we use for monitoring elephant populations can and should incorporate recent scientific advances. As an example of such recognition, the CITES Monitoring the Illegal Killing of Elephants (MIKE) programme has produced new guidelines and standards in an attempt to improve the traditional monitoring protocols for elephants [Craig 2004; Hedges and Lawson 2006]. However, neither of these manuals covers all the methods available to those needing to monitor elephant populations. Moreover, there have been significant advances in the last 5–6 years. We have, therefore, prepared this manual incorporating the new approaches with the hope that it will be useful for elephant biologists, reserve managers, conservation agencies and individual conservationists engaged in monitoring wild elephant populations.

2.2 The Sampling-based Approach to Monitoring

We recognise the need to prevent this manual from turning into an exercise in 'reinventing the wheel' by trying to review all established scientific methods of animal population estimation: there are several excellent general manuals that deal with these issues [Seber 1982; Bookhout 1994; Wilson et al. 1996; Thompson et al. 1998; Borchers et al. 2002; Williams et al. 2002; Conroy and Carroll 2009]. We expect the users of this manual to familiarise themselves with basic statistical methods and standard literature on population estimation. However, for the sake of completeness, brief overviews of current sampling-based population estimation methods are provided in this manual.

More importantly, as we saw in Chapter 1, the seemingly disparate approaches to counting animals are conceptually unified, under a rigorous theoretical framework established in the standard works of reference [Seber 1982; Thompson 1992; Lancia et al. 1994; Thompson et al. 1998; Williams et al. 2002]. Readers are directed to these sources for further detail.

Specifically, this manual covers the manner in which monitoring approaches deal with the core problems of spatial sampling and detectability. As you will recall from Chapter 1, spatial sampling concerns the frequent inability to use animal survey methods over an entire area of interest. In such cases, we need to survey some subset of the entire area of interest and then use these results to draw inference about the entire area. Detectability concerns the typical inability to detect and count all animals present in an area that is selected for survey. Regardless of the particular survey method, comparisons of resulting count statistics over time or space require consideration of the associated detection probabilities (the probability that an animal appears in the count statistic). Some approaches to animal monitoring permit direct estimation of these detection probabilities (these methods tend to entail the greatest requirements for effort and resources) and permit strong inferences, whereas others rely on strong assumptions about the absence of variation in these probabilities over time and/or space and typically yield weaker inferences. Focussing on the key features of animal monitoring will facilitate useful consideration and comparison of the various approaches suggested for monitoring elephants.

2.3 Defining Objectives

The first step in monitoring any elephant population is to define the objectives of the exercise that you want to undertake. These specific objectives are linked to the two broad monitoring goals we identified at the beginning of this chapter. Any monitoring programme can have one or more of the following specific objectives:

1. Mapping the distribution of elephants (e.g., at a site, landscape or regional/national scale).
2. Estimating trends in the distribution of elephants in order to understand whether the area occupied is stable, increasing or decreasing.
3. Assessing the threats to elephant populations and the trends in those threats.
4. Estimating the size or density of elephant populations (e.g., number of elephants/100 km^2) in sites of particular importance (e.g., key protected areas).
5. Estimating elephant population trends in order to understand whether populations are increasing or decreasing in selected sites or landscapes.
6. Estimating the vital rates of annual survival, recruitment and population change.

The methods available to meet these objectives for elephant monitoring programmes are summarised below (Table 2.1) and vary according to the spatial scale at which you need to work, the nature of the elephants' habitat and the likely size of the elephant populations of interest (Table 2.2). Please note that it will not always be possible to meet your desired objectives, for example, if the elephant population of interest is too small and patchily distributed across a large landscape.

2.4 Assessing Available Resources

The achievement of the objectives outlined above also depends on the time, manpower, technical skills and financial/material resources that are at your command. Therefore, assessing the resources available to you is the second important step of elephant monitoring. Usually, monitoring of elephants is carried out either by government agencies (e.g., forestry or wildlife departments), the staff of non-governmental agencies and universities that are conducting focussed research or surveys, or by consultants. The survey personnel available may vary greatly in terms of their technical skills and field abilities. Their numbers may range from a handful of highly trained scientists or naturalists to dozens or even hundreds of field personnel without scientific skills.

The skills required for carrying out field surveys of elephants also vary greatly. Good 'field skills', including the ability to walk long distances, observe carefully and recognise and record animals or their signs accurately, are of prime importance. People with such field skills may be wildlife biologists, wildlife department staff or local hunters/naturalists. Very different kinds of skills are necessary for designing the surveys and for analysing the data that result from the field surveys. We will call these 'analytical skills'. These skills include knowledge of population sampling methods, ability to organise and analyse field survey data and interpret the results correctly. It is critically important for you to assess the kinds of skills that are available in your situation.

Similarly, the material resources available for elephant monitoring vary depending on the local context. In most cases, particularly where wildlife or forestry departments carry out the elephant monitoring, only basic tools may be available. Such 'basic tools' may include maps of the area being surveyed, tape measures, machetes and the like. These tools are absolutely essential to carry out any monitoring programme. Sometimes, in addition to the basic tools, 'specialist tools' such as sighting compasses, topofils, range finders and global positioning systems (GPS) may be available. If these tools are not available, all or many of them will need to be purchased. Sometimes, even 'advanced tools' such as camera traps, radio telemetry equipment, computers and special software may also be available. Again, if they are not, then they may need to be purchased depending on the methods selected for the monitoring programme.

2.5 Decision-making: Matching Objectives and Resources

A successful monitoring programme depends on your being able to decide on a survey scheme that defines the achievable objectives carefully, based on available manpower, technical skills and material resources. Selecting wonderful goals based on wishful thinking is not useful. The goals must be realistic in your specific context; if not, the failure of your monitoring programme is almost guaranteed.

Once you have examined the resources available at each administrative unit/level (e.g., province or protected area), you can determine the type of monitoring that is feasible by matching the available resources, local ecological conditions and potential survey methods. The guidelines provided in this chapter and, especially, in Tables 2.1 and 2.2 should help you select the monitoring methods that you can employ cost-effectively and reliably.

If there is a serious mismatch between resources available and the objectives you hope to attain, you should not attempt to meet objectives 4–6. There is absolutely nothing wrong with this: monitoring of elephant distribution (and the threats to elephant populations) is a critically important first step in implementing any landscape level conservation programme. In such a case, we recommend that you initiate your survey efforts with objectives 1–3. Gradually, over the years, you can build up the capacity and resources to try to meet additional objectives if and as needed.

TABLE 2.1 *Deciding which methods to use depending on what you need to know*

What you need to know	*Which method(s) to use*	
	Site (≤ 5000 km²)	*Landscape (> 5000 km²)*
Elephant occurrence, range and distribution (occupancy)	Detection–non-detection survey, repeated to assess trend (Chapters 6 and 11).	Detection–non-detection survey, repeated to assess trend (Chapters 6 and 11).
Determinants (including habitat type/quality) of elephant occurrence, range and distribution	Occupancy surveys using elephant dung to assess detection–non-detection of elephants and covariate modelling to evaluate hypotheses for occupancy in relation to both human activity and ecological features (Chapters 6 and 11).	Occupancy surveys using elephant dung to assess detection–non-detection of elephants and covariate modelling to evaluate hypotheses for occupancy in relation to both human activity and ecological features (Chapters 6 and 11).
Elephant population density and abundance, and trends in density and abundance	Terrestrial sightings surveys using line transects, give density, which can be converted to abundance; repeated over time to assess trends (Chapters 3 and 7). Aerial surveys, repeated over time to assess trends (Chapter 8). Capture–recapture surveys (give abundance, which can be converted to density) using fecal DNA or, in a few places, sightings or camera/video traps; repeated over time to assess trends (Chapters 5 and 10). Dung count surveys using line transects to estimate dung density plus dung decay rate estimation plus defecation rate estimation (or use of appropriate defecation rate data from another study); gives density, which can be converted to abundance; repeated over time to assess trends (Chapters 3, 4 and 9).	For non-concealing habitat types, aerial surveys, repeated over time to assess trends (Chapter 8). For very large areas of concealing habitat types such as forests (> c. 25,000 km²), use two-phase sampling and modelling in a Bayesian framework. In the first phase, occupancy is estimated by surveys to detect elephant sign (e.g., dung piles) in all selected sites in the landscape, where selection may be of all sites available, or a random sample of sites. In the second phase, if a detection threshold is achieved, capture–recapture sampling is conducted to estimate abundance. Detection and capture–recapture data are then used in a joint likelihood to model probability of detection in the occupancy sample via an abundance–detection model. Capture–recapture modelling is used to estimate abundance for the abundance–detection relationship, which is used to predict abundance at the remaining sites, where only detection data were collected. Repeated over time to assess trends (Chapters 6, 11 and 13). For intermediate sized areas (> 5000 km²) and very large areas (> c. 25,000 km²) of concealing habitat types (e.g., forest), experiment with marked sign (dung pile) counts and two visits per transect so as to remove the need for pre-survey dung decay monitoring and, if this method is successful, use the effort and money saved to facilitate a multi-scale stratified survey across the landscape. Selected sites of significant elephant abundance and/or those that are also important for management, e.g., protected areas or MIKE sites, can be more intensively surveyed and treated as separate strata in the analysis. Repeated over time to assess trends (Chapter 4).

What you need to know	*Which method(s) to use*	
	Site (≤ 5000 km²)	*Landscape (> 5000 km²)*
		For intermediate sized areas (5000 km² to a maximum yet to determined) of concealing habitat types (e.g., forest), consider experimenting with dung counts and rainfall models of the dung decay process. Repeated over time to assess trends (Chapters 3, 4, 9 and 13).
Demographic parameters: survival rates, emigration rates, movement or transition rates, fecundity, population growth rates	Capture–recapture surveys using fecal DNA or, in a few places, sightings or camera/video traps; repeated over time to assess trends (Chapters 5 and 10).	Abundance–occupancy relationships from distributional surveys, to understand and document large-scale population dynamics and the consequences of environmental change (Chapters 6 and 11).
Abundance and distribution of threats	Patrol-based data collection; dedicated survey based data collection in conjunction with, e.g., transect-based surveys for elephant sign (Chapter 12).	Patrol-based data collection; dedicated survey based data collection in conjunction with, for e.g., occupancy surveys for elephant sign (Chapter 12).

TABLE 2.2 *Summary of key requirements, advantages and disadvantages of the recommended survey and monitoring methods*

Method	*Requirements*	*Key advantages*	*Key disadvantages*
Occupancy (detection–non-detection) surveys for elephant sign (Chapters 6 and 11)	• Independent repeat surveys can be conducted in grid cells or some other defined sampling unit, ideally over a short period of time	• Can be used when elephants cannot be seen readily (because they occur in concealing vegetation types such as rainforest)	• Choice of sampling unit (e.g., grid cell) size needs to be guided by knowledge of likely home range size in order to distinguish true occupancy from use, but home range sizes for elephants—especially forest elephants-are not well known
Terrestrial sightings surveys using line transects (Chapters 3 and 7)	• Elephants can be seen readily (because they occur in non-concealing vegetation types) • Elephants do not move away (or towards) the observers in response to the observers' movements before the observers have detected the elephants	• Can be a cost-effective method to estimates density/abundance for medium to large populations • Can provide data on population sex- and age-structure	• Difficult if terrain hinders following a straight line • Not cost-effective if population size is very small (a few 10s of elephants) as effort required to achieve tolerable precision will be too high
Aerial surveys (Chapters 3 and 8)	• Elephants can be seen readily (because they occur in non-concealing vegetation types)	• Allows relatively quick/efficient coverage of large areas • Can provide data on population sex- and age-structure • Can provide data on abundance and distribution of elephant carcasses (and carcass: live animal ratios)	• Access to a suitable airplane and appropriately qualified pilot/observers maybe an issue • Can be expensive • Can return imprecise estimates if elephant encounter rates are low • Tends to produce underestimates of elephant abundance due to imperfect detection on the transect line (distance sampling) or in the sampling unit (strip transects)

Method	*Requirements*	*Key advantages*	*Key disadvantages*
Capture–recapture surveys using fecal DNA (Chapters 5 and 10)	• Total elephant population size is likely to be less than a few thousand animals (above this size a very large number of samples would have to be collected and analysed making the cost prohibitive)	• Provides detailed data for each animal 'captured' (can be particularly helpful in situations where illegal killing is biased towards, e.g., adult males) and, depending on study design, allows estimates of survival rates, emigration rates, movement or transition rates, fecundity and population growth rates (so, more informative than dung counts) • Can be used when population size is too small (a few 10s of elephants) for terrestrial sighting-based surveys using line transects to be cost effective • Can be used when elephants cannot be seen readily (because they occur in concealing vegetation types such as rainforest) • Can be used when it is not possible to estimate dung disappearance (decay) rates for the site • Can be used when no appropriate defecation rate data are available • Should return a more precise estimate of population size than dung count based methods • Less time-consuming than dung count based methods (because no pre-survey dung decay estimation required) • Cost is likely to be lower than dung count based methods	• Access to a suitable laboratory and appropriately qualified staff maybe an issue (there are currently relatively few laboratories set-up for fecal DNA analysis and very few in elephant range States so the need to export samples may also be a problem) • Fresh elephant dung maybe difficult to find in some areas (but use of detection dogs may be helpful)
Capture–recapture surveys using direct sightings or camera/video traps (Chapters 5 and 10)	• A good 'network' of clearings, waterholes, etc. exists at which sightings can be obtained or camera traps can be positioned so as to obtain reliably whole-body photographs of elephants	• Provides detailed data for each animal 'captured' (can be particularly helpful in situations where illegal killing is biased towards, e.g., adult males) and, depending on study design, allows estimates of survival rates, emigration rates, movement or transition rates, fecundity and population growth rates (so, more informative than dung counts along line transects) • Can be used when population size is too small (a few 10s of elephants) for terrestrial sighting-based surveys using line transects to be cost effective	• Effort to precision ratio typically high compared to terrestrial sighting-based surveys using line transects and so capture–recapture field costs are typically higher • Few places are suitable: in most forested areas obtaining whole-body shots of elephants with sufficient frequency will be impossible; in more open areas direct sighting based methods (terrestrial or aerial surveys) are likely to be more appropriate • Camera traps are expensive, subject to problems with humidity and liable to be stolen in some areas

Method	*Requirements*	*Key advantages*	*Key disadvantages*
		• Can be used when elephants cannot be seen readily during aerial surveys or along terrestrial sighting transects (and/or move away in response to observers before they are detected) • Can be used when it is not possible to estimate dung disappearance (decay) rates for the site • Can be used when no appropriate defecation rate data are available • Should return a more precise estimate of population size than dung count based methods • Less time-consuming than dung count based methods • Cost may be lower than dung count based methods	
Dung count surveys using line transects (Chapters 3, 4 and 9)	• Ideally, dung pile encounter rates along transects should be >1/km • Dung disappearance (decay rates) can be estimated in a spatially unbiased manner for the whole site over the period leading up to and including the dung count survey • Appropriate defecation rate data are available	• Can be used when elephants cannot be seen readily (because they occur in concealing vegetation types such as rainforest) • Can also provide data on population age-structure if dung dimensions are recorded • Can return more precise estimates than aerial or terrestrial sighting-based surveys aerial surveys because the sighting-based surveys record the instantaneous distribution of elephants, and the variation between transects is usually high, often giving estimates with wide confidence limits	• Time-consuming, particularly because of the need to begin dung decay rate monitoring many months before the survey • Obtaining spatially unbiased dung disappearance rates for the whole site can be prohibitively difficult (because it requires very high effort levels) • Estimating dung density is problematic if significant areas of the site are seasonally (or permanently) inundated • Difficult if terrain hinders following a straight line • Labour-intensive and thus, likely to be expensive

We emphasise that if you really want to estimate parameters such as population size, absolute densities and survival and recruitment rates, you have to employ the advanced methods described in Chapters 3, 4 and 5. However, we note that these are methods are expensive and often extremely time-consuming. Such advanced techniques cannot be applied for routine population surveys over the entire distributional range of any elephant species. The advanced methods are relevant for monitoring elephants over a small fraction of the species' vast ranges. On the other hand, the two most critical needs of elephant monitoring in much of Asia and Africa—mapping spatial distributions and levels of threat over large regions and determining population trends in key areas—are widely attainable goals using the relatively simple methods outlined in Chapters 8, 9, 10, 11 and 12.

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CHAPTER 3

Distance Sampling along Line Transects: Statistical Concepts and Analysis Options

Samantha Strindberg

3.1 Introduction

The method of line transect sampling belongs to a family of density estimation approaches collectively known as distance sampling (see Glossary). Line transect surveys can be used to estimate absolute densities of elephants that occur in fairly open habitats where it is possible to observe the individual animals, and densities of elephant dung piles (as detailed in Chapters 4 and 9) that can be transformed into absolute densities of elephants by estimating and correcting for dung decay and deposition rates. During line transect surveys, observers traverse a series of transect lines recording either all objects within a set distance w from the line (strip transect sampling) or perpendicular distances to all observations (line transect distance sampling). During strip transect surveys it is assumed that all animals/dung piles within distance w of the line are counted and elephant densities are calculated by dividing the total number observed by the area surveyed (aggregate transect length multiplied by twice the width w). Distance sampling does not make this assumption; instead the probability of detecting an animal/dung pile is modelled as a function of the observed perpendicular distances and combined with the animal/dung pile encounter rate and estimated group size (only for direct observations of elephants) to calculate the elephant density in the study area [Buckland et al. 2001].

To understand how line transect surveys fit into the overall statistical framework introduced in Chapter 1, let us revisit the canonical estimator introduced in the same chapter that was expressed as:

$$\hat{N} = \frac{C}{\hat{p}\alpha} \quad (3.1)$$

where $\hat{N}$ is estimated abundance, C is the number of objects counted, $\hat{p}$ is the estimated proportion of objects on the sampled plots that is detected and α is the proportion of the study area surveyed; $\hat{p}$ relates to detectability and α relates to spatial sampling.

For very small study areas (usually in the order of 100 km^2), if the habitat is open, a census count could be conducted, where each elephant in the geographic area of interest is recorded. In this case, there is no statistical estimation involved; $N=C$ and the density is simply obtained by $D=N/A$, where A denotes the surface area of the study region. Clearly, great care must be taken to ensure that every animal is recorded in the area of interest and no double counting takes place, which can be very difficult to achieve. Given that this often requires a leap of faith in practice, we do not recommend this approach (see also Chapter 1).

For a strip transect survey, only a portion of the area of interest is covered and we assume that all the animals/dung piles within the sampled area are counted and thus $\hat{p} = 1$. If L is the total length of all the transects, then for a strip transect we can conceptualise the area surveyed as a long, thin rectangle of width $2w$ and length L. Thus, the area surveyed is $2wL$ and the proportion of the study region with area A surveyed is $\alpha = 2wL/A$. Consequently, equation (3.1) can be written as

$$\hat{N} = \frac{AC}{\hat{p}2wL}. \qquad (3.2)$$

Density and abundance are related as $N = D \times A$, so the equation can be rewritten as

$$\hat{D} = \frac{C}{\hat{p}2wL}. \qquad (3.3)$$

With strip transects, unbiased density or abundance estimation relies on meeting the fundamental assumption that all objects of interest within the sampled area are counted. To avoid violating this assumption, the width w of the strip is usually fairly narrow. Hence, for animals, such as elephants, that frequently occur in groups which are sparsely distributed across large geographic regions, this can be a very inefficient sampling method. However, there have been recent advances in aerial survey methods with the use of high-resolution imagery instead of human observers. This new technology makes it conceivable that at some point in the future, for elephants in open habitats, the fundamental assumption of strip transects may be met and their efficiency improved. However, until then, another issue that arises for human observers is the difficulty in determining where the strip boundary falls and thus which animals fall inside the sampling units for groups that straddle the edge of the strip. This usually leads to positive bias, as there is a tendency to include more animals than should be counted. It is extremely difficult to ensure that detection is certain within the survey strip, especially when counting dung piles. Hence, it is recommended that an approach, such as distance sampling, that permits estimation of detectability be used.

Distance sampling along line transects can be thought of as a generalisation of strip transect sampling, where the assumption that all objects in the strip are counted is relaxed. Although the method does not assume that all objects in the vicinity of the transect line are observed, it does assume that objects on the line itself are certain to be observed. Distance sampling permits the use of a much wider strip, an infinite strip width potentially, which avoids the need to determine exactly where a strip edge lies for those observations that would otherwise straddle the boundary of the sampled area. Instead, the perpendicular distances to the animals/dung piles that are detected are recorded (Figure 3.1). For direct observations of elephants, if it facilitates the survey process or provides more accurate data, other measurements are recorded instead and converted to perpendicular distances during analysis of the data. Note that ideally, for elephants, only visual detections are recorded and aural detections are not recorded (or at least filtered out prior to analysis), because the distance measurements to aural detections tend to be inaccurate (introducing potential bias) and the detection process differs (introducing imprecision). However, small sample sizes with visual detections may force the use of aural detections. During analysis, a detection function is fitted to these distances, which allows us to estimate the proportion of animals/dung piles in the sampled area that are detected ($\hat{p}$). If sufficient data are available, different detection functions may be calculated for visual and aural detections. Once this quantity has been estimated for distance sampling along line transects, we can then estimate density and also animal/dung pile abundance if A is known using equations (3.3) and (3.2), respectively.

Given that elephants often occur in groups, it is the group that constitutes the observational unit during sighting-based distance sampling along line transects, and the perpendicular distance to the centre of the group is

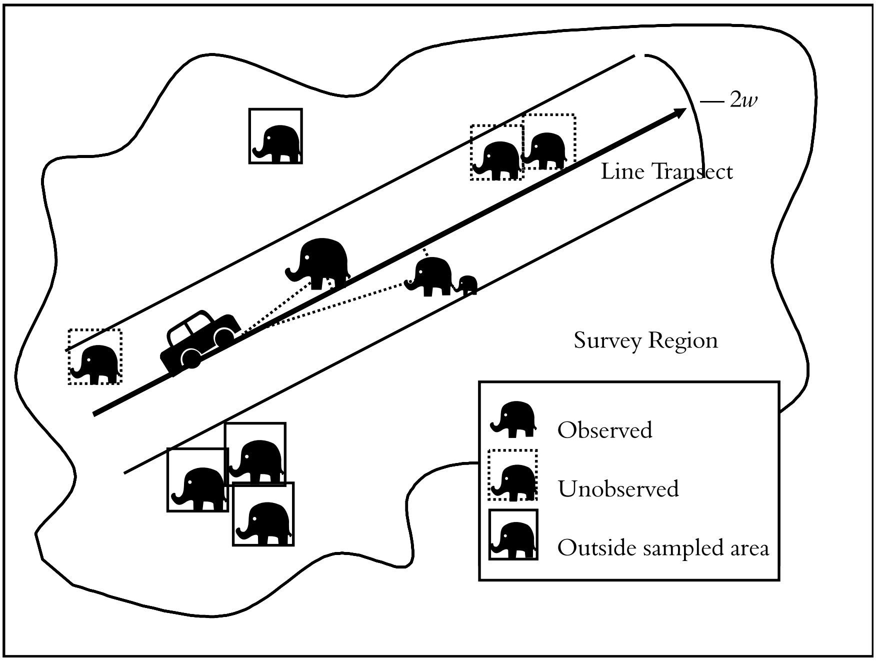

Figure 3.1 *During a line transect based distance sampling survey, observers move along line transects while recording the perpendicular distance to all animals/dung piles from the line. For direct observations of elephants, the unit of observation is the group and perpendicular distance to the centre of the groups is recorded (or other measurements to the centre and the perpendicular distance is calculated subsequently); group size is also recorded. Some of the elephant groups/ dung piles are seen by the observers, while others are not; some fall outside the sampled area—the sub-region at a distance* w *from the line. Methods are used to correct for these elephants missed by the observer. Note that surveys for elephants may be terrestrial or aerial; the former is used for dung piles.*

required. Groups are usually referred to as 'clusters' in distance sampling literature. With the elephant group as the object of interest, we additionally need to record the group size. Let $E(s)$ denote the expected group size. Equation (3.3) can be used to estimate the density of elephant groups in the study area. To obtain an estimate of elephant density, this equation is multiplied by the estimated expected group size $\hat{E}(s)$ as follows:

$$\hat{D} = \frac{C\hat{E}(s)}{\hat{p}2wL} \qquad (3.4)$$

As mentioned elsewhere in this manual, information on sex and age structure is valuable for monitoring purposes. Thus, during direct observation distance sampling surveys, not only group size, but also information on age and sex should be recorded, if possible. Similarly, during dung pile surveys the circumference of the three largest intact boli (all, if fewer than three intact boli remain in the dung pile) should be measured to allow for elephant age structure analysis (see Chapter 4 for more details).

Given the relative simplicity of density or abundance estimation from strip transect surveys and the more complex nature of the analysis if we cannot assume that all objects in the strip are detected, the remainder of this chapter will deal solely with distance sampling along line transects. We will focus on (i) the theory behind the method, (ii) the critical assumptions underlying density or abundance estimation with distance sampling and the potential biases that arise if these are violated, (iii) survey design options with an emphasis on reducing variability and other alternatives for improving estimator precision and (iv) data analysis with a short overview of the DISTANCE software [Thomas et al. 2010], which is the standard for distance sampling survey design and analysis. This chapter only touches upon the basics of line transect distance sampling and readers should consult the references for a more thorough description.

3.2 LINE TRANSECT DISTANCE SAMPLING THEORY

In the introduction, we saw how distance sampling fits into the generic framework for estimating density and abundance, which involves taking into account detectability and also spatial sampling. Line transect distance sampling was presented as a generalisation of strip transects, where it is not assumed that all objects of interest are seen. Given the observations made during the distance sampling survey, the density estimate can be obtained by taking account of those objects missed and the proportion of the study area sampled. In this section, we outline in more detail how density estimates are obtained and how the associated variances are estimated.

Prior to conducting the line transect survey, k transect lines of length $l_1,\ldots, l_k$ (with total length $L=\sum_{j=1}^{k} l_j$) are randomly located within the study area (survey design is covered in more detail in Section 3.4). During the survey, observers traverse these lines and record all observations of elephant groups/dung piles (for our purposes a 'group' can be an individual animal or a larger number of animals where group membership is ideally defined in the field protocol before the survey in terms of the spatial or other characteristics requisite for individuals to constitute a group). Observers record data necessary to obtain the perpendicular distance to the centre of the group and the group size. Sometimes, only observations at a distance w from the line are recorded, but more frequently all observations are recorded and w is set during analysis (which is covered in more detail in Section 3.5).

Line transect distance sampling surveys for elephants can be conducted from an airplane, from a vehicle, from elephant back, or on foot; surveys for dung piles need to be conducted on foot. Depending on the type of survey, different kinds of measurements will be taken to calculate the perpendicular distance x from the object of interest to the transect line during analysis; generally, for an aerial survey, a clinometer reading to obtain the angle of declination ϕ* to the centre of the elephant group as it passes abeam is taken (where 0° is at the horizon and 90° is directly below the aircraft) and the altitude of the airplane h is recorded; for a terrestrial survey with direct sightings of elephants, it is more usual to obtain a radial (sighting) distance r and sighting angle θ[1] (Figure 3.2). For the aerial survey, $x = h/\tan\phi$ and for the terrestrial survey, $x = r\sin\theta$. For dung piles, perpendicular distances are measured directly (see Chapter 9).

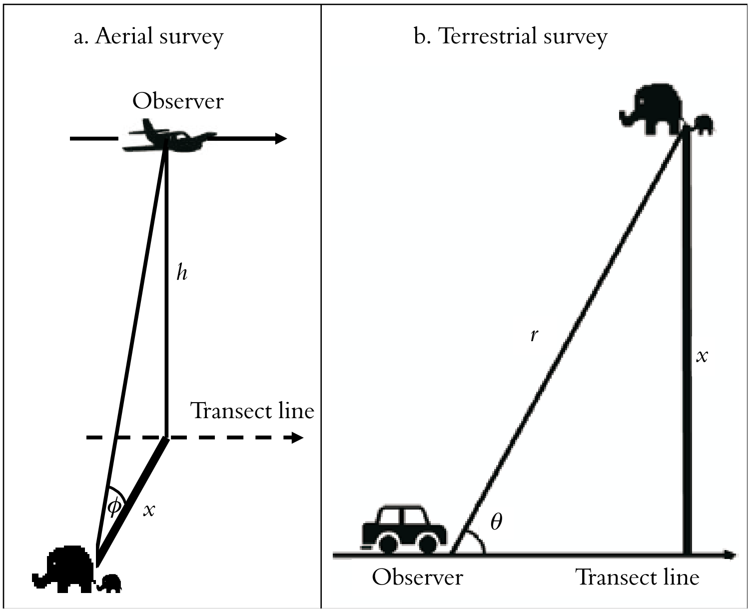

Figure 3.2 *Perpendicular distance x to the centre of the elephant group is calculated (a) from the altitude of the airplane h and angle of declination ϕ by applying the formula $x = h/\tan\phi$, for an aerial survey (note that the alternate interior angle has the same degree measurement as the angle of declination), and (b) from the radial (sighting) distance r and sighting angle θ by applying the formula $x = r\sin\theta$, for a terrestrial survey conducted by vehicle, from elephant back, or on foot. See Figure 9.3 for a depiction of measuring perpendicular distance to the centre of a dung pile.*

* angle between the horizon and the sighting

[1] The sighting distance is the distance from the observer to the centre of the group of animals; the sighting angle is the angle between the transect line and an imaginary line drawn between the observer and the centre of the group.

As distance from the transect line increases, the number of observations drops off, as the animals/dung piles are more likely to be obscured by vegetation; in the case of animal surveys, acoustic cues that may lead to visual contact diminish. Thus, even though there is no decrease in the expected number of animals/dung piles found at increasing distance from the transect line, if we plot a histogram of the number of detections against distance from the line, we see fewer detections at larger distances (Figure 3.3a). During a standard distance sampling analysis, a detection function is fitted to the perpendicular distances for each observation to estimate the proportion of animal groups/dung piles in the surveyed strips that are counted (p). We define a detection function $g(x)$ which gives the probability of detecting an object, when it is at a distance x from the line, where x is between 0 and w. As the distances from the line increase, the probability of detection decreases. We can conceptualise this by considering the detection curve that has been fitted by eye to the data in Figure 3.3a. To estimate p, the area under the curve corresponding to the observed number of objects is divided by the area of the rectangle corresponding to the expected number of objects. In this example, the area under the curve equals 27.25 and the expected number of objects is given by the area of the rectangle $1.5 \times 25 = 37.50$. Hence, the proportion of animal groups/dung piles counted $\hat{p} = 27.25/37.50 = 0.7676$.

In practice, when estimating p by means of the detection function $g(x)$, the observations are rescaled so that $g(0) = 1$ (i.e., detection on the transect line is certain) and a parameter μ equal to the area under the curve of $g(x)$ is defined by

$$\mu = \int_{x=0}^{w} g(x)dx.$$

The parameter μ is referred to as the effective strip half-width and is the half-width of the strip extending either side of the transect line such that as many animal groups/ dung piles are seen beyond the strip as are missed within it. This effective strip width allows us to estimate the proportion of the study area being sampled. As can be seen from Figure 3.3b, the proportion of animal groups/dung piles detected and counted

$$\hat{p} = \frac{\int_{x=0}^{w} \hat{g}(x)dx}{1.0 \times w} = \frac{\hat{\mu}}{w}.$$

Thus, the canonical formula shown in equation (3.4) can be rewritten as:

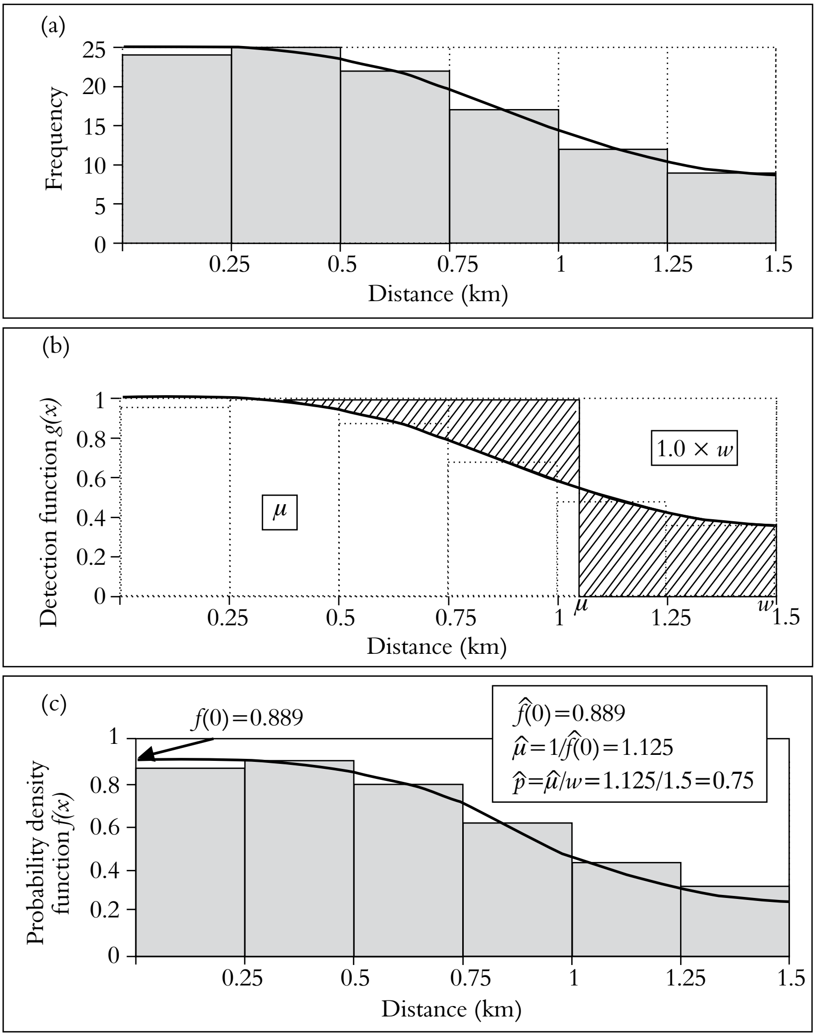

Figure 3.3 *(a) An example of plotting distance sampling observation data in 6 distance intervals, with the tendency for fewer objects to be detected as distance from the line increases. The dashed lines show the expected number of objects in each interval. The curve has been fit to these data by eye and the area under the curve represents the number of observations that were made, while the area above the curve represents the missed observations. Thus, the proportion of observations counted is equal to the area under the curve divided by the total area, i.e.,* $\hat{p} = 27.25/(1.5 \times 25) = 27.25/37.50 = 0.7676$ *approximately. (b) A detection function is defined such that* $g(0) = 1$, *and the effective strip width* μ *is the distance at which as many objects are seen beyond* μ *as are missed within* μ. *The proportion of objects detected and counted* $\hat{p} = \int_{x=0}^{w} \hat{g}(x)dx \,/(1.0 \times w) = \hat{\mu}/w$.

Note that μ *has the interesting characteristic of representing both a distance and an area measurement. (c) The DISTANCE software can be used to estimate* μ *by fitting a probability density function f(x) of perpendicular detection distances to the data to obtain* $\hat{\mu} = 1/\hat{f}(0)$.

$$\hat{D} = \frac{C\hat{E}(s)}{\frac{\hat{\mu}}{w}2wL} = \frac{C\hat{E}(s)}{2\hat{\mu}L} \qquad (3.5)$$

To estimate μ, a probability density function (pdf) of perpendicular detection distances $f(x)$ is used where $f(x)$ is the detection function and $g(x)$ is rescaled so that the area under the function equals one, i.e., $f(x) = g(x) / \mu$, so that

$$\int_{x=0}^{w} f(x)dx = \frac{1}{\mu}\int_{x=0}^{w} g(x)dx = \frac{\mu}{\mu} = 1.$$

In addition, since we assume $g(0) = 1$, it follows that the pdf evaluated at $x=0$ is given by $f(0) = 1/\mu$, which is why equation (3.5) is often written as follows:

$$\hat{D} = \frac{C\hat{f}(0)\hat{E}(s)}{2L}. \qquad (3.6)$$

So, estimating p involves modelling the pdf of the perpendicular distances and evaluating the fitted function at $x=0$. Estimating p is reduced to modeling the pdf of perpendicular detection distances $f(x)$ because fitting a pdf is a standard, well researched statistical problem. The DISTANCE software is the custom-built dedicated application used to do this type of model fitting and is further described in Section 3.5 [Thomas et al. 2010].

To convert the elephant group density to animal density, we also need to estimate the expected group size $E(s)$. If large and small groups are equally visible at any distance from the transect line, $E(s)$ can simply be estimated by taking the mean of the observed group sizes. Frequently, this is not the case, as large groups tend to be more visible, especially as distance from the line increases. This phenomenon leads to size bias, because large groups are over-represented in the sample. There are a number of approaches for dealing with this type of size-biased sampling where the detection probability is a function of both distance from the observer and group size—they are briefly covered in the next section.

The variance of the density estimate $\text{var}(\hat{D})$ can be approximated using the delta method. Assuming that correlations between the components of the estimation are zero, the variance for the generic estimator of $\hat{D}$ for line transect sampling is given by

$$\hat{\text{var}}(\hat{D}) = \hat{D}^2 \left\{ \frac{\hat{\text{var}}(C)}{C^2} + \frac{\hat{\text{var}}(\hat{f}(0))}{[\hat{f}(0)]^2} + \frac{\hat{\text{var}}(\hat{E}(s))}{[\hat{E}(s)]^2} \right\}. \qquad (3.7)$$

The variance of the counts among transect lines, while taking into account the potentially different line lengths, is usually used to estimate the variance in number of animal groups seen var(C) [Buckland et al. 2001: pp. 78–79, 108–109, 154–155; Fewster et al. 2009]. To obtain a reliable estimate of this variance at least 15–20, but preferably more than 25 replicate lines are required. A less than ideal solution that can be applied when there are insufficient transect lines to estimate var(C) empirically, is to estimate this variance by assuming that it is proportional to the expected value of C. As the detection function is fitted to the distance data, a likelihood-based estimate of var($\hat{f}(0)$) is produced. The approach for estimating $E(s)$ will determine how var($\hat{E}(s)$) is estimated [Buckland et al. 2001: pp. 72–74, 120, 164]. If the components comprising equation (3.7) are correlated, then a non-parametric bootstrap can be used to estimate var($\hat{D}$). This statistical technique does not require any distributional assumptions and involves randomly resampling the data to obtain a large number of estimates of D from which var($\hat{D}$) is then estimated. The resamples are taken at the level of the transect line, as these are considered to be independent, rather than at the level of the observation [Buckland et al. 2001: pp. 82–84, 117, 161–164].

When considering precision of a density estimate, it is convenient to use the coefficient of variation (CV), where $\mathrm{CV}(\hat{D}) = \sqrt{\mathrm{var}(\hat{D})}\,/\hat{D}$. CV($\hat{D}$) gives the size of the standard error of the density estimate $\sqrt{\mathrm{var}(\hat{D})}$ relative to the size of the estimate $\hat{D}$. As a unit-less quantity it can be used to compare different studies that may use different units or have very different estimates of density or abundance. Thus, equation (3.7) can equivalently be written as follows:

$$\widehat{\mathrm{CV}}(\hat{D}) = \sqrt{[\widehat{\mathrm{CV}}(C)]^2 + [\widehat{\mathrm{CV}}(\hat{f}(0))]^2 + [\widehat{\mathrm{CV}}(\hat{E}(s))]^2} \qquad (3.8)$$

Burnham et al. (1987: pp. 211–213) showed that log-based confidence intervals give a better measure of the precision of $\hat{D}$ than the standard symmetrical 95% confidence intervals. Thus, the approximate asymmetric 95% confidence intervals are given by $\hat{D}/C$ and $\hat{D}.C$ where $C = \exp\{1.96\sqrt{\hat{\mathrm{var}}\,[\ln(\hat{D})]}\}$ with $\hat{\mathrm{var}}(\ln(\hat{D})) = \ln\{1 + [\mathrm{CV}(\hat{D})]^2\}$. In practice, the 1.96 that corresponds to a normal distribution is generally replaced by another constant [Buckland et al. 2001: pp. 77–88, 118–119] and this is what is used by the DISTANCE software.

3.3 ASSUMPTIONS OF LINE TRANSECT DISTANCE SAMPLING AND BIASES THAT ARISE WHEN ASSUMPTIONS ARE VIOLATED

In this section, we look at the five critical assumptions underlying distance sampling along line transects and consider the biases that are introduced if these assumptions are not met. These assumptions are covered in considerably more detail in [Buckland et al. 2001: pp. 29–37, 130–133].

3.3.1 An adequate number of line transects are located randomly with respect to the distribution of the animals/dung piles

By locating the line transects according to a well-defined survey design, there is no need to assume that animals/dung piles in the population being sampled are randomly distributed in the study area (an assumption that is unlikely to be true). Random placement of an adequate number of line transects (the exact number depends on the variability in elephant group/dung pile density over the region of interest; 25 replicate lines is a reasonable recommendation, but sometimes 15–20 lines may suffice) by means of a survey design algorithm helps ensure valid statistical inference on two levels:

One can reliably extrapolate from observations made during the survey in the sampled area to the entire study area. This relies on the assumption that the surveyed lines are representative of the study area as a whole, i.e., randomness and sufficient replication ensures that lines pass through areas with densities representative of the entire region of interest rather than some smaller set of areas with possibly atypical densities.

One can reliably extrapolate from the observed distances to estimate the proportion of objects counted (p)*. This relies on the assumption that all animal groups/dung piles in the surveyed strips are uniformly distributed in the interval* $[0, w]$.

Without a random design, one needs to assume that animal groups/dung piles in the population are randomly distributed, which is unrealistic, or one has to resort to model-based inference, which relies on the possibility of fitting an unbiased model to the survey data (see Section 3.4.6). Thus, the simplest and most robust option is to use a random survey design. In Section 3.4, we focus on various survey design options.

If the assumption of random transect placement is violated, then the resulting density estimates have the potential to be either positively or negatively biased. For example, if the transects are placed along or in the vicinity of trails and elephants use those trails preferentially, elephant density and abundance will be overestimated, if applied to a larger region that may also contain sub-regions with lower elephant densities. On the other hand, elephants will avoid trails frequented by people, especially if they engage in elephant poaching. In this case, elephant density and abundance will be underestimated. With an insufficient number of randomly placed lines, the potential exists to sample only areas with atypical densities by chance. In addition, inadequate replication leads to poor estimates of precision. If it is suspected that elephants are reacting to the transect lines, then a separate dung survey at set distances from the lines that seem to be most affected can be conducted to investigate the degree of avoidance or attraction.

Another important consideration concerns the probability of sampling a particular location (referred to as the coverage probability) for a given type of random survey design. Ideally, every location in the study area should have the same probability of being sampled, if the standard analysis technique is to be applied. Those types of designs where the coverage probability is variable have the potential to produce biased estimates. If standard methods are applied during the data analysis phase and coverage probability is assumed to be even when it is not, and if high (low) density areas were sampled more intensively, this would lead to a positive (negative) bias. If the differences in coverage probability are extreme then it may be advisable to use an estimator that takes account of this, such as the Horvitz-Thompson estimator. However, this type of estimator is likely to increase the variance of the estimate [Strindberg 2001; Strindberg and Buckland 2004].

3.3.2 Groups of animals/dung piles whose centres are on or very near the line are detected with certainty

For distance sampling along line transects, the derivation of the density estimator is based on the assumption that all animal groups/dung piles are detected at zero perpendicular distance from the line, i.e., that $g(0) = 1$. If this assumption does not hold because animal groups/dung piles whose centres are on or very near the line are missed, then estimates of density or abundance will be negatively biased as the proportion of animal groups/dung piles counted, p, will be underestimated. When only the assumption of perfect detection on or near the line fails, the negative bias is a simple function of the proportion of objects on the line that are missed. For example, if 20% of groups with centre on or near the line are missed then the estimate of animal density or abundance will be 20% too low. In particular, when dealing with elephant groups, this assumes no movement, so that the entire detection function is effectively scaled by 0.8. Depressed $g(0)$ could also occur due to evasive movement, but this would not scale the entire detection function by 0.8. In reality, several assumptions may fail at the same time.

The assumption of perfect detection on or near the line can be relaxed in either one of the following cases:

- If detection of objects of interest at a short distance away from the line is certain;
- If it is possible to estimate the proportion of sighted objects of interest on the line.

The first case can be applicable to aerial surveys when observers cannot see the line nor potential animals beneath the aircraft due to the characteristics of the aircraft, for example. If animal groups/dung piles on (or at a short distance from) the line are not detected with certainty, then the proportion missed has to be estimated to avoid negatively biased estimates. Detection may be less than certain either because (a) observers simply miss animal groups/dung piles on the line (perception bias) or (b) some animals are unavailable for detection some of the time (availability bias), e.g., elephants obscured by dense vegetation during aerial surveys. In the former case, the best solution is to train observers well and improve the survey protocol. Issues that need to be considered include the experience of the observers, using more than one observer to cover the line, using better optical aids, and the speed at which observers travel along the transect line. Aerial surveys in particular require careful consideration in this regard as the speed of the airplane provides greater potential for missing animals (whether or not the airplane type allows observers to see the transect line); high-tech options such as video cameras can be used to allow for detections on or near the line to be noted after the survey. For both perception and availability bias, methods to estimate the true value of $g(0)$ can be applied [Laake and Borchers 2004]. These methods require that the study area is surveyed by two independent sets of observers more or less simultaneously and are thus more complex and costly to implement and analyse (the DISTANCE software supports the analysis of double-observer distance sampling data by means of its Mark–Recapture Distance Sampling (MRDS) analysis option).

3.3.3 Animals are detected at their initial location

If animals systematically move towards or, more typically, away from the observers, and such responsive movement takes place before the animals are detected, then estimates of density or abundance will be positively or negatively biased, respectively. In the former case, there are too many detections near the line, causing a 'spike' in the data at zero distance, and in the latter case too many a small distance away from the line (more likely for elephant surveys). See Figure 3.4a and b for examples of data showing responsive movement.

It is difficult to obtain reliable estimates of density if there is responsive movement and it is better to have field procedures that minimise disturbance as observers move along line transects. Responsive movement may be less of a problem for aerial surveys, due to the speed of the observers relative to the animals (ideally observers should be moving at least twice as fast as the animals), than for terrestrial surveys. For terrestrial line transect surveys, the problem can be avoided if observers look far enough ahead to detect animals before their presence causes a reaction in the animals; ideally, observers should obtain the distance and angle measurements (or declination angle and aircraft height measurements for aerial surveys) before the animals move. Unless animals that were visible disappear into the vegetation, an estimate of the group size may be easier to obtain once the animals respond.

In line transect surveys, slow non-responsive movement of the animals relative to the speed of the observers is generally not problematic.

3.3.4 Measurements from the line to the centre of each detected animal group/dung pile are exact

For a terrestrial elephant survey the radial distances and sighting angles should be recorded correctly and without measurement error; similarly, for the clinometer and altitude readings taken during an aerial survey. It is especially important that distances to objects near the line transect can be calculated both precisely and accurately. A prerequisite for this is that the location of the line is clearly defined and known to the observers. Random measurements errors that are not too large generally still allow for a reliable estimate of density or abundance, especially if the sample size is large.

If distances to observations are rounded to convenient values (see Figure 3.4c), then it is possible to deal with this 'heaped' data during analysis by grouping data into distance intervals, where cut points for the intervals are chosen so that heaps fall approximately at the midpoints of the intervals. However, not much can be done about systematic bias in distance measurements (unless bias correction factors are estimated by means of experiments); with consistently overestimated/underestimated distance measurements densities will be underestimated/overestimated. This is also the reason why observers should measure the distance to the centre of an elephant group/dung pile rather than to the closest or first individual/dung bolus seen; the latter option may seem more convenient, but will lead to positively biased density and abundance estimates. The procedure for correctly measuring the distance to the centre of a dung pile is detailed in Chapter 9. For direct surveys of elephants, if it is difficult to determine the location of the centre of the group at the time of the survey, then a potential solution is to take measurements to the left- and right-most individuals in the group relative to the transect line and to calculate the distance to the group's centre during analysis. The field protocol should clearly define what is meant by an elephant group in the context of distance sampling, rather than what might constitute a group when taking account the social

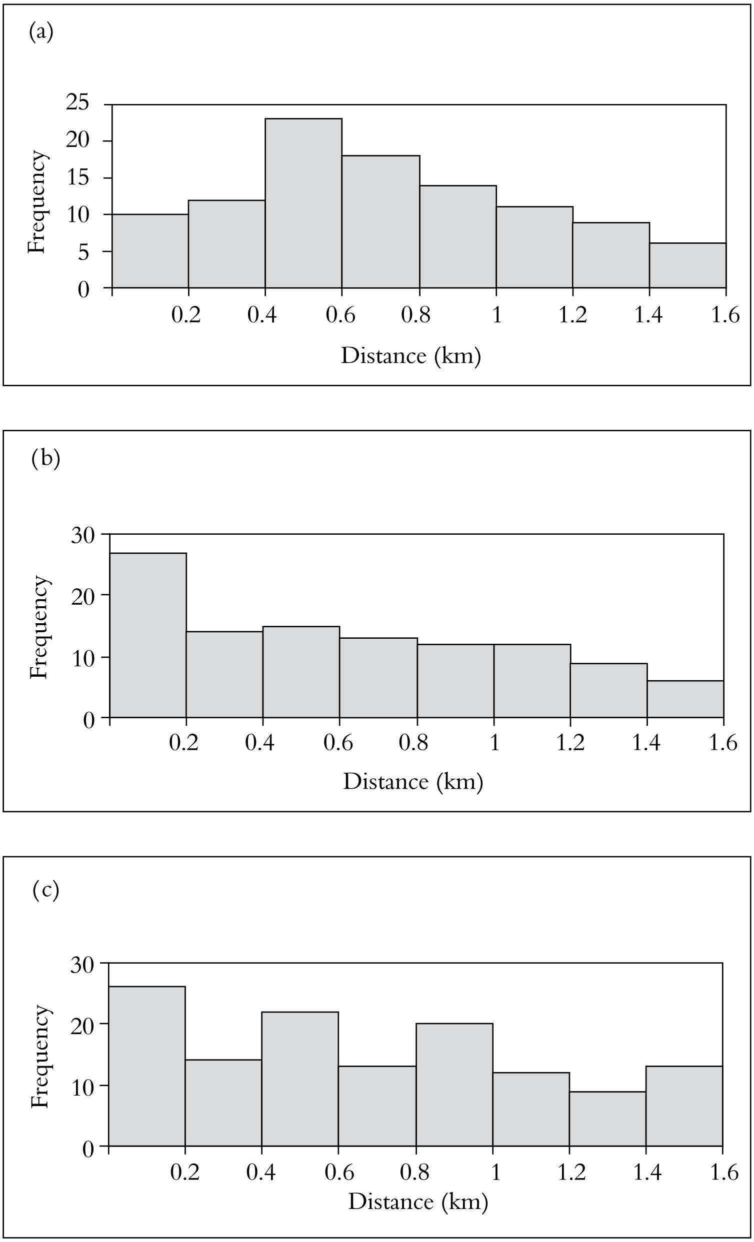

Figure 3.4 *Examples of problematic line transect distance sampling data: (a) evasive movement where animals have moved away from the observers prior to detection; (b) spiked data probably caused by rounding perpendicular distances or angles to zero, or by movement towards the observers prior to detection (less likely for elephant surveys); (c) heaped data where distances are rounded to convenient values.*

characteristics of the species. Thus, the group as a unit of observation for the purposes of distance sampling may be defined as a set of visible individuals where the nearest other member of the set is less than a predefined distance away (see Chapter 7).

For terrestrial sighting based elephant surveys, care must be taken to avoid a 'spike' at zero distance. This occurs if sighting angles to an animal or centre of a group a considerable distance ahead, is rounded to zero. For dung pile surveys, a spike at zero distance occurs if dung piles that straddle the transect line are recorded as having a perpendicular distance of zero rather than measuring and recording the perpendicular distance to the centre of the dung pile (as described in Chapter 9). See Figure 3.4b for an example of this kind of data problem. Technical aids (reticle binoculars, optical and laser range-finders, compasses, clinometers, etc.) should be used to improve the accuracy and precision of measurements. Taking measurements to a stationary feature in the environment at or close to the point of interest can greatly improve measurements (although sometimes several measurements must be made and aggregated if vegetation or other obstacles hinder a direct measurement or the distance is greater than the capability of the measuring instrument being used).

If the survey conditions make it especially difficult to record detection distances precisely, then an option is to record data by distance interval (this may be particularly relevant for aerial surveys). For line transects, intervals are usually narrower near the line and increase in width with increasing distance from the line, with 5–7 distance intervals being recommended and careful measurement near the interval cut points being required (again, for aerial surveys, these cut points will most likely be indicated by markings on the struts or window of the aircraft).

3.3.5 Animal group sizes are recorded accurately

Using the group as the unit of observation can lead to size bias if large groups are over-represented in the sample. With size-biased sampling, the detection probability of the group is a function of both distance from the observer and its size. There are several ways of dealing with this problem namely: (i) truncation—the data are right truncated to reduce the correlation between detection and group size; (ii) stratification by group size—the data are stratified by group size, where each stratum corresponds to a subset of the data with similar group sizes (with respect to detection) and the total abundance is obtained by summing by stratum estimates; (iii) weighted average of group sizes—this method assumes that the effective strip half-width is proportional to the logarithm of group size; (iv) regression estimators—these are used and the mean group size is then taken to be the size at the line (distance or detection at a given distance can be regressed against either group size or the logarithm of group size when group size is highly variable); (v) treating group size as a covariate—this is used when fitting the model for the detection function; and (vi) analysis by individuals —individual animals rather than groups (for more details, see Buckland et al. 2001: pp. 71–76, 122–130, 164–171).

Of the above methods, truncation is a simple and robust method and usually the data are less severely truncated to fit the detection function than when estimating mean group size. The recommendation is that the truncation distance be approximately equal to the width of the shoulder of the detection function (mathematically, the shoulder is the portion of the curve where the derivative $g'(0)$ is zero). However, truncation may not be an option if sample sizes are small. In this case, stratification by group size may be an option, but again only if there is sufficient data to estimate a separate detection function for each stratum. In the case of small sample sizes, expected group size can be estimated by taking a weighted average of group size, which is a better method for estimating group abundance, whereas stratification is a better way of estimating animal abundance. Regression methods can be effective in estimating group size and the slope of the regression of group size (or log group size) on distance tends to have a positive slope (as group size increases with distance), and on detection probability a negative slope. Sometimes the sign of the slope is reversed. This happens when observers underestimate the size of groups and the degree of underestimation increases with distance. Even in this case, however, using regression should give a valid estimate of mean group size. Analysis by individuals will give an unbiased estimate of animal density or abundance as long as all individuals on or near the line are detected (it is acceptable for observers to miss some animals in the group further from the line). This option reverts back to the individual as the unit of observation and requires the measurement of distances to individuals, which may also be a solution for surveys where it is impossible to obtain distances to the centre of groups. Distance sampling is particularly robust to the lack of independence between sightings of individuals that do in fact belong to the same group (see next section).

3.3.6 Other assumptions

Although they are not critical assumptions, there are other aspects of line transect distance sampling that are important to consider. One of these is whether or not detections are independent events. When detections are dependent (e.g., animals fleeing and disturbing others that are subsequently detected) this has little effect on the point estimate of density or abundance. However, analytical estimates of sampling variance will be negatively biased. This problem can be alleviated by using empirical estimators or re-sampling methods for variance estimation (e.g., using the bootstrap or jackknife that only assumes independence between transect lines).

An obvious case where this assumption is violated is when animals tend to aggregate and occur in groups. If animals aggregate in loose, poorly defined groups, then it may be necessary to treat each individual animal as an observation even if this violates the assumption (this should not be a major problem for elephant surveys). Otherwise, as we described previously, we treat the elephant group as the object of interest and measure the distance to the centre of the group, as well as the group size.

If animals move in response to the observers and are thus detected several times on the same or adjacent transect line, it can cause substantial positive bias (assuming repeat counting is common during the survey). If the same animal is detected more than once while sampling the same transect *at different times*, this is not a problem. Distance sampling theory also allows for an animal to be detected from different transects due to random movement of the animal.

It is worth noting that observations made behind the observer can be recorded, unless they occur prior to the start of the transect line (in which case they fall outside the sampled area).

Distance sampling theory performs well when detectability is certain near the line and remains certain or nearly certain for some distance from the line. Thus, the potential for detecting animal groups/dung piles should not drop off abruptly at a short distance from the line transect. Although this shape criterion is not an assumption, but a practical consideration, it is required to provide reliable estimates of density and abundance.

3.4 SURVEY DESIGN AND OTHER CONSIDERATIONS TO IMPROVE PRECISION

Variability in the density and abundance estimate is caused by (i) variance in observed sample size C or encounter rate C/L, (ii) variance in the estimated detection probability $\hat{p}$ or equivalently $\hat{f}(0)$, (iii) variance in the estimated expected group size $\hat{E}(s)$ and (iv) variance due to other multipliers. This emerges directly from the equations used to estimate density (Equations 3.4–3.6) that result in the estimate of variance given by equation 3.7. No multipliers are usually required for sighting-based elephant surveys but they are required for elephant dung pile surveys, where $E(s)$ is not required (see Chapters 4 and 9). Thus, we can reduce variability in the density and abundance estimate by improving the precision of these four components that contribute to the overall variability.

3.4.1 Reducing variance in observed sample size or encounter rate

Spatial variation in animal group/dung pile density between transect lines causes variance of observed sample size C or encounter rate C/L. This is often the largest contributor to the variance of the density estimate. Precision can potentially be improved by a number of different means, the first of these being stratification. If there exists heterogeneity in the population, then defining strata that are internally homogeneous reduces variance. By means of stratification, we attempt to make encounter rates along transects corresponding to a particular stratum as similar as possible, and encounter rates along transects corresponding to different strata as different as possible. To improve overall precision, different stratifications may be selected for different components—encounter rate, detection probability, mean group size, multiplier—of the density estimator. Stratification by habitat type (open grassland, swamp forest, etc.) is often sensible as one might expect both density and the probability of detection to change by habitat type. This type of stratification during the design phase of the survey is only possible if the habitat types are not too fragmented and intertwined. An option for regions with patchy (fine-grained) variation in habitat types is to keep a record of when the habitat type changes or to sub-divide the transect line and classify these line segments according to the predominant habitat type. One would then have a total for the amount of effort spent in each habitat type, which would allow for post-stratification by habitat type during analysis. Variables such as season or time of day might also affect encounter rate, or the other components of the density estimator, and stratification by these variables should be considered. If something is known about the relative number of animals within each stratum, then an approximate rule of thumb is to allocate effort proportional to abundance to achieve the best overall precision. See Buckland et al. (2001: pp. 246–248) and Cochran (1977: pp. 96–98) for a more detailed and exact description of optimal effort allocation between strata. Thus, precision can be improved by allocating more survey effort to those strata that have more elephants, but the distribution of effort by stratum is not as important as the total line length L, and clearly, the more overall effort, the greater the improvement in precision.

Stratification may also be used for logistical reasons or because density estimates are required for certain sub-regions of the survey area for management purposes (e.g., inside a national park *versus* outside). In the former case, for example, the study area may be stratified according to ease of access and less effort may be allocated to hard to access strata for reasons of cost-efficiency. This may lead to some loss in precision, but logistics or the need for estimates by predefined survey units may require such stratification nonetheless. If the stratification occurs for other than logistical reasons and if nothing is known about density in each of the strata then effort should be allocated in proportion to stratum size.

If strata are defined or if there is simply a single stratum corresponding to the entire survey region, then to further improve precision one should orientate transect lines parallel to any gradients of density within each stratum. In this way, variation in encounter rate is maximised within transects and minimised between them. So, for example, if one suspects that density decreases with increasing distance from the edge of a habitat, a topographic feature such as a river, or a human modification to the landscape such as a road, then transects would be placed approximately perpendicular to the edge of the habitat type, river, or road.

As mentioned previously, to ensure a representative sample and to get a reliable estimate of variance in observed sample size (or equivalently encounter rate) you need at least 15–20 replicate lines per stratum, preferably 25 or more. The larger the number of line transects, the more likely it is that you will obtain a representative sample and a reliable estimate of variance. This consideration usually limits the number of strata that can be defined, as each needs an adequate number of transects. In general, a design that has a larger number of shorter lines is preferable to one with a smaller number of longer lines when it comes to obtaining a representative sample and also a more precise estimate of the variance of the density and abundance estimate.

For each survey stratum, not only the orientation of the lines, their number and length, but also their location relative to one another has implications for the precision of the density estimate. Survey designs that locate line transects systematically with a random start give a more even spatial spread of survey effort over the survey region than their non-systematic counterparts, where each transect line is randomly located (see Figure 3.5). A systematic survey design whose line transects have a more even spatial distribution is more robust and likely to lead to less variation in the estimates of density or abundance, as it is less susceptible to variations

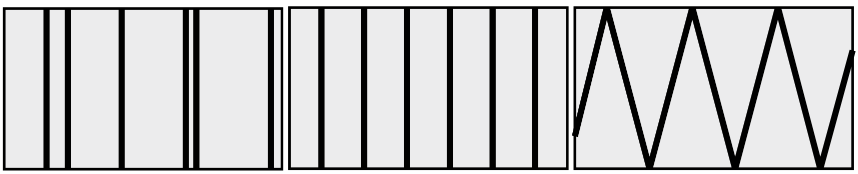

Figure 3.5 *A line transect survey comprises a series of lines. Illustrated here within a simple rectangular survey region are examples of three commonly used designs: randomly-spaced parallel lines (left); systematically-spaced parallel lines with a random start (middle); and a randomly placed continuous zigzag (right).*

in population behavior [Strindberg 2001; Strindberg and Buckland 2004]. In other words, transect lines that have a more even spatial distribution tend to improve precision, as they ensure that a more representative sample is selected from the population. There have been recent advances in estimating the variance of encounter rate for all types of designs; for systematic designs, in particular, it is now possible to exploit the greater precision these tend to afford [Fewster et al. 2009]. Aside from the issue of precision, systematic designs are also more efficient and consistent when it comes to the distances the observers need to cover during the survey, due to the even spatial distribution of the transect lines. These and other designs are described in more detail by Strindberg et al. (2004).

For line transect surveys, continuous systematic designs such as zigzags are sometimes the preferred option as they are more cost-effective (see Figure 3.5). Since observers do not have to move from one survey line to the next while not recording sighting data, continuous systematic designs provide an added efficiency advantage of no dead time between sampling periods. However, they do not provide as even a spatial distribution of their transect lines compared to a systematic design with discrete parallel transect lines and are more complicated to design in irregular survey regions. Aside from this, they have the potential to cause some bias in the estimation process (Strindberg and Buckland 2004).

Accuracy and precision of density and abundance estimates, as well as the efficiency achieved, are determined to a large extent by the survey design that dictates how the sample is collected. To obtain estimates of animal density or abundance based on valid statistical inference the observations should be obtained by means of a probability sample. This requires that the line transects be located randomly. All survey design options discussed above presuppose a randomised sampling design. This design process is facilitated by the development of automated design algorithms that randomly superimpose line transects on the survey region of interest [Strindberg 2001; Strindberg et al. 2004]. By automating the survey design process, it is also possible to contrast designs with regard to properties such as the spatial distribution of sampling locations within the survey region, the distances covered by observers to obtain the sample data and the probability of a particular location being included in the sample (coverage probability).

The DISTANCE software has an automated survey design component and Geographic Information System (GIS) functionality that can be used for the design of distance sampling surveys [Thomas et al. 2010]. In order to design a survey using this software, one has to define the survey area in a spatially explicit manner by means of an ESRI shapefile. This component permits the selection of a design from among a number of different possibilities and the exploration of the design properties given the logistical

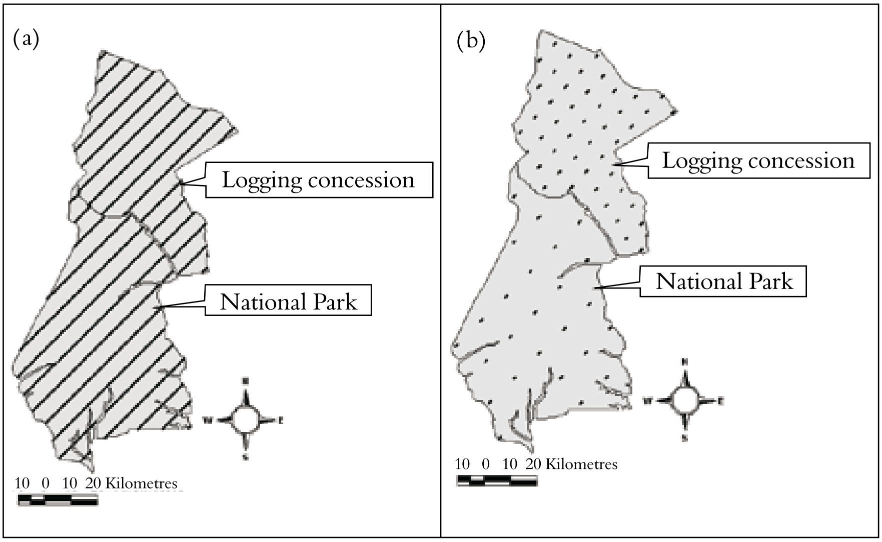

Figure 3.6 *Consider a survey region that comprises two strata, namely a national park and a logging concession, where the latter is more heavily impacted by human activity. Transect lines are oriented in a northeast-southwest direction as this is suspected to coincide with the gradient in elephant density in the area. (a) If the habitat type were open and suitable for direct surveys of elephants by means of either an aerial or a terrestrial survey, then a design with systematic parallel lines located with a random start in each stratum might be appropriate. Note that the spacing between sequential line transects is 6.5 km in the national park, but 6 km in the logging concession to ensure sufficient replication (15 lines in each stratum). (b) In the case of a closed forest habitat type, a line transect based survey of elephant dung (covered in Chapter 9) would be a likely option. In both cases, the amount of time it takes to cover a kilometre of transect line and to move between transects is an important design consideration, as total survey time is always limited by cost and other logistical constraints. In closed habitat types, it is often difficult to cut transects and to move along them in a straight line, whereas ground can be covered much more quickly if animal paths are used. Hence, a design comprising 1–2 km long line transects systematically spaced with a random start with a larger spacing between transects could be a good design option. In this example, the 25 and 47 transect lines are 1 km long and have a systematic spacing of 12 km and 7 km in the national park and logging concession, respectively. Given the roads in the logging concession, it is easier to access this stratum and thus it is more cost-efficient to allocate more sampling effort to this stratum (even though overall precision may not be improved because elephant numbers are reduced in this stratum due to human disturbance).*

constraints for the survey in question. A number of frequently used line transect designs, both systematic and non-systematic, with discrete or continuous transect lines, have been implemented within the automated survey design component of the DISTANCE software. The designs shown in Figure 3.6 are examples of survey plans that can be produced using this feature of DISTANCE.

3.4.2 Reducing variance in detection probability

As for encounter rate, stratification (or post-stratification) can be used to improve the precision of $f(0)$. If detection changes by habitat type, observer, environmental conditions, etc., then estimating the detection function separately by strata defined by these variables should decrease its variance. However, as adequate sample sizes are required for reliable estimation, the number of strata should be such that enough observations occur in each stratum. The data may be stratified differently when estimating encounter rate or mean group size versus $f(0)$ especially since encounter rate and mean group size can usually be estimated reliably from smaller sample sizes.

Another option for improving the precision of $f(0)$ is to include multiple covariates (e.g., habitat type, season, observer, group size or environmental conditions) into the distance analysis. These variables are incorporated as a covariate when fitting the model for the detection function [Marques and Buckland 2004a; Marques and Buckland 2004b]. This can be an efficient way to reduce variance when it is not possible to obtain sufficient sample sizes for stratified estimation of detection and also has the potential to help us understand how variables affect detectability. The methods assume that these types of covariates influence the scale of the detection function, but not its shape (simply plotting the observations separately by covariate should tell one whether this assumption is likely to hold). Thus animal groups/dung piles at the same distance from the transect line can have different probabilities of detection depending on their associated covariate values. Aside from the Conventional Distance Sampling (CDS) analysis option implemented within the DISTANCE software, which facilitates a standard analysis, the Multiple Covariate Distance Sampling (MCDS) analysis option is also available.

3.4.3 Reducing variance in expected group size

An option for reducing the variance in estimated expected group size $\hat{E}(s)$, if group sizes change seasonally, is to survey when group sizes are smaller. This facilitates group size estimation and also increases encounter rates (if there is interest in obtaining seasonal estimates of density, however, then one would have to survey during the various seasons regardless of expected group sizes). Additionally, if sample sizes are adequate, then it may be possible to post-stratify by group size during analysis to improve precision.

3.4.4 Reducing variance due to other multipliers

Sometimes, the estimation of animal density and abundance requires multipliers in the formulation of the estimator. Some multipliers are constant and do not have any associated variance. Examples include multipliers that reflect the sampling fraction—for example, if only one side of the lines is surveyed during a line transect survey, then the density estimate would be multiplied by 2, or the number of times each of the transect lines were surveyed—for example, if the lines were each covered 3 times, then the density estimate would be divided by 3. Other multipliers have an associated variance. For example, if detection on the line is not certain then it may be necessary to estimate the proportion of objects detected on the line and this multiplier will be incorporated into the density estimate (if only half of the animals on the line are seen during a transect survey, then the density estimate is multiplied by 2). With surveys of dung piles (as discussed in Chapters 4 and 9), two multipliers are needed to convert sign density to animal density, namely dung deposition and disappearance (decay) rates. To obtain the estimate of animal density the estimate of dung density is divided by both of these rates, which tend to have little impact on overall precision [Plumptre 2000; Marques et al. 2001—specifically for decay rates], although they are difficult to estimate.

3.4.5 Precision versus available resources

Generally, when designing a survey, a balance has to be reached between the precision of the density estimate and the resources available for the survey in terms of time and money. This trade-off between desired precision and the cost of implementing the survey usually dictates the survey effort and design used in sampling a particular study area. A pilot survey is the best way to estimate the amount of survey effort required to achieve a desired precision. The time and cost constraints associated with a particular type of survey in a given study area will usually dictate whether the desired precision is feasible and which survey design is most suitable for the given circumstance.

As described in the introduction, the coefficient of variation (CV) is a useful unit-less quantity that can be used to compare different studies. If groups are the objects of interest, as they would most likely be in a sighting-based line transect survey of elephants, detection on the line is certain and there are no multipliers, then by applying the formula

$$L = \left(\frac{b + [\hat{\text{std}}(s)/\bar{s}\,]^2}{[\text{CV}_t(\hat{D})]^2} \right) \left(\frac{L_0}{C_0} \right) \qquad (3.9)$$

one can estimate the total length of transect line, L, required for a given encounter rate $\dfrac{C_0}{L_0}$ and a target CV for the density estimate $\text{CV}_t(\hat{D})$. The standard deviation of group size is $\hat{\text{std}}(s) = \sqrt{\sum_{i=1}^{C'} \dfrac{(s_i - \bar{s})^2}{(n-1)}}$, where $\bar{s}$

is the mean group size and s_i the size of the i^{th} group, which assumes group size is independent of detection distance. The parameter b is known as the dispersion parameter or variance inflation factor and is approximately given by

$$\frac{\text{var}(C)}{C} + C\frac{\text{var}(\hat{f}(0))}{[\hat{f}(0)]^2}.$$

The dispersion parameter generally takes a value in the range 1.5–3. It would take on its smallest value if the spatial distribution of the animals were random, as then one would expect the count on each line to approximately follow a Poisson distribution (i.e., $\hat{\text{var}}(C) \cong C$). If the population is highly aggregated, then b takes on larger values. To avoid underestimating L for planning purposes, it is suggested that one uses a value of 3 for b (assuming it is not possible to estimate b from a pilot study or use a value calculated previously from a similar study). Ideally, a pilot study would be carried out to estimate the encounter rates expected during the actual survey, and the mean and standard deviation of the group size. These values can then be plugged into the above equations to estimate the amount of effort required to achieve the desired precision. A simple pilot study during which distances to the animals are not measured can be conducted to estimate these values. If the pilot study is more comprehensive, and also includes distances to detected animals, then the dispersion parameter b can be approximated by $C_0 \cdot \{\text{CV}(\hat{D}_0)\}^2$, where C_0 is the number of animal groups counted during the pilot survey and D_0 the corresponding density.

If the available resources determine the total effort in terms of line length, L, then it is possible to estimate $\text{CV}(\hat{D})$ using the formula

$$\text{CV}(\hat{D}) = \sqrt{\frac{(b + [\hat{\text{sd}}(s)/\bar{s}\,]^2)L_0}{LC_0}}. \quad (3.10)$$

If $\text{CV}(\hat{D})$ is too large, then it may not be worthwhile conducting the survey, if a certain precision is required. Similarly, we can calculate the amount of effort, L, required to achieve our desired $\text{CV}_t(\hat{D})$ and possibly conclude that we do not have the resources; then it is necessary to decide whether a reduction in precision is feasible given the goals of the survey. All of these equations assume that the lines are distributed randomly (or systematically with a random start) within the study area. Additionally, if detection on the line is not certain and $g(0)$ or other multipliers (such as decay and defecation rates for dung pile surveys) need to be estimated, then greater effort is required to achieve a target precision (equivalently the same amount of effort will give lower precision). For more detailed explanations and example calculations, see Buckland et al. (2001: pp. 241–244).

3.4.6 Options for reducing bias and improving precision

For animals that live at low density but are also aggregated into groups, as elephant populations frequently are, the line transect sample sizes (either of direct sightings or dung piles) may be small even if a great deal of sampling effort is invested in the survey (note that for a set total survey effort L and a given encounter rate from a pilot survey of C_0/L_0, the resulting sample size can be estimated as $C = L \times C_0/L_0$). This often results in imprecise estimates that are potentially biased given the unreliability associated with fitting the detection function. To reliably model detection one should aim for a sample size of 60–80 observations (elephant groups/dung piles) for line transect surveys. The exact sample size required depends on the nature of the data; usually fewer observations are needed if detectability is certain near the line and remains certain or nearly certain for some distance from the line, whereas a larger number is needed if detection drops off rapidly with distance from the line.

Adaptive sampling (see Glossary) is one way to increase the sample size, thus also increasing precision and reducing bias [Thompson and Seber 1996]. An inherent problem with adaptive sampling is that usually the total survey effort required to complete the survey is unknown in advance, which can create severe logistical problems. However, an adaptive line transect sampling method that allows the amount of effort (in terms of survey time) to be fixed in advance has been developed [Pollard et al. 2002; Pollard and Buckland 2004]. This adaptive sampling method works by increasing effort in high density areas, with the amount of adaptive effort varying depending on whether the survey is behind or ahead of schedule. For elephant surveys, it may be possible to apply an adaptive survey protocol to a terrestrial or perhaps even an aerial setting, although the latter might be more difficult given the speed of the observers. In both instances, an increase in complexity in the field protocol and analysis is to be expected.

The previous discussions about reducing variance by means of survey design or analysis techniques are for standard distance sampling which combines both model- and design-based inference[2] for estimating density or

[2] With design-based inference no assumptions are made about the sample population and the sampling elements are chosen randomly and independently of the population. The survey design determines the sampling process that introduces selection probabilities for each sampling element. These selection probabilities determine the properties of the estimator. The sampling process itself is the source of all the uncertainty. With model-based inference the

abundance. The former is used when fitting the detection function in order to estimate density within the sampling units and, as mentioned previously, relies on the assumption that all animal groups/dung piles in the surveyed strips are uniformly distributed in the interval $[0, w]$. The latter relies on the properties of the survey design to extrapolate from the sampling units to the larger survey region and makes no assumptions about the characteristics of the population in the study area during a standard distance sampling analysis. An alternative approach for estimating abundance and possibly also reducing variability is through entirely model-based inference [Thompson 1992]. If certain variables are thought to influence density and distribution, then a model that incorporates these variables can be fitted to the distance sampling data to potentially improve precision [Hedley and Buckland 2004a; Hedley et al. 2004b; Johnson et al. 2010]. The variables can either be collected during the survey as ancillary data or they can be obtained from other sources, e.g., from a GIS or from other spatially explicit data sources for the study area. These spatially explicit models allow one to investigate factors influencing abundance (habitat type, other environmental variables, distance to human settlements, etc.) and to calculate animal abundance for sub-regions in the study area (see Stokes et al. (2010) and Yackulic et al. (2011), for an application of these types of models to line transect based surveys of elephant dung). The Density Surface Modelling (DSM) feature available in the DISTANCE software takes specifically formatted distance sampling data and predicts the spatial distribution of animals in the survey region as described by Hedley et al. (2004b).

Finally, it is important to note that fitting a spatial model does not necessarily rely on data collected from randomly located transects. However, it is preferable to employ a randomised sampling scheme, as both the standard and entirely model-based analysis options are then available. This is a far less risky strategy, as it does not restrict one's options to finding an unbiased model that fits the data well.

3.5 Data Analysis and The Distance Software

We briefly consider here the steps one might follow during an analysis of distance sampling data, including data entry and validation, data exploration, model fitting and selection, final analysis and inference. Again, Buckland et al. (2001) should be consulted for more detailed information. We also briefly introduce the analysis of distance sampling data using the DISTANCE software [Thomas et al. 2010] that fortunately comes with a detailed user's guide.

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assumption is made that the distribution of all possible realisations of values of the variable of interest can be described by some stochastic model.

3.5.1 Data entry and validation

If data are recorded on paper forms, it should be done accurately and neatly. These data should be stored electronically as soon as possible – entered into a spreadsheet, such as MS Excel, and validated to ensure that transcription errors are corrected and feasible data values are entered. If data were entered electronically in the field, e.g., using Cybertracker or another type of handheld computer, then some reformatting may be required before the data are ready for analysis, but validation can occur as the data is collected. It is extremely important that data validation occurs as soon as possible, as it becomes harder or impossible to sort out data errors later during analysis.

3.5.2 Data analysis with DISTANCE

DISTANCE is a custom windows-based computer package for the design and analysis of distance sampling surveys of wildlife populations [Thomas et al. 2010]. DISTANCE evolved from program TRANSECT and there were earlier DOS versions of the software. DISTANCE 3.5 is a windows-based version of the DOS version of the software. New versions of the software are regularly released as additional features are added or existing features updated. The software has context sensitive online help and a comprehensive user's guide. The most current version of the software can be downloaded at no cost from the DISTANCE web-site (www.ruwpa.st-and.ac.uk/distance).

DISTANCE projects are made up of a project file with a 'dst' file extension and an associated data folder with a 'dat' suffix. Internal data are stored in DistData.mdb file within the data folder. The data folder is also used by default to store the GIS information (ESRI shape files) for the project, if it exists. The freely available R statistical software (www.r-project.org) needs to be installed to use the Mark-Recapture Distance Sampling (MRDS) analysis or Density Surface Modelling (DSM) features available in DISTANCE and implemented as R libraries. An R folder is created automatically within the DISTANCE data folder the first time the MRDS or DSM libraries are called. It contains the R object file (.RData) and image files generated by the R statistical software package.

Besides creating a new project (which can be set up using another project as a template), it is possible to import data or project files from earlier versions of the software. If the survey data are stored in a package such as MS Excel then it is relatively straightforward to import the data into DISTANCE from such software by saving the data in a predefined order (described in detail in the online help) as a tab delimited text file. This is much easier and less error prone than typing the data into DISTANCE. The data should be imported in its disaggregated form, i.e., if distance data were collected ungrouped, then they should not be grouped even if they are subsequently analysed that way. Data should be entered taking into account stratification and sampling units (usually lines). Data can also be linked from external databases in a variety of formats. It is possible to export DISTANCE projects to zip archive files, facilitating transfer to other computers or users, and to open projects directly from the archive file.

The software's graphical user interface comprises a *Project Browser* that has tabs for the *Data Explorer*, *Maps*, *Designs*, *Surveys* and *Analysis*. The first tab, as its name suggests, allows for data entry and data exploration or alterations. If the study area is spatially referenced (e.g. the definition of the study area is available as a shape file or the geographic coordinates are known), then the study area details can be entered and displayed using the *Maps* feature. Additionally, for such study areas, various characteristics for potential survey designs can be explored using the automated design generation that is part of the *Designs* feature. Once this phase is complete, user-specified survey plans can be generated using the *Surveys* feature and the transect coordinates exported for import into the Global Positioning System (GPS) unit that will be used in the field. The *Analysis* tab is the focus of any analysis in DISTANCE.

As for any other analysis package, it is essential to understand the underlying methods before launching into the analysis. The analysis options should be selected with care, rather than simply using the defaults. Each analysis is made up of two components, namely a *data filter* and a *model definition*. The former allows you to select a subset of the survey data, group data into intervals for analysis or to truncate the data. The latter allows you to define how the selected data should be analysed by choosing options such as the detection function model, how potential size bias should be dealt with, or how to calculate variances, for example. The *New Analysis Components* window makes it easier to keep track of *Data Filters* and *Model Definitions*. Besides the Conventional Distance Sampling (CDS) methods, DISTANCE also has a Multiple Covariate Distance Sampling (MCDS) analysis component that allows for multiple covariates when fitting the model for the detection function, a Mark-Recapture Distance Sampling (MRDS) analysis component for the analysis of double-observer distance sampling data, and a Density Surface Modelling (DSM) analysis component that predicts the spatial distribution of animals, as described in the previous sections (refer to Buckland et al. 2004 for the theoretical underpinnings of these more advanced distance sampling topics and the DISTANCE user's guide for details on their application within the software). The focus here is on the standard distance sampling analysis via CDS.

Although there is no single approach to completing a distance sampling analysis, it should start with a good deal of data exploration before model fitting and selection take place and the final estimates are obtained and inferences are drawn.

3.5.3 Exploratory phase

Histograms of the data should be plotted under several different groupings. This can be done either using the DISTANCE software or any other package that has graphing facilities. It can even be done by hand using pen and paper and this may be particularly beneficial for novices to distance analyses. They can then gauge whether there are any problems with the data and estimate what the probability detection function might look like (see Figure 3.3a). Results obtained by means of this simple analysis could be compared to the results produced by DISTANCE.

Within DISTANCE an initial analysis with many (10–20) cut points and a simple model (e.g., half-normal) should be carried out. During this phase, it's best to not try to estimate density, but simply look at the histograms to see whether assumptions have been violated and there are problems with the data, such as heaping, a spike at zero distance, evasive movement or outliers.

The problems caused by heaping can potentially be reduced with appropriate grouping and if one fits a model with a wide shoulder (note that goodness-of-fit tests are sensitive to heaping and data should be appropriately grouped when performing these tests). It will be difficult to obtain reliable density estimates if evasive movement has occurred. Outliers that are caused by incorrect data entry should be corrected if possible during data validation. Then truncation can be used to eliminate the remaining outliers. Right truncation is generally recommended for robust estimation of the detection function. For line transects a rule of thumb is to truncate when $g(x) \approx 0.15$ (truncating approximately the largest 5% of distances usually works too).

Data can be left truncated for those surveys where detection is believed to be certain at some distance from the line, e.g., aerial surveys. Sometimes left truncation can be used when there are too few or too many observations near the line (this is not generally a recommended approach, as it is difficult to gauge when and which degree of left truncation is appropriate). If the former is due to evasive movement and the latter due to heaping at zero or responsive movement towards the line, then the estimates will be positively (too many observations further away) and negatively (not enough observations further away) biased, respectively. However, left truncation may give estimates more representative of the study area as a whole for data collected along paths where animal density tends to be higher or lower than in the remainder of the study area—although we recommend strongly against using paths to collect elephant sighting or dung count data if a line transect based approach is being used (*cf.* the occupancy surveys described in Chapters 6 and 11).

Decide whether to analyse data as grouped (use the *Data Filter* in the DISTANCE software) or ungrouped (use the *Model Definition* in the DISTANCE software). If the exploratory phase of the analysis shows signs of rounding at convenient values, then the former option should be selected with cutpoints defined in such a manner that rounding distances lie approximately at the midpoints of intervals; this way observations will tend to fall within the correct distance interval. For aggregated populations, check for evidence of size-bias and apply one of the methods described previously to take care of this problem if necessary.

3.5.4 Modelling the detection function

The detection function $g(x)$ gives the probability that an animal at distance x is detected from the line. To estimate density, a distance sampling analysis relies on fitting a model of $g(x)$ to the observed distances (where x corresponds to the perpendicular distances $x_1, \ldots, x_C$ recorded during the line transect survey or calculated from the radial distances and angles or angles of declination and altitude), which allows one to estimate the proportion of animal groups/dung piles within surveyed strips that are detected and counted (p). A precise and unbiased estimate of animal density relies on the selection of an appropriate model for $g(x)$. Such models have certain desirable properties, namely (in order of importance) model robustness, a shape criterion and efficiency.

Model robustness A robust model needs to be general and flexible so that it can fit a variety of shapes for the detection function. The models used generally have 2, 3, or a variable number of parameters. Models should also be pooling robust, which means that the data can be combined (pooled) over different factors that affect detectability (habitat, observer, weather, etc.) and still provide a reliable density estimate. In other words, the density estimate produced by stratifying the data by habitat, observer, weather, etc., should be approximately the same as the estimate produced from the combined data.

Shape criterion As mentioned previously, the potential for detecting animals should not drop off abruptly at a short distance from the line transect. In other words, $g(x)$ should have a 'shoulder' near the line transect. Given this property, spiked functions near zero are excluded from consideration. It is worth noting that histograms of the detection distances often do not reveal the presence or absence of a shoulder, especially if histogram groupings are large.

Efficiency An efficient model is one that has a small variance. Maximum likelihood methods are used as they ensure a minimum variance asymptotically, i.e., as the sample size increases. This characteristic is only useful if the model is robust and when the shape criterion can be met. Otherwise, you may get a very precise estimate that is wrong!

Models of the form $g(x) = \text{key}(x) \times [1 + \text{series}(x)]$ have these characteristics, where key(x) is a key function and series(x) is a series expansion. The modelling process involves two steps:

- Selection of a key function based on the histogram data (after truncation);
- Adjustment of the key function by means of the series expansion.

Key functions include the uniform, half-normal and hazard rate functions (the exponential with a simple polynomial adjustment should only be used in extreme circumstances to salvage truly spiked data—if possible the hazard rate model should be used if the spike is real, and even then it may lead to very imprecise estimates depending on the data). Series expansions include the cosine series, simple polynomials and Hermite polynomials. Sometimes, a key function without a series adjustment is sufficient.

Model fit The χ^2 goodness-of-fit (gof) test is used to test the fit of the $g(x)$ model to the distance data. The test is based on the grouping of the distance data and compares the observed frequencies C_i (dependent on the groupings selected) to the expected frequencies $\hat{E}(C_i)$ under the model in the usual way $\chi^2 = \sum_{i=1}^{u} \frac{[C_i - \hat{E}(C_i)]}{\hat{E}(C_i)}$, which is approximately χ^2 distributed with $u - q - 1$ degrees of freedom, if the fitted model is the true model (where u is the number of groups and q is the number of parameters estimated). A defect of the χ^2 gof test is that it has difficulty discriminating between different models at the most critical region near $x = 0$, unless given enough data, and the results are very dependent on the groupings selected. The power of the gof test is low, too, and should not be relied on when selecting a model for $g(x)$, but the test is useful for highlighting problems with the data. Alternatives available for analyses on exact distances (i.e., distances that are not grouped into intervals) are the Cramér-von-Mises and Kolmogorov-Smirnoff gof tests. These avoid arbitrarily grouping exact data and are also available in DISTANCE. In addition, for analyses on exact distances, quantile–quantile (qq) plots provided by the DISTANCE software are a graphical means for identifying problems with the data, e.g., rounding to preferred values or systematic departures from the fitted model (see the 'Qq-Plots' and 'CDS goodness-of-fit tests' entries in the DISTANCE help file for more detail).

Model Selection In general, as the number of parameters in the model increases the bias decreases, but the sampling variance increases. Hence, the number of parameters selected needs to be a compromise between bias and variance. Model selection should only take place once the data have been adequately truncated and various data groupings considered. The fit of the model to the distance data near the line is extremely important (except in the case of heaping at zero). Akaike's Information Criterion (AIC) and likelihood ratio tests are used in model selection.

The AIC attempts to find a balance between the number of model parameters q and the model fit, and in this way provides a trade-off between variance and bias (more parameters improve model fit and reduce bias, but the cost is an increase in model complexity and variance). For a given data set, the model with the smallest AIC is selected. The AIC is given by $AIC = -2 \log_e(\mathbf{L}) + 2q$, where $\log_e(\mathbf{L})$ is the log-likelihood function evaluated at the maximum likelihood estimates of the model parameters. There also exists a form of AIC, whose derivation is based on assuming normality, that takes sample size n into account. If denoted by AICc then

$$AICc = -2 \log_e(\mathbf{L}) + 2q + \frac{2q(q+1)}{n-q-1}.$$

Both the AIC and the AICc give similar values for larger sample sizes, but for smaller sample sizes or a large number of parameters relative to the sample size, AICc is always better than AIC and should be used.

Likelihood ratio tests can only be used to compare nested models, so they can be applied when selecting the number of adjustment terms. Usually AIC (or AICc) is used as it can be applied to compare non-nested models as well.

Note that for analyses of data from surveys of direct observations of elephants that replace groups by individual animals, data will be overdispersed, so one should not rely on goodness-of-fit tests, likelihood ratio tests or AIC for model selection as they will tend to favor models that overfit the data. One solution is to fit a model to the group data and then reapply a model with the same number of adjustment terms to the individual animal data.

To select a model, fit a small number of key/series adjustment combinations (e.g., uniform + cosine, half-normal + Hermite polynomial, and hazard-rate + cosine). Look at the histograms, goodness-of-fit, AIC and summary tables to choose a model. The most important thing to consider is goodness of fit test results close to $x = 0$. To improve fit, it may be necessary to revert to the exploratory phase. Occasionally, it will be difficult to select between models that fit the data well and have similar AIC values (difference between them less than 1). In these cases, a solution is to resort to multi-model inference. For a detailed treatment of model selection and multi-model inference, see Buckland et al. (2001), Burnham and Anderson (2002) or Buckland et al. (1997).

3.5.5 Final analysis and inference

Once a model has been selected for the distance data, then consider (i) options for variance estimation (e.g., bootstrapping to estimate the variance of the estimate – bootstrap with more than one model selected, if model choice is uncertain and influential), (ii) stratifying some or all of the components of estimation and (iii) inclusion of covariates in the analysis. Fit the data using the favoured model or models and selected options to obtain the estimate of density or abundance. Finally, extract summaries from the analysis and histograms for reporting.

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CHAPTER 4

Estimating Absolute Density from Dung Pile Density

Simon Hedges

4.1 An Introduction to Dung Counts

4.1.1 Theoretical basis

Indirect survey methods, such as dung count based methods, allow us to estimate animal abundance and density when the sign produced by the animals (dung piles in this case) are more easily detected than the animals themselves (e.g., because they live in concealing habitat types such as forests at low density or move away before they can be seen). The use of dung count based survey methods to assess animal population size and trend is well established [for reviews see Neff (1968); Putman (1984); Barnes (2001); see also Buckland et al. (2001: 182–189)].

There are two main types of dung count: fecal standing crop (FSC) methods and fecal accumulation rate (FAR) methods. FAR methods measure the rate of dung pile accumulation between two points in time. This is achieved by making two visits to the same plots or transects and counting the number of dung piles deposited since the first visit. Provided the interval between visits is shorter than the most rapidly decaying dung pile's lifetime, animal abundance can be calculated from fecal accumulation rates and the mean defecation rate over the period of accumulation. Fecal standing crop (FSC) methods, by contrast, determine dung pile density (without revisiting areas) and relate this to dung pile decay rate and mean defecation rate [Neff 1968; Putman 1984; McClanahan 1986; Buckland et al. 2001: 182–189; Laing et al. 2003; Campbell et al. 2004; Smart et al. 2004; Walsh and White 2005; Jenkins and Manly 2008].

In general, we do not recommend FAR methods for estimating elephant abundance for a number of reasons. Most importantly, typical FAR methods are more time-consuming, and thus more expensive and labour-intensive than FSC methods that do not require repeated visits. Use of FAR methods also reduces the proportion of the survey area which can be covered per unit effort, since time which could be spent surveying new transects (or plots) is spent revisiting old ones. The use of permanent transects is also problematic in areas of dense vegetation, since elephants are likely to preferentially use the cut transects and so the estimates of dung pile density produced from counts along such transects are likely to be biased [Barnes 1996; Buckland et al. 2001; Nchanji and Plumptre 2001; Hedges and Tyson 2002]. We note, too, that the CITES MIKE* programme's *Dung Survey Task Force* also recommend against the use of FAR methods; however, see Section 4.6.2.5.

For the FSC methods recommended here (and by the CITES MIKE programme), one must estimate dung pile density and then use knowledge of the expected number of dung piles per elephant to convert dung density into an estimate of elephant density. Following Buckland et al. (2001: 182–189), if R is the estimated dung pile density (number of dung piles per unit area) and c is the estimate of the mean number of dung piles per elephant available to the surveyors during the time of survey, elephant density is estimated by R/c. To obtain an estimate of the mean number of dung piles per elephant available to the observer during the survey period, one must estimate mean dung pile production rate (defecation rate) per elephant per time period. One must also estimate mean dung pile lifetime or decay rate. The mathematics of the relationships among animal abundance and animal sign (e.g., dung pile) abundance via sign creation and survival probabilities are discussed in detail by Buckland et al. (2001: 182–185) and Buckland et al. (2004: 377–385). The theory and practice of using distance based sampling methods (including the line transect methods recommended in this manual) to estimate dung pile densities are explained in Chapters 3 and 9 and in a book-length treatment by Buckland et al. (2001).

The use of dung count based surveys for elephants, primarily those living in forest environments, is now well established and useful discussions can be found in Barnes and Jensen (1987), Hiby and Lovell (1991), Dawson

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and Dekker (1992), Barnes (1993, 1996), Varman et al. (1995), Barnes et al. (1997a; 1997b), Walsh and White (1999), Plumptre (2000), Barnes (2001), Marques et al. (2001), Nchanji and Plumptre (2001), Walsh et al. (2001), Barnes (2002), Barnes and Dunn (2002), Hedges and Tyson (2002), Laing et al. (2003), Campbell et al. (2004), Walsh and White (2005), Hedges and Lawson (2006), Kuehl et al. (2007), Jenkins and Manley (2008), Hedges et al. (in review) and Tyson et al. (in review).

4.1.2 Application

Can dung counts really provide accurate estimates of animal abundance? A number of studies have attempted to test the accuracy and precision of dung counts for elephants and their results were encouraging [Jachmann and Bell (1984); Dawson (1990); Jachmann (1991); Plumptre and Harris (1995); Varman et al. (1995); also see Barnes (2001), Eggert et al. (2003) and Hedges et al. (2007a; in review)]. More generally, dung counts have been shown to give estimates that are as accurate as other methods for a wide range of mammals and even for lizards [Barnes 2001].

Furthermore, dung counts have the advantage, at least in theory, that they can give estimates that are more precise than aerial sample surveys of elephants and other large mammals [Jachmann 1991; Barnes 2002]. This is because dung counts record the accumulated presence of the animals over the preceding weeks and months, so the variation between transects is low. Even when the variance in defecation and decay rates is combined with the variance in dung density, the variance of the final estimate of elephant (or other species) abundance may still be modest – although in practice this is often not the case usually because of highly variable decay rates and/or defecation rates [e.g., Walsh and White 1999; Plumptre 2000; Campbell et al. 2004; Hedges et al. in review]. In contrast, aerial surveys and terrestrial sighting-based surveys record the instantaneous distribution of animals, and the variation between transects is usually high, often giving estimates with wide confidence limits, particularly for aerial surveys [Jachmann 1991; Barnes 2002; also see Ellis and Bernard 2005; Msoffe et al. 2010].

Like other sample survey methods, dung counts will only provide good estimates if close attention is paid to detail and key assumptions are not violated. There are a number of aspects which need particular attention when planning dung counts; of particular concern are: (1) the methods used for estimating dung pile density, (2) consistent classification of dung piles into stages to facilitate decay rate monitoring, (3) the appropriate methods for estimating dung pile decay rates, (4) estimation of defecation rates, (5) the approach adopted to analyse the resulting data and, if one wishes to assess population age-structure as well as abundance, (6) the determination of elephant age from dung dimensions.

4.2 Estimating the Density of Dung Piles per unit area

4.2.1 Line transects

Line transect methods—including their use to establish the density of dung piles—are very well established and require no further introduction here (see, for example, the book-length treatment by Buckland et al. 2001); detailed guidance for the use of line transects to estimate elephant dung pile density are provided in Chapter 3 and Chapter 9.

4.2.2 'Recce' transects

From the late 1980s through the early 2000s, a number of so-called 'recce' (reconnaissance) survey methods were developed to be used in conjunction with line transects in an attempt to improve the precision of, primarily, dung count based surveys. These recce methods were developed by Richard Barnes, Jefferson Hall, Peter Walsh, Lee White, Steve Blake, Rene Beyers, and others in Africa [Barnes 1989; Hall et al. 1998; McNeilage et al. 1998; Walsh and White 1999; Beyers et al. 2001; Blake 2002] and further developed and evaluated in Asia by Hedges et al. (2000, 2002).

The 'recce' transect method, as described by Walsh and White (1999), involves walking along a 'path of least resistance' through the forest and counting all dung piles found, but not measuring perpendicular distances to these dung piles. Walsh and White found that dung pile encounter rates on recce transects were strongly correlated with encounter rates on nearby line transects, which, they argued, allowed recces to be used to estimate dung pile density providing the functional relationship between encounter rates on recces and line transects was derived from a subset of recces matched with line transects. Once this relationship has been established, a combination of recces and line transects should provide a more precise estimate of dung pile density than line transects alone, because recces require roughly three times less effort than line transects and thus more ground can be covered by a given number of surveyors [Walsh and White 1999; Walsh et al. 2001]. Work in Asia by Hedges et al. (2000, 2002), in which recces and line transects were used within a non-purposive[1] stratified random sampling

[1] The basic recce method—walking more or less on a bearing while counting dung piles—can be used within a non-purposive survey design [e.g., using a stratified random survey design as per Hedges et al. (2000, 2002) in Indonesia] or it can be used in a purposive manner (e.g., following game or poacher trails while on patrol) as was originally proposed for Central Africa. Thus, the recce method can be used as part of a design-unbiased or a model-unbiased survey strategy (Hedges et al. 2000; Hedges and Tyson 2002). It is important to note here that, since 2003, recces to estimate animal abundance in Central Africa have also adopted straight lines

strategy in order to produce design-unbiased estimates of elephant density, also found recces to be quicker than line transects by a factor between 1.5 and 3, depending on terrain and dung pile density. In addition, the recces only required three people per team, while line transects required five or six. Thus, at first glance at least, the use of recces and line transects appeared to be substantially more efficient (in terms of effort and cost, if not precision per unit effort) than line transects alone, even when a non-purposive stratified random sampling design was used. The actual increase in efficiency depended on terrain, accessibility and dung pile density; for example, in very remote areas, a non-purposive stratified random sampling design results in significant increases in travel time compared to a purposive design of the type often used in Central Africa [Hedges et al. 2000, 2002].

The combined line/recce transect approach of Walsh and White was widely adopted for elephant surveys in the forests of Africa, including those conducted as part of the CITES MIKE programme. However, there was always a debate about the most appropriate survey design to use when employing line and recce transect methods and, in particular, whether a purposive design was appropriate [Walsh 1999; Buckland 2000; Burn and Underwood 2000; Hedges et al. 2000; Thomas et al. 2001]. Eventually, a consensus began to emerge, with the general agreement that a line and recce transect combination should be used in a non-purposive design, at least for the MIKE Program [Buckland 2000; Thomas et al. 2001]. Nevertheless, despite the early agreement about the utility of recce transects, a number of problems were identified:

- The methods used to calibrate recce transects against line transects differed between studies. Walsh and White (1999) suggested calibrating a subset of the recce transects against line transects running parallel to the recces and 50 m distant from them, which was also the method used by Hedges et al. (2000, 2002) in Indonesia. However, evaluations of recce methods conducted during the MIKE Program's Central African Pilot Project used a number of different methods to derive recce-transect ratio estimators, e.g., recces conducted perpendicular to the line transects and recces continuing from the end points of line transects [Beyers et al. 2001; Thomas et al. 2001].
- There was unresolved uncertainty about the amount of effort that should be allocated to establishing the functional relationship between dung pile encounter rates on line and recce transects appears [Burn and

arranged in a design calculated using the DISTANCE software program (these are known as Guided Recces and never use trails or roads). Recces using trails or roads are now known as Travel Recces and, at least in Central Africa, are not now used for anything other than as a guide to where poachers are active.

Underwood 2000; Hedges and Tyson 2002]. In other words, how often should recces be calibrated and in how many areas per survey site?

- Doubts about the power of combined line/recce transect surveys to detect changes in population size, particularly when elephant populations are small. As discussed elsewhere in this manual, precision can be improved by re-surveying transects in subsequent years [see also Plumptre 2000; Beyers et al. 2001; Buckland et al. 2001] and this was shown to be possible using GPS technology during the MIKE Central African Pilot Project, provided transect start points were well marked and the transects were conducted along closely controlled compass bearings [Beyers et al. 2001]. It is not possible to re-survey recce transects with anything like the same precision [Hedges and Tyson 2002].
- Finally, the work of Hedges et al. in Indonesia showed that survey teams tend to increase their recce-walking speed over time and the number of dung piles found on recces relative to the number found on paired line transects also declines over time. Common sense suggested that this was a causal relationship, and re-training resulted in slower recce-walking speeds and greater parity between dung pile encounter rates on recces and paired transects. The quality of line transects did not decline so rapidly, presumably because line transects require greater rigour, involve more people per team (so there are more people to spot dung), and are inherently slower than recces [Hedges et al. 2000, 2002]. Moreover, many problems with line transect data collection (e.g., rounding of perpendicular distances to 10s of centimetres and over representation of 0 m distances) can be identified easily from a simple examination of the data while recces only produce encounter rates that do not so obviously reveal weaknesses in the way the data were collected.

As a result of these concerns, the CITES MIKE Technical Advisory Group (TAG) did not endorse the use of recce transects for the MIKE program. Likewise, and for the same reasons, we do not recommend using recces here. We note, too, that subsequent to the development of the recce method a number of other methods have evolved to the point where they are now useful constituents of the elephant surveyor's toolkit: occupancy based methods [Chapters 6 and 11] and fecal DNA based capture–recapture methods [Chapters 5 and 10].

4.2.3 Strip transects

Generally, line transects are preferable to strip transects for dung surveys because dung pile visibility declines rapidly with distance from the observer, thus many dung piles are likely to be missed unless the strip transects are rather narrow in which case the density estimates will have wide confidence intervals [Burnham et al. 1985; Barnes 1993, 1996; Buckland et al. 2001].

However, in areas of tall grass and other concealing vegetation where dung pile visibility is very low, actively searching for dung piles in narrow strip transects might be preferable if large numbers of dung piles are likely to be missed along line transects. If strip transects are used either as a stand-alone survey method or in combination with recce transects, it will be necessary to demonstrate that all dung piles within the strip transects are found by the surveyors. This can be done by recording perpendicular distances from the centre of the strip transect to the geometric centre of the dung piles and analysing the resulting detection functions [Burnham and Anderson 1984]. It will also be important to emphasise to surveyors that dung piles should only be included in the strip transects if their geometric centres fall within the strip width [see Chapter 9 for further detail].

4.2.4 Adaptive sampling

Adaptive cluster plot based sampling has been proposed as an alternative to line transects, recce transects or conventional plot / strip transect based methods in the hope that its use will provide improved estimates of elephant population density [Burn and Underwood 2000]. See Section 4.4.6 for a brief discussion of adaptive line transect sampling methods. We evaluated the use of adaptive cluster plot methods for dung count based elephant surveys in southern Sumatra. We adopted a survey protocol similar to that suggested by Burn and Underwood: this entailed searching strip transects consisting of a sequence of 2×2 m plots (cells), moving onto the next plot in the strip transect if no dung was found but searching all adjacent plots if it was. If dung was found in any of these adjacent plots searching continued in the plots around these until a network bounded by empty plots was obtained. Searching then resumed along the strip transect as before [the basic methods are described in Thompson and Seber (1996) and Burn and Underwood (2000)]. We also experimented with 4×4 m and 6×6 m adaptive plots [Hedges et al. 2000, 2002].

We concluded that adaptive cluster plots presented too many logistical problems including, most seriously, the difficulty of laying out grids of adjacent plots in hilly and/or densely vegetated areas, and so we decided not to adopt this method for subsequent survey work in Sumatra. Additional problems with the technique, which, we suggest, argue against the use of adaptive plot-based techniques even in areas of relatively easy terrain include the fact that sample effort per area is unknown at the beginning of the survey, and so the survey teams cannot easily predict the time they will need to spend in the area (although there have been advances in the use of adaptive methods that help overcome this constraint—see Chapter 3 for an example related to line transect survey methods). A further issue is the need to allow sufficient 'extra' time per survey trip to allow the survey team to record any large clusters of dung piles found in plots [Hedges et al. 2000, 2002]. This last problem is likely to offset any gain in efficiency relative to conventional survey methods because the number of survey routes will be limited, or the routes would have to be rather short, so that sufficient time is available if large clusters of dung piles are found [Buckland 2000]. Buckland also argues that the efficiency of adaptive sampling is only appreciably higher than for conventional random or stratified random sampling if the distribution of objects is very clustered, but this is when the logistical difficulties of adaptive sampling are greatest. In light of these problems, we do not recommend use of adaptive cluster plot methods for dung count based surveys of elephants.

4.2.5 Other issues

Permanent swamps or seasonally inundated areas present a problem for dung count based survey methods. Leaving aside the logistic problems of actually conducting survey in such areas, there is the obvious problem that although elephants are likely to use such areas it will be impossible to count their dung piles accurately. For seasonally inundated areas this problem can be overcome by conducting surveys at appropriate times of year; for large areas of permanent swamp sighting-based methods will be needed [Chapters 2, 7, 10 and 11].

4.3 CLASSIFICATION OF DUNG PILES INTO RECOGNISABLE STAGES

Dung count based surveys rely on the field workers responsible for monitoring decay rates and those counting dung piles along transects being able to consistently classify the dung piles into the appropriate classes based on their state of decay. It is very important therefore that the system adopted is simple to use and robust.

Until recently, the most commonly used system of classifying dung piles, used in almost all dung based elephant surveys to date, was the A–E system of Barnes and Jensen (1987). A system for estimating the age of dung piles (relying partly on changes in dung odour) was used in conjunction with the Barnes and Jensen classification system during the MIKE pilot project in Central Africa [White and Edwards 2000; Beyers et al. 2001]. At least one other system has been employed: that of Wing and Buss (1970), which was subsequently modified by Wiles (1980). We briefly review these classification systems below, explaining their limitations, then describe two alternatives including the standard MIKE dung pile classification system [the S system; Hedges and Lawson 2006], which was developed by the WCS elephant projects in Indonesia and the Lao PDR [Hedges et al. 2000; Hedges and Tyson 2002].

4.3.1 The Wing and Buss / Wiles system

The Wing and Buss (1970) / Wiles (1980) system is presented in Table 4.1 (below). As will be seen, the main problems are the lack of definitions for key terms, and, more seriously, the ambiguous cut-off points between the classes, which rely on expert opinion and are therefore problematic for surveys involving large numbers of people [Hedges and Tyson 2002].

TABLE 4.1 *The Wing and Buss / Wiles system for classifying dung piles*

Easily recognisable	Little noticeable deterioration. Boli remaining essentially intact and identification of dropping easy.
Recognisable	Extensive decomposition, erosion, settling and rearrangement of fecal materials may have occurred, but sufficient concentration of materials remain to allow definite recognition by an experienced worker during a field count.
Barely recognisable	Decomposition and removal of fecal material so extensive that only with care and examination of indirect evidence can the remaining materials be identified as components of an elephant dropping. May fail to be recognised by an experienced worker during a field count.
Not recognisable (Gone)	The removal or decomposition of fecal material so complete that identification as an elephant dropping no longer possible.

4.3.2 The MIKE Central Africa Pilot Project / White and Edwards system

A system for estimating the age of dung piles was used in the MIKE Central Africa Pilot Project [White and Edwards 2000; Beyers et al. 2001; Table 4.2] in addition to the A–E system partly to gather data on the seasonal use of areas. However, this ageing system is also problematic—mainly because of the difficulty of deciding what constitutes 'still smells of dung' and applying it consistently, especially when many people are going to be using the system [Hedges and Tyson 2002]. A second, related problem with this method stems from the stated need to break open dung piles to determine whether they still smell of dung, a process that facilitates the decay process. Thus, in addition to being difficult to apply consistently, the system can also introduce a systematic bias: i.e., decay rates of monitored dung piles will be quicker than unmonitored ones [Hedges and Tyson 2002]. Note, however, that this second problem would be solved by use of the so-called retrospective method of monitoring decay rate [Laing et al. 2003], which requires field workers to locate cohorts of fresh dung piles by making a number of visits (≥ 6 visits) to the area prior to the survey but only revisiting them once—at the time of the survey—to assess whether they are still present or have decayed (see Sections 4.4.1, 4.6.2.4 and Chapter 9). Finally, the criteria for each class do not necessarily co-vary in the manner implied by the definitions of each class; for example, it is possible for a dung pile's 'overall form [to] still [be] present although boli may be partly or completely broken down into an amorphous mass' but for the dung pile to still have 'odour' (*cf* the definition for 'recent' dung piles).

TABLE 4.2 *The MIKE Central Africa Pilot Project / White and Edwards system for classifying the age of dung piles*

Fresh	Sometimes still warm(!), with fatty acid sheen glistening on exterior and strong smell.
Recent	Odour present (break the boli), there may be flies, but the fatty acid sheen has disappeared.
Old	Overall form still present although boli may be partly or completely broken down into an amorphous mass, no odour.
Very old	Flattened, dispersed, tending to disappear.
Fossilised	The dung matrix has become soil, but the presence of a few resistant fibers or seeds still marks the location of an extremely old dung pile.
Mummified	Baked by exposure to sun, visibly once an elephant dung pile, but with no substance remaining.

4.3.3 The A–E system of Barnes and Jensen

The A–E system requires the surveyor to classify dung piles into six classes (A, B, C1, C2, D, E; Table 4.3) based on their state of decomposition [see Barnes and Jensen (1987: 3–4) or Dawson and Dekker (1992: 25–28)]; in some areas classes A and B have been amalgamated. Unfortunately, several problems exist with the system. Firstly, assigning dung piles to classes can be fairly subjective, especially as definitions of key term such as 'intact' are not given. This can lead to both intra- and inter-study differences between surveyors' classification of dung piles [Barnes 1996; Nchanji and Plumptre 2001; Hedges and Tyson 2002]; for example, in some surveys in Central Africa classification of dung piles as class E varied greatly between observers, even after weeks of training. Secondly, the classification of many dung piles relies on assessing whether 'more than half the boli are still distinguishable as boli'. However, for this system to be applied unambiguously in the field one has to know how many boli there were in the first place—which is clearly impossible. And equally clearly, if one takes the definitions to refer to the actual number of boli still remaining in the field (rather than the number originally deposited) a dung pile can 'undecay', e.g., from stage C2 to stage C1 [Hedges 1993; Hedges and Tyson 2002].

TABLE 4.3 *The A–E system of Barnes and Jensen*

A	Boli intact, very fresh, moist, with odour
B	Boli intact, fresh but dry, no odour
C1	Some of the boli have disintegrated, but more than half are still distinguishable as boli.
C2	< 50% of the boli are distinguishable; the rest have disintegrated.
D	All boli completely disintegrated; dung pile now forms an amorphous flat mass.
E	Decayed to the stage where it would be impossible to detect at 2 metres in the undergrowth; it would not be seen on a transect unless directly underfoot.

4.3.4 The MIKE S system

The Barnes and Jensen A–E system described above can be modified to make it easier to apply consistently and thus make it more suitable for use in multi-team surveys [Hedges 1993; Hedges and Tyson 2002; Hedges and Lawson 2006]. The essential modifications being: (1) providing a definition for what constitutes an 'intact' bolus and (2) classifying dung piles according to the proportion of the remaining boli that are still intact (Box 9.1). As mentioned above, this second modification does mean that dung piles can 'undecay', but in practice, this problem has an insignificant effect on decay rates [Hedges, unpub. data] provided they are estimated using so-called retrospective methods [Laing et al. 2003; Sections 4.4.1 and 4.6.2.4] because these methods only require the field teams to revisit the monitored dung piles once and decide whether they are still present or not. What is more, because retrospective methods assess decay rates in the period leading up to and including the survey to estimate dung pile density, any 'undecaying' of dung piles in the monitored plots should mirror what is happening to the un-monitored dung piles available to the surveyors conducting the dung pile density estimation transects.

4.3.5 The need for a multi-stage classification system and an experimental alternative to the S system

It is clear that one of the main problems with classifying dung piles is that the process is rather subjective and so it can be difficult to apply any system consistently [Hedges 1993; Hedges and Tyson 2002; Walsh and White 2005; Hedges and Lawson 2006; Kuehl et al. 2007]. Having multi-stage classification systems arguably exacerbates this problem. In principle, all that is needed is a simple two-stage system ('dung pile is still present' / 'dung pile is deemed to have disappeared'). Clearly, a consistent system for deciding whether a dung pile is deemed to have disappeared would still be needed, but such a binary system may facilitate greater consistency.

However, having several classes allows for greater flexibility. For example, consider a survey team that finds that a large proportion of the dung piles they had monitored over the months before their survey are still in classes S1–S3 at the time of their survey. If the team were to calculate elephant density using an estimate of dung pile density derived from dung piles in classes S1–S3 then the estimate of elephant density would be biased. However, providing that the majority of monitored dung piles had made the transition to stage S3 prior to the survey, an unbiased estimate of elephant density could be calculated from an estimate of dung pile density derived from dung piles in classes S1–S2. A simple 'present' / 'disappeared' system does not allow such flexibility [Hedges and Tyson 2002; Hedges and Lawson 2006]. In addition, when dung pile decay times are very long, as they often are for elephant dung, a long period of monitoring decay rates will be required and this will require either a permanent presence in, or several separate missions into, remote areas (which can be extremely expensive and/or difficult). Thus there is often a need to manipulate decay rates to reduce the decay rate monitoring period [Sections 4.6.2.4 and 4.6.2.5].

Based on our extensive field experience in Asia, the S system described above [also see Section 4.3.4] seems to be an improvement on earlier systems; however, getting field teams to apply it consistently is still a challenge and requires considerable investment of training time. Thus, there is a need for a simple, objective and unambiguous system of classifying dung piles, which retains the flexibility of multi-stage classification systems and thus the option of manipulating decay times. An attractive possibility for such a system is to use dung height [Walsh and White 2005; Kuehl et al. 2007]. We could define dung height as the vertical distance between the ground on which a dung pile is sitting and the maximum height (or perhaps better still, diameter) of the tallest dung bolus in the pile. The decay threshold (i.e., the height below which a dung pile is considered to be decayed) can then be defined *post hoc* from the decay monitoring and transect data sets. Clearly, increasing the threshold height will increase the number of dung piles crossing the threshold, which will decrease the length of the dung decay monitoring period required, including the pre-survey periods required by so-called retrospective methods [see Sections 4.4.1 and 4.6.2.4; also see Section 4.6.2.5]. However, increasing the threshold height will also (1) increase the number of dung piles whose initial height is below the threshold height (e.g., those from infants and juveniles or the 'cow pat' type of dung pile that can result when elephants eat a lot of very moist fresh foliage and/or fruits) and (2) reduce the number of dung piles included in the transect data set. Hence, the essential problem is to optimise the efficiency gain accrued by reducing decay time with the precision loss resulting from smaller sample size [Kuehl et al. 2007]. We note, too, that for elephant dung, there is a possible confusion between height (which we suggest should be bolus diameter for boliform dung) and length if a bolus is standing on its end. This can result in the 'undecaying' problem discussed by Hedges (1993) and Hedges and Tyson (2002), see above, and will require care in the field. Nevertheless, given the advantages described above, we recommend that a dung height based decay system be evaluated for elephant dung in sites in Asia and Africa [see Chapter 9 for practical guidance].

4.4 Dung Decay Rate

4.4.1 The problem of highly variable decay rates, and the critical difference between prospective and retrospective methods of estimating decay rates

Dung surveys require data on the abundance of dung piles, defecation (dung production) rates and dung pile decay rates (sometimes referred to as disappearance rates). The last of these is usually the most problematic, and is the subject of this section. Please note that throughout this manual, the term 'decay' not 'disappearance' will be used to refer to the disappearance of dung piles irrespective of the means by which the process occurs. For example, dung piles that have been washed away by water, destroyed as a result of trampling or feeding activities of animals or broken-down (decayed) as a result of bacterial processes, are all considered to have 'decayed'.

The design of experiments to allow robust estimation of dung pile decay rates has received surprisingly little attention. Many elephant surveyors have not assessed dung decay rate at their sites, instead they have used data from other sites often many hundreds or even thousands of kilometres from where they were working [Hedges and Tyson 2002]. This is not to criticise these surveyors, because they often had no choice. Nevertheless, decay rates can be highly variable within and between sites (spatial heterogeneity) and over time (temporal heterogeneity). For example, the reciprocal of mean duration time, which is—or was—often used to calculate decay rates (Dawson 1990; Barnes and Barnes 1992), has been calculated from several studies in Africa and the results nicely illustrate the problem of between site variation. For northeastern Gabon, Barnes and Barnes obtained decay rates of 0.022–0.026, and for Lopé Reserve in central Gabon, White (1995) obtained a decay rate of 0.018. In nearby southwestern Cameroon, Nchanji and Plumptre (2001) obtained decay rates of 0.013 to 0.007, or 38.8% to 72.2% of the values from Gabon. Nchanji and Plumptre note that some of this variation may be due to differences in the researchers' determination of when dung piles have disappeared, which can be rather subjective depending on the classification system used and is itself a significant problem (see Section 4.3), but this cannot account for all the difference.

White's study shows that the fruit content of the elephants' diet is likely to have a large effect on the mean duration of dung piles. In addition, the work of White (1995), Barnes et al. (1997a), Nchanji and Plumptre (2001), Barnes et al. (2006) and Breuer and Hockemba (2007) clearly show that climate, and especially rainfall, irradiance and temperature, play a major role in determining dung decay rates. Thus inter-site differences in rainfall regime and elephant diet (especially the fruit content of the diet), and probably vegetation type, prevent simple extrapolations between sites and seasons. This has major implications for dung based elephant surveys and is a strong argument against the use of decay rates from other sites.

An additional powerful argument against the use of decay rates from other sites or seasons is provided by the work of Walsh and White (2005) and Kuehl et al. (2007). Their work was on Western Gorilla (*Gorilla gorilla*) nest and dung decay rates, respectively, but has obvious implications for elephant dung decay rates, too, given the importance of rainfall as a driver of dung decay rates (see above and Section 4.6.2.3). Walsh and White found, from their large-scale simulations over a 50-year time series, that each site often showed multi-year rainfall trends. For example, annual rainfall at Gamba in Gabon trended upwards in the 1990s but rainfall at Minkebe (also in Gabon) tended downwards. Such rainfall trends have the obvious potential to produce spurious trends in estimates of abundance if estimates of nest or dung decay rate are made at one point in time then applied to survey data collected at other times. Kuehl et al. found very substantial spatial heterogeneity in gorilla dung decay rates and, as they note, such spatial heterogeneity is not peculiar to gorilla dung but results from the fact that the environmental conditions that determine dung (and other sign) decay rates vary at virtually all spatial scales. They conclude that, '*the pervasive tendency to extrapolate decay rate estimates derived from studies in convenient locations* *to large survey zones is likely inducing large biases in abundance estimates for many large mammal species*'.

What is more, within site variation in dung decay rates is also very significant and is thus an argument against using decay rates from spatially unrepresentative samples of dung piles as well as against using decay rates from periods other than the survey period (e.g., from one or more years ago). Work in East Java revealed that Javan Deer (*Cervus timorensis*), Eurasian Wild Pig (*Sus scrofa*), Banteng (*Bos javanicus*) and Water Buffalo (*Bubalus arnee*) dung decay rates can vary tremendously over very small distances within the same vegetation type, and over short periods of time [Hedges and Meijaard, in prep.]. We conducted experiments during which we established, simultaneously, cohorts of dung piles (each cohort contained 20–30 separate dung piles) in plots 500–1000 m apart, but in the same vegetation type. We then monitored the decay of each pile until all piles had disappeared. These experiments showed, for example, that one of a pair of cohorts could disappear completely within 24 hours, while most of the dung piles in the other cohort of the pair might persist for two weeks. Furthermore, another pair of cohorts established, say, four days later, in the same locations would decay at very different rates to their immediate predecessors. We measured a number of variables at each site (rainfall, soil moisture, canopy cover, etc.) but the predictive power of these variables was disappointingly low [Hedges and Meijaard, in prep.]. Admittedly, the magnitude of this spatial and temporal heterogeneity in dung decay rate declined as the size (and 'cohesiveness') of the dung increased, i.e., the variation was greatest for species producing small pellets (e.g., deer), high–intermediate for wild pigs (big pellets, often with several adhering to each other), and lowest for the 'big pile' producers (Water Buffalo and Banteng) [Hedges and Mols unpub. data; Hedges and Meijaard in prep.]. Clearly, this is encouraging for elephant surveyors. However, we also found that the decay rate of Water Buffalo and Banteng dung was, at certain times of year, dependent on the availability of *Acacia nilotica* seed pods plus the number of wild pigs in the area (the pigs root through the dung looking for the *A. nilotica* seeds)—fascinating from an ecological point of view but a real problem for dung surveyors.

The few studies that have looked at within-site variation in decay rates for elephant dung have also found high levels of variation. For example, Plumptre and Harris (1995) found that the decay rates of African Elephant (and African Buffalo, *Syncerus caffer*) dung piles varied greatly within vegetation types and between different vegetation types over the year in Parc National des Volcans in Rwanda. While Nchanji and Plumptre (2001) found statistically significant differences in the mean duration of dung piles between different months and different seasons in Banyang-Mbo Wildlife

Sanctuary (BMWS) in south-western Cameroon. Environmental variables, with the exception of rainfall, were of little utility in predicting the duration of the dung piles. Rainfall in the month of deposition was significantly correlated with the square root of mean duration time; a similar result was also found by Barnes et al. (1997a) in southern Ghana. However, the sum of rainfall in the deposition month and the following month was a slightly better predictor in BMWS; also see Barnes and Dunn (2002). The duration of individual dung piles in BMWS was highly variable, even for those deposited a few metres apart on the same day. Mean duration ranged from 17 days for a dung pile deposited in July (wet season) to 234 days for a dung pile deposited in January (dry season). White (1995) also found significant differences in decay rates between months, but the differences did not follow simple dry season/wet season relationship even though the differences were related to climate. One of the principal determinants of elephant dung decay rates in White's study site in the Lopé Reserve in Gabon was the fruit content of the elephants' diet.

Highly variable elephant dung decay rates have also been found in Asia. For example, Wiles (1980) found that decay rates were four to six times higher in the wet season than the dry season at his study site in south-western Thailand. He did not look at within season variation, presumably because his sample size was too small. The results of a much more extensive study in southern Sumatra showed statistically very significant differences in the mean duration of dung piles within and between months [Hedges, unpub. data].

Most fundamentally, in almost all studies to date, dung piles have been monitored until they disappear, with the monitoring initiated *at the same time* as the dung surveys themselves. This approach, termed the *prospective* method by Laing et al. (2003) can lead to significant biases, as will be obvious when one considers that monitoring fresh dung piles until they disappear provides us with an estimate of the mean life span of freshly dropped dung piles, not the mean life span of the dung piles found during the survey (Marques et al. 2001; Laing et al. 2003). Fortunately, an alternative approach, increasingly known as the *retrospective* approach, has been developed [Laing et al. (2003); also see Hiby and Lovell (1991), Marques et al. (2001) and Buckland et al. (2001: 186–187)]. Retrospective estimates of dung decay require the surveyors to locate and mark cohorts of fresh dung piles by making a number of visits to the area prior to the survey. These marked dung piles are then revisited to establish whether they are still present or have decayed at the time of the survey. Logistic regression techniques can then be used to estimate probability of decay as a function of time, and possibly of other covariates, and the mean time to decay is estimated from this function [Buckland et al. 2001: 186–187; Laing et al. 2003]; free software is available for making these calculations [Appendix 4].

To avoid significantly biased estimates of elephant abundance, all dung surveys should therefore use the retrospective method of estimating dung pile decay rates [*sensu* Laing et al. (2003)]. This means that it is essential to conduct decay experiments prior to every survey, at every survey site. Moreover, the procedures for searching for fresh dung piles should aim to ensure that representative samples of the survey site's major vegetation types, rainfall zones and topography (slope) are obtained (see Sections 4.4.2 and 4.6.2.4 and, especially, Chapter 9 for practical guidance on this point).

There has been very little field experience with the retrospective method to date and so feedback is encouraged in order to improve the guidance provided here. However, recent elephant survey work that has used retrospective methods demonstrates the applicability of the method in at least some situations [Hedges et al. 2005; Hedges et al. 2007a; Hedges et al. 2008; Gumal et al. 2009; also see Hedges et al. in review]. Nevertheless, the retrospective method is very demanding of time and resources (manpower, money) because of the need to begin dung pile monitoring many months before the dung density survey and, especially, (2) the need for spatially representative samples of decaying dung piles. For these reasons, alternatives to dung counting methods such as fecal DNA based capture–recapture methods will often be preferable [Sections 4.6.2.4, 4.6.2.5, and 4.8; Chapters 2, 5, and 10].

4.4.2 Other issues

4.4.2.1 Creation of dung monitoring 'plots' versus monitoring *in situ*

Given the challenge inherent in finding and monitoring fresh dung piles in representative samples of the survey site's major vegetation types, rainfall zones and topography (slope) zones, there is a natural temptation to establish dung decay monitoring plots by moving freshly dropped dung piles to locations chosen to represent, say, different vegetation types. However, the decay rates in such plots may in fact not be truly representative of decay rates in those vegetation types. A particular concern here is the density of dung piles in a plot; for example, artificially high densities of dung piles may attract dung beetles and lead to higher than typical decay rates. A further issue is the loss of dung piles to flooding, such as in areas close to rivers. Though such plots are unlikely to be established in areas, dung piles will be dropped there. Thus the use of plots is likely to underestimate the decay (disappearance) rates of dung piles for the reasons given above (or conversely, creating plots may result in an overestimate of the decay rate due to the physical disturbances cased by moving the dung piles). The ideal is, therefore, to locate freshly dropped dung piles and monitor them *in situ*, but we recognise that the ideal may not be obtainable at all sites.

4.4.2.2 Monitoring only 'fresh' dung piles

It has been suggested that dung decay rates may be less variable while dung piles are fresh, and so it may be advantageous to count only those dung piles which are in the early stages of decay, for example in classes A to B of the Barnes and Jensen (1987) system [Nchanji and Plumptre 2001]. Moreover, by restricting our dung surveys to only relatively recently dropped dung piles (i.e., by classing a dung pile as having 'disappeared' quite early in the decay process), we can restrict the pre-density-survey decay rate monitoring period required by the retrospective approach [Sections 4.3.5, 4.4.1, 4.6.2.4 and 4.6.2.5]. However, as Nchanji and Plumptre note, even though the spread of decay periods around the mean decay time from stages A to B will be less than from D to E, with respect to the mean decay, the variance may not be very different. An additional problem, which they do not address, is that the number of recent dung piles found during surveys may well be prohibitively small, requiring too high a survey effort to achieve adequate precision [Barnes 2002; Chapter 3]. For example, in our elephant survey in Bukit Barisan Selatan National Park in Sumatra, Indonesia [Hedges et al. 2005], we found 2828 dung piles in classes A to D along 211.06 km of recces, but only 35 dung piles in classes A to B. These data (and data from Central Africa—F. Maisels pers comm 2010) suggest that restricting dung surveys to recently dropped dung piles is not a suitable method for dealing with high levels of variability in dung decay rates. An alternative to using only recent dung piles (classes A to B) may be to restrict analyses to dung piles in intermediate stages; for example, during the May–Nov 2001 survey we found 1100 dung piles in classes A to C1 and 1709 dung piles in classes A to C2 along 211.06 km of recces. This trade-off requires further attention, a conclusion also reached by Plumptre (2000); see also Sections 4.3.5 and 4.6.2.5.

4.4.3 Implications for dung count based surveys

What does the discussion above imply for surveyors wanting to use elephant dung count based methods to assess elephant abundance? The take home message from the studies reviewed above is that we cannot assume that a decay rate determined at a single plot in vegetation type A will be valid throughout all other patches of vegetation type A within a study site (let alone vegetation types A, B, C, etc. in another site thousands of kilometres away), or that a decay rate determined in January 2011 will be valid for March 2011 or indeed January 2012, as Hedges and Tyson (2002), Barnes et al. (2006) and others have also argued. Walsh and White (2005) and

Kuehl et al. (2007) reach the same conclusion from their study of gorilla nest and gorilla dung decay rates, respectively. This means that decay rates should be monitored at all sites where dung based survey methods are used to monitor elephant populations, with the monitoring beginning before (often many months before) every survey to estimate dung density (i.e., the retrospective method), and with spatially representative samples of fresh dung piles identified and included in the monitoring effort at regular intervals before the survey. Thus, sites with large elephant populations or sites where dung piles can be readily found are likely to be preferable. If it is simply not possible to monitor decay rates in a spatially unbiased manner in your area then you should refer to the alternative survey methods discussed elsewhere in this manual [Chapter 2 and Section 4.8; also see Section 4.6.2.5].

4.5 Defecation Rate

To estimate elephant population density from dung pile density requires knowledge of defecation rates (dung production rates) as well as dung decay rates. Defecation rate data for all elephant taxa, however, are scarce. Obtaining defecation rate data from wild elephants, particularly in forests, is difficult and potentially dangerous—although it can be done with the aid of experienced trackers [see, e.g.,Tchamba 1992; Theuerkauf and Ellenberg 2000; Nchanji et al. 2008]. In theory, captive elephants can be used to produce the necessary data but, in practice, there are serious concerns about the influence of a typical captive elephant's diet on its defecation rate.

Studies of forest-dwelling elephants in Africa have typically found defecation rates between 16.2 and 19.77 per 24 hours, with no evidence of seasonal variation (Merz 1986; Tchamba 1992; Theuerkauf and Ellenberg 2000; Table 4.4). However, a more recent study of forest elephants in south-western Cameroon by Nchanji et al. (2008) found significant monthly and inter-annual variation in defecation rates, with mean defecation rates higher for the wettest period of the year than for the relatively dry months. Nevertheless, the number of elephants in the Nchanji et al. study was small (four) and it is not clear how significant the statistical differences they found between seasons are in terms of their significance for dung count based survey methods since the seasonal defecation rates are not given.

The work of Tyson et al. (in review) suggests that for elephants in weakly seasonal environments in Asia, defecation rates are likely to fall within the same range as that found in African forest elephants. Tyson et al. used 10 tamed but wild-caught Asian Elephants to determine daily (24-hour) defecation rates. Fieldwork was carried out in Way Kambas National Park in Sumatra (in Indonesia) during the 2000 dry season and the 2001 wet and dry seasons. Elephants were allowed to forage naturally in the

TABLE 4.4 *Published defecation rates for Asian and African Elephants*[*]

Location	*Season*	*Forest type*	*Method*	*No. of elephants / sample size*	*Defecation rate per 24 hours*	*CV (%)*	*Reference*
Asian Elephants							
South-eastern Sri Lanka	?	N/A	O/C	37 / 129.2 elephant-hrs	15.0	74.2	Vancuylenberg (1977)
Parambikulam Wildlife Sanctuary, southern India	Post-monsoon	Moist-deciduous + wet evergreen	C	4 / 288 elephant-hrs	15.1	12.5	Dawson (1990)
Parambikulam Wildlife Sanctuary, southern India	Early dry	Moist-deciduous + wet evergreen	C	3 / 260 elephant-hrs	13.2	15.3	Dawson (1990)
Mudumalai Wildlife Sanctuary, southern India	Dry (mid)	Moist + dry deciduous	C	7 / 492 elephant-hrs	9.3	17.83	Dawson (1990)
Mudumalai Wildlife Sanctuary, southern India	Early monsoon	Moist + dry deciduous	C	6 / 156 elephant-hrs	13.3	32.0	Dawson (1990)
Mudumalai Wildlife Sanctuary, southern India	Monsoon	Moist + dry deciduous	C	6 / 115.5 elephant-hrs	14.6	13.9	Dawson (1990)
Mudumalai Wildlife Sanctuary, southern India	Post-monsoon	Moist + dry deciduous	C	7 / 148 elephant-hrs	15.9	20.7	Dawson (1990)
Mudumalai Wildlife Sanctuary, southern India	?	Moist + dry deciduous	O?	? / 88 hrs	16.33	?	Watve (1992)
Way Kambas National Park, SE Sumatra, 1994	Dry	Wet evergreen	C	5 / 20 elephant-days	11.83	10.99	Reilly (2002a)
Way Kambas National Park, SE Sumatra, 1997	Dry	Wet evergreen	C	12 / 50 elephant-days	13.04	15.84	Reilly (2002a)
Way Kambas National Park, SE Sumatra, 1998	Rainy	Wet evergreen	C	4 / 28 elephant-days	17.93	22.87	Reilly (2002a)
Way Kambas National Park, SE Sumatra, 2000 and 2001	Rainy and dry	Wet evergreen	C	12 / 1420 elephant-days	18.1	14.4	Tyson et al. (in review)

African Elephants							
Tsavo East National Park, Kenya	?	Savanna/woodland	C	4 / 308 elephant-hrs	17.1	33.0	Coe (1972)
Sengwa Wildlife Research Area, Rhodesia	Wet and dry	Savanna/woodland	T/O	48 / ?	14 males, 10 females	?	Guy (1975)
Ruaha National Park, Tanzania	Dry	Savanna/woodland	O	?	9.6	?	Barnes (1982)
Ruaha National Park, Tanzania	Wet	Savanna/woodland	O	?	32	?	Barnes (1982)
Kasunga National Park, Malawi	Dry	Savanna/woodland	T	147 elephant-hrs	15.7	?	Jachmann and Bell (1984)
Nazinga Game Ranch, Burkina Faso	Dry	Savanna/woodland	T	? / 88.2 elephant-hrs in total for both seasons	14.1	?	Jachmann (1991)
Nazinga Game Ranch, Burkina Faso	Wet	Savanna/woodland	T		27.2	?	Jachmann (1991)
Kibale, Uganda	?	Wet evergreen/ rainforest	O	132 / 400 elephant-hrs	17.0	3.4	Wing and Buss (1970)
Santchou Reserve, Cameroon	Wet and dry	Wet evergreen/ rainforest	T	? / 3091 elephant-hrs	19.77	4.2	Tchamba (1992)
Banyang-Mbo Wildlife Sanctuary, Cameroon	Wet and dry	Wet evergreen/ rainforest	T	4 / 6912 elephant-hrs	15.9	?	Nchanji et al. (2008)
Parc National des Volcans, Rwanda	?	Wet evergreen/ rainforest	T	Small	16.2	2.8	Plumptre (2000)
Bossematié Forest Reserve, Ivory Coast	Dry	Wet evergreen/ rainforest	T	33 / 88.5 elephant-hrs	16.6	106.6	Theuerkauf and Ellenberg (2000)
Bossematié Forest Reserve, Ivory Coast	Wet	Wet evergreen/ rainforest	T	59 / 137.5 elephant-hrs	18.1	79.6	Theuerkauf and Ellenberg (2000)
Tai National Park, Ivory Coast	?	Wet evergreen/ rainforest	T	?	18.0	1.2	Merz (1986)

* O = observation of wild elephants, C = observations on captive or tamed elephants, T = tracking wild elephants and counting dung piles, CV = coefficient of variation, ? = unknown.

forest during the day and were chained in areas of natural forage during the night. After eliminating incomplete data, arising from disturbances caused by flooding and wild elephant raids, Tyson et al. obtained 33 complete 24-hour periods for each of the 10 elephants in the first trial. The second and third trials produced data from 55 and 54 complete 24-hour periods, respectively. This study therefore has the largest sample size—by a large margin—of any elephant defecation rate study in either Africa or Asia (Table 4.4). Median defecation rate for the 10 elephants in each trial was 18 per 24 hrs; with an overall mean of 18.07 with 95% CI [17.93, 18.20] and a standard error of 0.0689. No seasonal or habitat-type (swamp grass, scrub and rainforest) related effect on defecation rate was found [Tyson et al. in review]. This lack of variation between season and between habitat types is encouraging, since for other species (primarily deer) a number of studies have shown that defecation rates are influenced by habitat quality and vary seasonally and with age- and sex-related differences in feeding behavior [Neff et al. 1965; e.g. Van Etten and Bennett 1965; Neff 1968; Dzieciolowski 1976; Mitchell et al. 1985; Rogers 1987; Mayle et al. 1996].

Furthermore, the work of Vancuylenberg (1977) and Watve (1992) in south-eastern Sri Lanka and southern India, respectively, also encourages the belief that defecation rates for wild Asian Elephants are likely to be between 16 to 18 per 24 hours (Table 4.4). Interestingly, Reilly (2002a) reported a figure of 17.93 per 24 hours for Asian Elephants in Way Kambas National Park in Sumatra during the wet season of 1998, but lower defecation rates in the dry seasons of 1994 and 1997 (11.83 per 24 hours in 1994, 13.04 per 24 hours in 1997). However, both 1994 and 1997 were El Niño Southern Oscillation (ENSO) years, and her site in Sumatra experienced a severe drought and extensive forest fires in both years. Under these circumstances, we would expect forage availability to be restricted with a concomitant reduction in the elephants' forage intake and defecation rates [Tyson et al. in review].

What do the results of these studies imply for dung count based estimates of elephant population size? Captive elephants can be used to obtain the necessary data in some areas, provided that the captive elephants are allowed to forage freely in typical elephant habitat [Tyson et al. in review]. These studies are expensive, however, and it seems unlikely that many further studies of defecation rate will be conducted in the near future although practical guidance for such studies is provided in Chapter 9. (The Tyson et al. study in Way Kambas NP, which was conducted with funding from WCS, USFWS/AsECF and WWF, cost about $21,000, mainly because of the large number of mahouts, field technicians and forest police who were involved). Moreover, no captive forest elephants exist in Central African forests. We need to ask, therefore, whether it would be appropriate to use the data from Sumatra and the Indian sites for other sites in Asia and the data from the African sites for other sites in Africa. We see three approaches to this problem and these are described below.

- *For areas where defecation rate studies have been conducted* Use the defecation rate data given in Table 4.4.
- *For forests in weakly seasonal areas with no available defecation rate data* Assume that (a) defecation rates do not show significant seasonal variation in forests (Merz 1986; Tchamba 1992; Theuerkauf and Ellenberg 2000; Tyson et al. in review) and (b) that a rate of 18.07 defecations per 24-hours (with standard error of 0.068918) is appropriate for forest sites in weakly-seasonal areas of Asia and Africa (see above).
- *For strongly seasonal areas with no available defecation rate data and for forest areas where the assumptions made above are considered inappropriate* There are no data from strongly seasonal areas of Southeast Asia, but such data do exist for southern India, and those data (together with those from Africa) suggest that defecation rates in these areas show major seasonal variation because of the large variations in the protein, fibre, and moisture content of elephant food stuffs (Guy 1975; Barnes 1982; Dawson 1990). For such seasonal areas, and for forest areas where the assumptions made above are considered inappropriate, we suggest that dung count data be corrected for dung pile decay rate but not for defecation rate, and that the resulting index of population density be used to evaluate trends. For this approach to be appropriate, all subsequent dung surveys would have to be conducted at the same time of year as the first survey, and there should be no significant intra-seasonal variation in defecation rate. Providing these conditions are met the indices of population density produced may be treated as direct analogues of absolute population density (Hedges and Tyson 2002; Tyson et al. in review). Another and probably superior alternative would be to use fecal DNA based capture–recapture methods (Chapters 5 and 10).

Even in the relatively non-seasonal equatorial regions, extreme climatic conditions, such as those caused by ENSO events, may have a pronounced effect on defecation rates of elephants in forests. It is recommended that surveyors of forest elephant populations who wish to calculate the number of elephants from dung surveys only count dung dropped during typical climatic conditions if they want to apply the 18.07 defecations per 24 hours rate recommended here or make comparisons between years as per the second option recommended above.

4.6 THE 'STEADY-STATE' APPROACH

4.6.1 What is it and is it useful?

The 'steady-state' approach was developed to facilitate relatively quick and computationally simple estimates of elephant population size from dung surveys [McClanahan 1986; Barnes and Jensen 1987]. In a system in which defecation rate and dung decay rate remain constant and the density of elephants is also constant, the amount of dung produced will equal the amount that disappears once the system is in equilibrium. It can be readily shown, either numerically, or by means of a simple simulation model, that once the system has achieved equilibrium (achieved a steady-state) the density of dung piles per square kilometre will remain constant. Thus the steady-state approach provides a simple method of converting estimates of dung pile density into elephant density, since:

$$E.D = Y.r, \text{ and therefore, } E = (Y.r)/D, \tag{4.1}$$

where E is the number of elephants per square kilometre, D is the number of droppings produced per elephant per day, Y is the number of droppings per square kilometre and r is the daily rate of dung decay.

The steady-state approach therefore has three requirements: defecation rate, decay rate and elephant density must all remain constant for sufficiently long periods to allow equilibrium dung pile density to be obtained. Furthermore, to be useful, the equilibrium period has to be long enough to allow surveyors to conduct a sufficient number of transects to estimate dung pile density with adequate precision. However, it is unusual for there to be sufficient data to assess whether any of the three requirements have been met. So, in practice, use of the steady-state approach requires surveyors to assume that the system is in equilibrium. Nevertheless, despite this need to rely on an untested assumption, the method is widely used, and while the assumption is usually acknowledged, the implications have rarely been tested, despite the cautionary remarks in McClanahan's paper.

Unfortunately, the few studies that have tested the assumptions of the steady-state approach suggest that equilibrium dung pile densities are rarely found:

- Plumptre and Harris (1995) showed that when elephants were present in their study site in Parc National des Volcans in Rwanda dung decay rates were very low, and consequently dung pile density continued to increase during the period that elephants were in the area, precluding the use of the steady-state approach. In addition, African Buffalo (*Syncerus caffer*) and Bushbuck (*Tragelaphus scriptus*) dung decay rates in the same area were also highly variable between months or seasons and vegetation types, which meant that use of the steady-state approach would have been invalid for these species too. They developed an iterative model to overcome this problem [Section 4.6.2.2].

- Barnes et al. (1997a) studied elephant dung decay rates in three sites in the lowland forests of southern Ghana, and showed that dung pile densities would vary from one month to the next, and between the same months in different years, even if elephant numbers and defecation rate were constant. They concluded that the validity of the steady-state approach must be questioned, and used their data to suggest a rainfall-based model that could be used as an alternative approach [Section 4.6.2.3].
- In elephant dung decay monitoring work conducted in the Pegu Yoma and Rakhine Yoma areas of Myanmar, Sukumar and colleagues found that there were periods when dung '*decayed at very slow rates (or hardly decayed at all) for several months and then disappeared abruptly, thus violating the steady-state assumption*' (Sukumar 1998).
- Nchanji and Plumptre (2001) studied dung decay rates in Banyang-Mbo Wildlife Sanctuary in south-western Cameroon, and their model showed that if there was no emigration of elephants from the population, there was a three-month period when the system was in a steady state. However, if emigration occurred during the wet season, as appeared to be the case, then the system was never in a steady state.
- Work by Hedges, Tyson, and colleagues in Bukit Barisan Selatan National Park in Sumatra (Indonesia) showed that elephant dung pile decay rates were too variable for the system to achieve steady state. Furthermore, use of the steady-state approach for a survey conducted in 2001 would have resulted in elephant density estimates that were higher than those derived using a non-steady-state approach—the DUNGSURV method of Hiby and Lovell (1991; Section 4.6.2.4)—by a factor of approximately 1.5. It was also clear that use of the steady-state approach for future surveys could indicate significant changes in elephant density in either an upward or downward direction even if the population remained stable [Hedges and Tyson 2002; Hedges et al. 2005].
- In a relevant (if non-elephant) simulation study, Walsh and White (2005) found very large fluctuations in the standing stock of gorilla nests at their sites and these fluctuation suggest that, for gorilla nests, the steady-state assumption fails in ways that are likely to induce substantial biases: both in 'one-off' survey estimates of gorilla abundance and—even more seriously—in trend-monitoring programs. As they note, '*the size of the biases observed in the simulation study, tens of percent, are of the same*

order as the gorilla abundance trends that monitoring programs can realistically be expected to detect. Furthermore, these estimates of bias are conservative in that they consider heterogeneity in only one factor that influences gorilla nest decay rate: rainfall. However, it is well known that other factors that influence gorilla nest decay rate (e.g., nest construction material) vary through time and space in ways that are not completely correlated with rainfall'.

Barnes et al. (1997a) argue that the steady-state approach was useful in the early days of dung counting, because it enabled surveyors to simplify a complex problem. However, as they add, '*it has always been a questionable* [method] *which introduced an error of unknown magnitude into the final estimate of elephant numbers.*' Moreover, it is important to note that estimates of elephant density derived using the approach are highly sensitive to the method of calculating dung decay rate [Barnes and Barnes 1992].

From the brief review above, it is clear that a steady-state is rarely if ever achieved, and that limitations of the method are now well-recognised. Fortunately, there are a number of alternatives to the steady-state approach to analysing dung count data, and these are reviewed below. Finally, we note that the MIKE program's *Dung Survey Standards* [Hedges and Lawson 2006] prescribe the use of non-steady-state approaches (and, specifically, the use of the method of Laing et al. 2003; Section 4.6.2.4).

4.6.2 Alternatives to the steady-state approach

In practice, the most problematic assumption of the steady-state approach is that of constant dung decay rates. Variation in defecation rates appears to be small, at least in forest environments [Merz 1986; Tchamba 1992; Theuerkauf and Ellenberg 2000; Tyson et al. in review; but see Nchanji et al. 2008; Section 4.5]. Immigration and emigration by elephants is a problem for all survey methods, and can be dealt with by conducting surveys at appropriate spatial scales and/or over appropriate survey periods. Fortunately, despite the popularity of the steady-state approach, several methods have been developed to eliminate or reduce the problems caused by variable dung decay rates: fecal accumulation methods [Section 4.6.2.1]; iterative models [Section 4.6.2.2]; rainfall-based models [Section 4.6.2.3]; and the retrospective methods of Hiby and Lovell (1991), Marques et al. (2001), Buckland et al. (2001: 183–186) and Laing et al. (2003), which are covered in Section 4.6.2.4.

4.6.2.1 Fecal accumulation methods (repeat counts)

One approach, which dates from the earliest days of dung surveys, is to conduct repeat counts along permanent transects (or in permanent plots) and count all new dung piles that have accumulated since the first count [Neff 1968; Bailey and Putman 1981; Jachmann and Bell 1984; Putman

1984; Plumptre and Harris 1995; Plumptre 2000; Nchanji and Plumptre 2001; also see Campbell et al. 2004 and Kuehl et al. 2007]. This approach is often referred to as the fecal accumulation rate (FAR) method—it is also sometimes called the accumulation method, the clearance plot method or the marked sign count—and was already introduced briefly in Section 4.1.1. Provided the counts are conducted at intervals of less than the minimum time for dung piles to disappear there is no need to correct the dung pile density estimates for decay, elephant densities are simply calculated from dung pile density, accumulation rate and defecation rate. Because this method reduces temporal bias by eliminating the steady-state assumption it has the potential to be more accurate than counting dung piles along transects (or in plots) which are not revisited. However, there are a number of serious problems with the method. Most seriously, the method is imprecise because sample size is determined by the number of recently deposited dung piles, which account for only a small proportion of the standing stock (i.e., the dung piles on the ground at the time of the survey). This means that FAR methods require much higher transect effort levels to gain a precise estimate of dung pile encounter rate than traditional one-visit (standing stock) methods [Section 4.1.1; but also see Section 4.6.2.5].

Repeat counts reduce the proportion of the survey area which can be covered too, since time which could be spent surveying new transects or plots is spent revisiting old ones. Minimum decay times are also rather short in many areas (days rather than weeks), which means that the number of new dung piles which will have accumulated by the time of the repeat surveys will be low and many transects or plots may have no new dung piles. This is again likely to lead to density estimates of low precision, especially when combined with the reduced number of transects that can be surveyed compared to one-visit methods. Indeed, it was the problems summarised above that lead to the development of the steady-state approach.

The use of permanent transects is also problematic in areas of dense vegetation, since elephants are likely to preferentially use the transects and so the estimates of dung pile density produced from counts along such transects are likely to be biased [Barnes and Jensen 1987; Barnes et al. 1995; Plumptre and Harris 1995; Varman et al. 1995; Barnes 1996; Buckland et al. 2001; Nchanji and Plumptre 2001]. Similar problems may be experienced with permanent plots too, as the process of searching the plot opens it up allowing animals in, or even attracting them to the area. This was a problem for Bushbuck dung counts in Rwanda [Plumptre and Harris 1995]. Finally, permanent transects also provide potential access routes for poachers [White and Edwards 2000; Buckland et al. 2001].

Because of the problems summarised above, fecal accumulation methods are not normally recommended for elephant surveys [Hedges and Lawson

2006]. However, recent work by Walsh and White (2005) and Kuehl et al. (2007) suggests a possible and seemingly promising new use for such methods—this is discussed in detail in Section 4.6.2.5.

4.6.2.2 Iterative models

Having shown that dung pile density in their study area was not in a steady state, Plumptre and Harris (1995) used a spreadsheet program to model iteratively the variation in dung decay. They constructed a matrix with the percentage of dung remaining in subsequent months for dung deposited in each month of the year. Therefore, for each month, the contribution to dung pile density from that month's depositions and previous months' depositions could be calculated by multiplying the percentages by the amount of dung deposited during the month to which the percentage refers. However, since they did know the amount of dung deposited per month, an initial educated guess was made for the amount of dung deposited per month per unit area per day (E_M), which was multiplied by the number of days in the month. These guesses were used to derive the dung pile density the model predicts would be found in a given month (NE). These density figures could then be compared with the actual dung pile density found in the month (NA) and a new E_M calculated by dividing NA by NE and multiplying by the initial E_M. The process was then repeated using the new value until the new E_M = the old E_M [see Plumptre and Harris (1995) for further detail].

The model is conceptually simple and easy to apply, given access to a spreadsheet program. However, there are a number of limitations. The application of the model described by Plumptre and Harris used data on dung decay rates gathered over one year, and then assumed that the decay rate of dung is the same at the same time of year each year, thus their method does not completely address the problem of temporal heterogeneity in dung decay rates. Plumptre and Harris acknowledged that this is not ideal, and indeed our data for deer, pig and bovid dung piles in East Java and for elephant dung piles in southern Sumatra show that decay rates are significantly different for the same months in different years [Hedges and Meijaard in prep.; Hedges and Tyson unpub. data]. Moreover, as Walsh and White (2005) point out, the Plumptre and Harris method ignores the problem of spatial heterogeneity in dung decay rates because studies conducted at only one or a few times and places are '*inherently incapable of doing a good job of predicting how rapidly indices* [dung/nests] *will decay at other times and places*'. For these reasons, we do not recommend using the Plumptre and Harris method [see instead the methods of Hiby and Lovell (1991) and Laing et al. (2003), which are described in Section 4.6.2.4].

4.6.2.3 Rainfall-based models

Another alternative to the steady-state approach is to use rainfall to predict expected dung pile density, and use this relationship to calculate elephant density from the estimate of actual dung pile density in the survey area, as suggested by Barnes et al. (1994; 1997a). Barnes and Dunn (2002) provide another example of this approach from their work in Liberia; their model is an improvement on that of Barnes et al. since it does not assume that rainfall accounts for all the variation in decay, but adds a stochastic element. Nchanji and Plumptre (2001) also found rainfall to be a reasonable predictor of mean dung decay times in their study site in south-western Cameroon. However, it is apparent that this method requires site-specific relationships between rainfall and dung decay rates to be developed. For example, White (1995) found that it was rainfall in the month of deposition plus the preceding two months that best predicted duration (he also found that fruit content of the elephants' diet in any given month was a better predictor of dung pile duration than rainfall); while Barnes et al. found that rainfall in the month of deposition was the best predictor in southern Ghana; and Nchanji and Plumptre found that rainfall during the month of deposition and the following month was the best predictor in south-western Cameroon.

A further complication is that the existing rainfall models assume that the dung pile density estimate was derived from transects that were surveyed simultaneously. This is clearly an unrealistic assumption and, as Barnes and Dunn note, this complicates the use of rainfall models. If this approach were to be used more widely, further work would be required to calibrate dung pile estimates from sets of transects surveyed in months with different rainfall totals [Barnes and Dunn 2002].

Furthermore, Walsh and White (2005) argue that even assuming that one can derive a rainfall model that explained a high proportion of variance in decay rate, the dung piles detectable at the time of the survey to estimate their density will include a variety of cohorts, each of which will have experienced a unique time series of environmental conditions different from that observed during the index decay study. Walsh and White's simulations suggest that, for long-lived sign such as elephant dung piles and ape nests, a single number such as the rainfall in the month before survey, is not likely to adequately capture the unique effects of each cohort's history.

Moreover, the work required to establish the site-specific relationships between expected dung pile density and rainfall is considerable and suggests that the retrospective methods of Hiby and Lovell (1991), Marques et al. (2001), Buckland et al. (2001: 183–186) and Laing et al. (2003), which are discussed in Section 4.6.2.4, are likely to be more efficient [also see Section 4.6.2.5].

4.6.2.4 'Retrospective' methods

As noted above, in most studies to date dung piles have been monitored until they disappear, and in many cases monitoring has been initiated at the same time as the dung surveys themselves [Section 4.4.1]. This approach, termed the 'prospective' method by Laing et al. (2003) can lead to significant biases (Buckland et al. 2001; Marques et al. 2001; Laing et al. 2003). Two alternative approaches, called 'retrospective' approaches by Laing et al., were developed independently by Hiby and Lovell (1991) and by Marques et al. (2001), Buckland et al. (2001: 186–187) and Laing et al. (2003).

The Hiby and Lovell (1991) method is '*based on the simple fact that the dung piles visible on a survey represent the remains of the dung piles deposited by elephants in the area over the weeks and months preceding the survey, whether or not steady-state has been reached*'. Their method relies on deriving a correction factor that relates observed dung pile density to the density of elephants, based on the probability of dung piles dropped prior to the survey still being visible during the survey:

$$Y(t) = E.F(t),$$

where $Y(t)$ = dung pile density at time t, E = elephant density, $F(t)$ = the correction factor relating dung pile density at time t to elephant density, and t is the time of the survey. Providing elephant density remains constant over a long enough period prior to the survey for the probability of dung piles remaining visible over the entire period to be negligible, a moment estimator for E, is simply

$$E = Y(t)/F(t).$$

Hiby and Lovell provided a free computer program, DUNGSURV, which calculates this estimator [Appendix 4]. However, if elephant density varies over the period prior to the survey, the estimate of E calculated by DUNGSURV is actually a weighted mean of the fluctuating density of elephants over the period. The DUNGSURV program provides information about the form of the weighting function. [For the derivation of $F(t)$ see Hiby and Lovell (1991).]

The Marques et al., Buckland et al. and Laing et al., retrospective approach is conceptually similar to that of Hiby and Lovell and was most fully developed by Laing et al. As with the Hiby and Lovell method, surveyors locate cohorts of fresh dung piles by making a number of regularly spaced visits ($\geq$ 6 visits) to the area prior to the survey; these marked dung piles are then revisited to establish whether they are still present or have decayed at the time of the survey. Logistic regression techniques are used to estimate probability of decay as a function of time, and possibly of other covariates, and the mean time to decay is estimated from this function [see Marques et al. (2001), Buckland et al. (2001: 186–187), and Laing et al. (2003); as already noted, free software is available for making these calculations (Appendix 4)].

There are several practical implications of these retrospective methods. Since retrospective methods aim to integrate decay rate spatial and temporal heterogeneity in the period preceding the survey, implementing them in an unbiased manner requires that the monitored dung piles are distributed representatively across a study area, not simply around a convenient location such as a national park ranger station or research centre [Buckland et al. 2001: 186–187; Laing et al. 2003; Hedges and Lawson 2006; Kuehl et al. 2007]. Ideally, this would involve a designed survey, for example one comprising several strip transects, randomly or systematically placed within the study area, to ensure that landscape/vegetation types are sampled in proportion to their occurrence [Buckland et al. 2001: 188; Laing et al. 2003]. In practice, obtaining this ideal is often too difficult but surveyors should attempt to include a spatially representative sample of dung piles in their monitoring experiments by, for example, searching for fresh dung in a number of well-distributed places across the survey area. For smaller survey areas with reasonable access, retrospective methods can be—and have been—used without prohibitive effort [e.g., Hedges et al. 2007a; Hedges et al. in review]. However, for large inaccessible areas, such as those in northern Myanmar, along the Thai/Myanmar border, in northern Sumatra, and in much of Central Africa, simply getting to remote sampling sites is often difficult, expensive and very time-consuming. It is worth noting here that it is possible to include additional covariates in retrospective models and this could turn out to be useful to account, at least partially, for spatial or landscape/vegetation type heterogeneity and thus lead to better estimates of decay rate and require less effort in the field [R. Burn pers comm 2010]. Unfortunately, there is no simplified version of the computation available (as yet) for the model with covariates.

Use of the retrospective method generally requires dung pile to be monitored for a lengthy period prior to the survey to ensure that the first cohorts of dung piles have disappeared. If significant numbers of dung piles from the first cohorts remain at the time of the survey the estimate of elephant density will be biased upwards. This can be a significant constraint on the use of retrospective methods. For example, pre-survey decay monitoring periods of about 10 months were needed for dung count based elephant surveys in Bukit Barisan Selatan National Park in Sumatra (Indonesia) and the Endau Rompin State Parks in Malaysia [Hedges et al. 2005; Gumal et al. 2009], an eight month period was required in Taman Negara National Park in Malaysia [Hedges et al. 2008], while about five months was required in the Nakai Plateau area of the Lao PDR [Hedges et al. 2007a]. When dung pile decay times are this long, establishing each cohort of dung piles will require either a permanent presence in, or a separate mission into, remote areas. Thus, for large and inaccessible sites, implementing retrospective methods in a spatially unbiased manner requires very high effort levels, and thus may also require considerable funds for field workers' salaries [Hedges et al. in review].

It is possible to reduce the pre-survey period over which dung piles need to be monitored prior to the survey by reducing the number of dung pile classes considered to be present, for example, by only including dung piles in the 'fresher' categories [Hedges and Tyson 2002; Hedges and Lawson 2006; also see Walsh and White 2005; Kuehl et al. 2007; Sections 4.3.5 and 4.6.2.5]. In the Bukit Barisan Selatan National Park example referred to above, including dung piles in only the 'freshest' four classes would have reduced the period over which dung piles need to be monitored prior to the survey from about 10 months to about five months (Hedges unpub. data). Since travel into remote areas represents a high proportion of the overall survey effort in large sites and landscapes, reducing decay time in this manner has the potential to significantly improve the efficiency (and reduce the cost) of retrospective methods. However, reducing the number of dung pile classes deemed to be present will increase the transect survey effort required to achieve satisfactory precision [Chapter 3]. Further work is required to evaluate this trade-off for elephant dung count based surveys, as we discussed in Section 4.3.5 [also see Section 4.6.2.5].

How many dung piles will need to be monitored and over what period? Buckland et al. (2001) and Laing et al. (2003) recommend that a time period is estimated over which 90% or more of the dung piles will be expected to have met the criterion that defines 'decay'. Ideally, this period would be estimated from past data from the study area. Searches for fresh dung piles should then commence this length of time ahead of the survey to estimate dung pile density. Note that if a time period that is too long is chosen, field costs will be higher than necessary but estimation of mean time to decay will not be compromised. However, if the pre-survey time period is too short, the estimate of mean time to decay will be biased, and this will result in a biased estimate of elephant abundance [Laing et al. 2003]. Laing et al. recommend that here should be at least five or six visits to the study area to search for fresh dung piles, roughly evenly spaced in time between the first visit and the subsequent survey from which dung pile density is estimated [see also Chapter 9]. Buckland et al. and Laing et al. argue that because variability in estimated decay rate will typically be smaller than variability in estimated density of dung piles, large sample sizes of fresh dung piles will not be needed, provided a spatially representative sample of dung piles is monitored; Buckland et al. suggest a minimum of 50 fresh dung piles should be identified and monitored in the pre-survey period.

Hiby and Lovell (1991) recommend that use of their DUNGSURV model requires a minimum of 100 dung piles, monitored over a period long enough to ensure all piles in the first cohort have disappeared by the time of the survey; and they show by approximate calculation that the contribution of decay rate estimation to overall variability in the elephant abundance estimate will be small if the number of monitored signs is about 100. They do not provide recommendations for the number of cohorts (which they call 'experiments') or the optimal interval between cohorts other than to suggest that '*experiments should be initiated frequently and fairly regularly. Long gaps between successive experiments should be avoided, particularly at times when decay rates may be changing rapidly*'.

Clearly more detailed guidelines could—and should—be derived by analysing the effect of differing cohort sizes and inter-cohort intervals using the large data sets available for elephant dung pile decay rates in Africa and Asia [e.g., those of White (1995), Barnes et al. (1997a), Nchanji and Plumptre (2001), Hedges et al. (2005; 2007a; 2008) and Gumal et al. (2009), as well as those of Nndjui Awo and Yaw Boafo in the Ziama forest (Guinea) and in Sapo NP (Liberia)].

Hiby and Lovell (1991) suggest that it is only necessary to return to the dung decay plots in order to assess how many are still present once, just before the survey. While it is true that no additional data are needed for their approach or that of Laing et al. (2003), we would argue that there are significant advantages to returning to the plots at regular (say, monthly) intervals in order to check that the identification tags needed to relocate the dung piles are still in place and to replace any that have been lost or damaged [Hedges and Lawson 2006; Chapter 9].

A further issue is that these retrospective methods assume that the dung pile density estimate was derived from transects that were surveyed simultaneously. Clearly, this is an unrealistic assumption, for example, our survey in Way Kambas National Park in Sumatra (Indonesia) involved 212.31 km of transects and required three months (Hedges et al. 2005). Nevertheless, this assumption is common to all dung based methods. As with all such surveys, the area should be surveyed as quickly as possible: '*it is better to use a large amount of manpower over a short period than a small amount over a long period*' [Hiby and Lovell 1991].

To conclude, despite the challenges discussed above we strongly urge all dung surveyors to use the retrospective approach to estimating dung decay rates and we note that the Laing et al. (2003) method is prescribed in the MIKE program's *Dung Survey Standards* [Hedges and Lawson 2006]. Practical guidance on the use of the Laing et al. methods is provided by Hedges and Lawson (2006) and in Chapter 9 of this manual. The prospective [Section 4.4.1] and steady-state approaches [Section 4.6.1] to incorporating decay rates into elephant density estimation should not be used because of the significant biases they are likely to introduce into your estimates of elephant density. If it is simply not possible to apply the retrospective method in a spatially unbiased manner in your area, then see the alternative survey methods discussed elsewhere in this manual [Section 4.8 and Chapter 2; also see Section 4.6.2.5].

4.6.2.5 Other methods: the Walsh and White / Kuehl et al. method

Walsh and White (2005) and Kuehl et al. (2007) propose a variant on the two-visit based methods discussed in Section 4.6.2.1, which aims to estimate the rate at which dung piles move across an objectively defined boundary, for instance a threshold dung height [see Section 4.3.5]. They argue that if the threshold is chosen so that (1) the threshold value is reached not long after dung deposition (i.e., to produce rapid decay rates) while only excluding a small proportion of dung piles and (2) the inter-visit interval is relatively short, then it may be reasonable to assume that the rate at which dung piles move across the threshold is equal to the rate at which they are deposited (i.e., the steady-state assumption is reasonable). Note that this method approach differs from the one proposed by Hiby and Lovell (1991) or Laing et al. (2003) discussed above in that (1) it is a prospective steady-state method rather than a retrospective method and (2) it does not require that dung detected during the first visit be fresh. Thus, by using all of the dung above a threshold height, sample sizes could be several times higher than those achieved using only fresh dung with concomitant gains in precision.

Nevertheless, the Walsh and White/Kuehl et al. two-visit method represents a similar bias reduction philosophy to that recommended by Buckland et al. (2001: 186–187) and Laing et al. (2003) when they call for a designed survey to estimate dung decay rates, for example one comprising strip transects, randomly or systematically placed within the study area, to ensure that landscape/vegetation types ('habitat types') are sampled in proportion to their occurrence. However, the Walsh and White/Kuehl et al. two-visit method is potentially a more attractive approach to estimating dung decay rates because there are serious doubts about the feasibility (logistical and financial) of a Buckland et al. or Laing et al. type 'designed survey' [Section 4.6.2.4]. While the Walsh and White/Kuehl et al. two-visit method showed clear promise for gorilla dung surveys [Kuehl et al. 2007], we have several concerns about the applicability of the Walsh and White/ and Kuehl et al. method for elephant surveys, which we discuss below.

Most seriously, because the Walsh and White/ and Kuehl et al. method is a 'prospective' steady-state method, it risks introducing the biases inherent to such methods. For example, if there is temporal heterogeneity in decay rates, which there almost always will be, prospective estimates are biased estimates of the required mean time to decay because they do not estimate the mean time to decay of the dung piles that are present at the time of the survey to estimate dung pile density (Laing et al. 2003; Section 4.4.1). Moreover, Kuehl et al. acknowledge that a problem of their prospective method is that dung pile age, which is an important predictor of decay probability (Laing et al. 2003), is unknown for the dung piles encountered on transects. However, their analyses suggest that most of the age dependence of dung decay probability applies later in the dung decay process—at least for gorilla dung in their study sites. Thus, they argue that setting a high threshold height may help to reduce the effect of age dependence.

While the Walsh and White/Kuehl et al. two-visit method addresses the inefficiencies of spatially representative retrospective methods or the biases that result from monitoring dung decay in a few likely unrepresentative areas and then using the resulting rates for the whole survey site, it does however introduce another source of bias by ignoring the dung decay process in the period prior to the survey (i.e., the period over which the dung piles found on the transects accumulated). What is unknown is just how significant a bias is introduced by assuming that the decay conditions following the detection of a dung pile are equivalent to those before, if dung decay rate is kept short by manipulating the decay (height) threshold.

Clearly, then, selecting an optimal dung height decay threshold is of crucial importance to the Walsh and White / Kuehl et al. two-visit method but we are concerned that for elephants it may not be possible to select a height threshold that produces sufficiently rapid decay time without excluding too large a proportion of dung piles: this needs to be evaluated in the field.

Another concern is that doubling the effort required per transect (because each transect is visited twice) results in half as many transects being surveyed per unit time. (Even though the method only requires the surveyor to relocate previously identified dung piles on the second visit the field time required is still likely to approach that required for the first visit, because most of the time in the field is taken with getting to an area.) This suggests that considerably fewer transects will be available for the estimation of dung pile density than would be the case if a standard one-visit per transect method were used. In addition, fewer transects will result in poorer spatial coverage of the survey site. We appreciate that considerable effort and time are saved by not needing to have a long period of pre-survey dung monitoring but we note that this does not result in more people being available for the transect surveys (and thus, an ability to conduct more surveys per unit time)—unless more people are hired—because very often the same people will be involved in both the pre-survey decay work and the survey itself.

Walsh and White and Kuehl et al. acknowledge that their two-visit per transect method requires more field effort to estimate dung pile density than the single visit methods traditionally used, but they argue that because sampling can be implemented in a representative manner so as to address spatial and temporal heterogeneity in the dung decay process, the extra field effort is well justified. What is more, they argue that the method also offers much improved precision if field workers estimate the relationship between accumulation rate or decay probability and remotely sensed environmental conditions at a representative subset of sampling sites visited twice, then interpolate values for other sampling sites given the prevailing environmental conditions and dung density during a single visit.

Finally, the Walsh and White/Kuehl et al. two-visit method is, by nature, a strip transect based method and precludes the use of line transect based methods. Thus it suffers from the relative inefficiencies of strip transects as well as the need to demonstrate that all dung piles within the strip transects are found by the surveyors [Chapter 3; Section 4.2.3]. Moreover, we are not convinced that a GPS coordinate or topofil distance along the transect will be sufficient to re-find the dung piles identified during the first visit (although use of new improved GPS units like the Garmin G60cx may remove this constraint). Topofil distances would probably be sufficient if permanent cut transects were used, but typically they are not for elephant dung pile surveys (for several good reasons, not least of which is that elephants like walking along cut transects resulting in biased dung pile density estimates). In practice, then, marking the dung piles will require use of metal stakes, painted marks on trees, etc. This is crucial since failing to find dung piles that are still present on the second visit and thus counting them as 'disappeared' will lead to erroneously high decay rates.

Nevertheless, given that the Walsh and White/Kuehl et al. two-visit method showed promise for gorilla dung surveys and is potentially a far more efficient way of addressing spatial and temporal heterogeneity in dung decay rates than the Buckland et al. or Laing et al. type retrospective 'designed surveys', it should be evaluated for elephant dung surveys. In particular, work is need to assess whether the bias involved in using a prospective steady-state method over a short period is small enough to justify the superior efficiency of the method.

4.7 Estimation of Elephant Age from Dung Dimensions

4.7.1 Introduction

As already discussed, knowing the age structure of elephant population is very helpful for estimating the impact of legal and illegal killing, captures or changes in habitat extent and quality [Sukumar 1989; Morrison et al. 2005]. Thus, it would add much value to dung surveys if in addition to producing estimates of population size they also produced information about population age structure, as this will help us understand population trends [Hedges and Tyson 2002; Hedges and Lawson 2006].

Three indirect methods for estimating elephant size or age from dung dimensions have been shown to be of value: dung mass [Coe 1972], bolus circumference [Jachmann and Bell 1979, 1984; Tyson et al. 2002] and bolus diameter [Reilly 2002b; Morgan and Lee 2003; Morrison et al. 2005]. With the exception of Morrison et al., who used known-age wild elephants, all these studies used either directly measured (semi-)captive elephants, or shoulder height estimates obtained from photogrammetry, to establish mathematical relationships between mean dung bolus diameter or circumference and elephant shoulder height.

Dung mass is impractical to measure during field surveys and the relationship between mass and elephant age demonstrated by Coe was derived from fresh dung, making Coe's method unsuitable for use during dung count based surveys since the majority of dung piles found will be many weeks or months old. Bolus circumference and diameter are, however, relatively easy to measure during surveys, and neither diameter nor circumference change appreciably with time for those boli which remain intact [Reilly 2002b; Tyson et al. 2002]. Reilly suggests that measurement of the greatest diameter is simpler and more precisely measured in the field. However, comparisons of bolus circumference and diameter showed that the coefficient of variation for circumference was always smaller than that for diameter [Tyson et al. 2002]. Morrison et al. (2005), noting that elephant dung boli approximate a cylinder with slightly elliptical ends, measured the long and short axes of the elliptical ends and took the mean of these two measures as the diameter for a bolus. They argued that this method is simpler than measuring circumference and potentially more accurate than using only maximum diameter.

It would appear, therefore, that valuable data on population age structure can be collected relatively easily during dung surveys if dung dimensions are measured. However, a number of limitations need to be recognised:

- For semi-captive Sumatran elephants, repeated measurements of dung size from several elephants over three 2-month periods demonstrated that mean circumference (and diameter) of boli showed large variations in a given individual [Tyson et al. 2002; but see Reilly 2002b]. However, for wild African Elephants in Amboseli National Park in Kenya, Morrison et al. (2005) found that dung boli produced by the same individuals within 6 months of each other did not have significantly different diameters.
- There are concerns about the use of captive elephants to derive curves relating elephant size or age to dung dimensions. Lindeque and van Jaarsveld (1993) suggest that captive elephants are likely to reach their growth plateaux much earlier than elephants living in the wild. This means that the elephants used to establish dung based ageing methods must be selected with care. Zoo animals are likely to be inappropriate, but wild-caught working elephants living in logging camps, for example, may be appropriate if they regularly forage freely in the areas around the camps—as in the Sumatran studies referred to above [Hedges et al. 2002; Reilly 2002b].
- When the sex of an elephant that produced a dung pile is unknown, and where sexual dimorphism in the population is pronounced, there is a problem in assigning an age class from dung size because the curves relating dropping dimensions to elephant size or age may differ for the two sexes [Jachmann and Bell 1984; Reilly 2002b; Tyson et al. 2002; Morrison et al. 2005]. In Jachmann and Bell's study of African Elephants, for example, the curves are different above 15 years of age: this means that for boli-size classes equivalent to this age and older, the defecations would have been produced by male and females of different ages, each sex being represented in proportion to the sex ratio of the age classes concerned. This problem is easy to deal with if the sex ratio of the population being studied is known, as it was in Jachmann and Bell's study site, but obviously for most forest sites it will not be known. However, Reilly (2002b) found no significant difference between the diameter of dung boli from males and females of the same age class of Sumatran elephant. Her results are encouraging, since they suggest that the relatively low level of sexual dimorphism shown by *Elephas maximus* (at least in Sumatra) compared to *Loxodonta africana* may facilitate the use of dung size based ageing techniques in the former species (or subspecies) in areas where sex ratios are unknown.

- A related issue is the problem potentially introduced by age- and/or sex-specific differences in defecation rates [e.g., Nchanji et al. 2008; Tyson et al. in review].
- Small boli may be overlooked (or have higher decay rates) potentially leading to underestimation of the number of juveniles in the population [Jachmann and Bell 1984]. For example, in Jachmann and Bell's site in Malawi, comparison of age-structures derived from photogrammetry and dung measurements showed a significant under-representation of calves under one year old in the dung counts. They calculated that this under-representation required a correction factor of 1.05 to be applied to the population estimate, with the 'extra' animals being added to the 0–1 year age class. The under-representation of calves is likely to be due to one or more of the following reasons: defecation rates of very young calves may be unusually low, particularly before they are consuming much vegetation; decay rates of young calves' dung piles are also likely to be significantly quicker than those of older animals (because of their higher surface area to volume ratio and milk-based diet); young calves' dung piles may be overlooked during surveys because of their small size [Jachmann and Bell 1984]. A possible approach to this problem might be to include dung circumference in the ancillary data collected during line-transect surveys, and stratify the data accordingly to allow the estimation of density estimates per dung pile size class (and thus, elephant size or age class). This would require surveyors to measure the circumference of two or three boli per dung pile in addition to recording the perpendicular distance from the transect line to the geometric centre of the dung pile. Clearly, this would increase the time required per transect but the additional data on population age- or size-structure might justify the extra effort. We feel this issue requires further attention, especially in light of the relatively low power of either sighting- or dung based counts to detect changes in elephant population size for small populations, and the concomitant need to pay more attention to changes in age-structure [Plumptre 2000; Barnes 2002].
- To reduce errors due to seasonal effects (e.g., dietary changes that might affect dung form), inter-year comparisons of age structures based on bolus size should only use data collected in the same season of each study year.
- Because of variation in size between boli from the same individuals, and because small changes in dung bolus diameter (and circumference) translate into potentially large differences in age as the growth curves approach an asymptote, the number of age or size classes which can be reliably distinguished from dung dimensions is likely to be relatively

small (probably ≤ 6 classes). Nevertheless, providing that broader age classes are used in assigning age as bolus diameter increases, the method can still be used to estimate the age-structure of elephant populations. For example, Morrison et al. (2005) recommend that all individuals above the age of 20–25 years should be grouped in one age class. The implications of this relatively low resolution for assessing trends in population age structure require further thought.

- Finally, a more general note of caution needs to be added to this discussion of the utility of dung based ageing methods. This concerns the interpretation of age ratios, which can be very misleading without other information on fertility and mortality rates for the two sexes [Caughley 1977; Sukumar 1989]. We do not dwell on this problem here, since it is not specific to dung based methods but a generally recognised problem in population biology.

4.7.2 Implications for dung count based surveys

Notwithstanding the problems discussed above, dung based ageing methods promise to become useful additions to the elephant surveyor's tool kit [Morrison et al. 2005; Hedges and Lawson 2006; Nowak et al. 2009]. Nevertheless, the following points should be borne in mind:

- Following Tyson et al. (2002), Hedges and Lawson (2006) recommended dung surveyors use dung bolus circumference as it appeared to be a more precise predictor of elephant size than dung diameter. Circumference is also easier and quicker to measure accurately in the field than dung diameter, and is likely to be less affected by deformation of dung boli resulting from impact with the ground [Tyson et al. 2002]. However, much of the work done on estimating elephant age from dung dimensions has been based on using dung bolus diameter. Nevertheless, since circumference and diameter are related to each other by the simple relationship, diameter = circumference / π, the age–dimension relationships based on diameter can still be used even when boli circumferences are measured in the field (for an example of this approach, see Morrison et al. (2005) who converted Jachmann and Bell's (1984) dung circumference data set into diameters to facilitate a comparison with their data).
- If possible, curves relating dung dimensions to elephant age or size should be developed for the region where the surveys will be conducted. Data presented in Jachmann and Bell (1984) illustrate the considerable variation in dropping size from different populations. Thus, differences in elephant body size between regions (e.g., the Asian mainland and Indonesia) are likely to lead to biases if curves relating dung dimensions to age or size developed in one region were used in another region.

However, the dung bolus diameter growth curve from Amboseli elephants was similar to that derived from another wild population of African Elephants (that in Kasungu National Park in Malawi, studied by Jachmann and Bell), suggesting that dung bolus diameter can, with caution, be used to assess age structure in areas where it is impossible to construct independent prediction curves of age and dung bolus provided that the asymptotic and lower dung diameters are comparable [Morrison et al. 2005]. Nevertheless, this inter-population variation suggests that further study of dung diameter/circumference–elephant age/size relationships is needed.

- An approach, which could be used to overcome the problem of unknown sex ratios in forest elephant populations, would be to combine dung measuring to determine age with DNA sampling from the same boli to determine sex. Clearly this method would not be appropriate for the ancillary data protocol for line transects discussed above and in Chapter 9, but it could be used as part of a dedicated dung based age- and sex-ratio assessment.
- If boli circumferences are not collected as ancillary data during the line transects surveyed for population size estimation, but are collected during stand-alone surveys, it will still be necessary to stratify the survey area in order to achieve representative coverage of areas used by different groups within the population (bachelor groups, maternal herds, etc.). Stratifying by expected elephant density will also facilitate more precise estimates of population age-structure.
- Measurements should only be taken from undamaged boli. The dry season is likely to be a better time to measure dung boli in many areas, because the relatively slow decay rates will allow larger sample sizes to be collected. Jachmann and Bell (1984) also suggest that the dry season is preferable because the fibrous nature of the elephants' diet and the more rapid desiccation at this time of year will reduce distortion and shrinkage.

Further practical guidance for use of dung dimensions to estimate elephant age during line transect based dung surveys is provided in Chapter 9.

4.8 Conclusions: When are dung count based surveys appropriate for Elephants?

4.8.1 Small populations and the problem of precision

As we saw in Section 4.1.2, dung counts can provide accurate estimates of elephant density and they can also provide more precise estimates than direct sighting based survey methods such as aerial surveys (see Barnes 2001). In addition, we saw in Chapter 3 and Section 4.2 that line transect methods are likely to be the most efficient, precise and unbiased method of estimating dung pile density and we follow Barnes and Jensen (1987), Buckland et al. (2001), Marques et al. (2001) and Hedges and Lawson (2006) in recommending dung surveyors use line transects for this purpose [but see Section 4.6.2.5]. Nevertheless, a feature of most sampling methods is an inverse relationship between precision and population size [Taylor and Gerrodette 1993; Barnes 2002]. In other words, sample counts of small populations usually have a large coefficient of variation (CV). For small elephant populations, only small numbers of dung piles will be recorded along transects. Thus, dung count based surveys of small populations of elephants (like many of those that exist in the forests of Southeast Asia and West Africa) will probably give poor estimates of little use for assessing population trends.

How small is too small? Unfortunately, it is difficult and likely to be misleading to try and formulate a rule of thumb of the kind 'dung counts should not be used where one suspects that elephant population density is less than n/km^2'. This is true for a number of reasons:

1. Dung pile density does not just depend on elephant density, it is also depends on dung decay rates. So, if decay rates are low dung pile density may still be sufficiently high to allow us to estimate the latter with adequate precision; but if dung takes a long time to decay a prohibitive amount of time may be needed to assess decay rates [Section 4.6.2.4].
2. Terrain is important. In difficult terrain, where it is very time consuming to conduct line transects, it may not be possible to complete a sufficient number of transects to achieve adequate precision in an appropriate time frame irrespective of elephant density (whereas, in areas of easy terrain, the same elephant density would lend itself to line transect surveys).
3. Dung count based estimates of elephant density depend on the precision of our estimates of dung decay and dung production rates, as well as our estimates of dung pile density, and so if our estimates of these other parameters are precise (e.g., with CV < 5%) then we can tolerate less precise estimates of dung pile density.
4. For most forest sites in Asia and Africa, the current 'estimates' of elephant abundance or density are in fact little more than guesses [Duckworth and Hedges 1998; Blake and Hedges 2004; Hedges 2006; Blanc et al. 2007], and may well be substantial underestimates of the true elephant population size [see, for example, Blouch and Haryanto 1984: 9; Blake and Hedges 2004: 1197; Hedges et al. 2005: 43–44]. Therefore, deciding whether dung count based survey methods are likely to be appropriate

on the basis of an existing 'estimate' may well preclude the use of dung count methods when they would in fact be appropriate.

For all these reasons, in the absence of good data for a site, we recommend conducting a pilot survey to estimate dung pile encounter rates and then using these estimates to determine whether dung counts are appropriate for the site in question. By appropriate we mean: 'is the estimated total line length required to achieve the target coefficient of variation achievable with the resources available and in the time available?' Pilot surveys are discussed in more detail in Chapters 3 and 9.

Clearly, before one can make these calculations one must also decide on an appropriate target CV for the estimate of dung pile density for the site in question. This decision will need to be made in collaboration with suitably experienced statisticians or other wildlife population monitoring experts. It is possible, for example, that a CV of 20–25% for the dung pile density estimate would be acceptable if the CV for the defecation rate and the site-specific dung decay rate were low (≤ 5%); if, however, they are significantly higher, then a target CV of ≤ 10% for the dung pile density will more likely be needed.

In those cases where dung count based surveys are not appropriate because they are unlikely to return sufficiently precise estimates, use other methods not dung counts. A similar conclusion was reached by Barnes (2002) from his analysis of the likely precision of dung counts and the expected power of dung based monitoring programs when elephant population size is low. In these situations, fecal DNA based capture–recapture methods are likely to be appropriate [Eggert et al. 2003; Hedges et al. 2007a; Hedges et al. 2007b; Hedges et al. in review; Chapters 5 and 10].

4.8.2 Dung decay rates and the problem of bias

To avoid the biases that can be created due to differences in field workers' determination of when dung piles have disappeared, which can be rather subjective, we urge dung surveyors to adopt the 'S system' that is described in the MIKE program's *Dung Survey Standards* [Hedges and Lawson 2006] and discussed in Section 4.3.4 of this manual. In addition, we strongly encourage dung surveyors to experiment with the dung height based system proposed by Walsh and White (2005) and Kuehl et al. (2007), which is discussed in section 4.3.5.

To avoid the serious biases that can be introduced by temporal and spatial heterogeneity in dung decay rates, we strongly urge all dung surveyors to use the retrospective method of estimating dung decay rates (Sections 4.4.1 and 4.6.2.4) and we note that the Laing et al. (2003) retrospective method is prescribed in the MIKE program's *Dung Survey Standards* (Hedges and Lawson 2006). Practical guidance on the use of the Laing et al. method is provided by Hedges and Lawson (2006) and in Chapter 9 of this manual. If it is simply not possible to apply the retrospective method in a spatially unbiased manner in your area because finding fresh dung piles is too difficult, the site is too large and/or inaccessible or time or other resources are too limited, then use one of the alternative survey methods discussed elsewhere in this manual (see especially Chapter 2). Once again, we note that fecal DNA based capture–recapture methods are likely to be appropriate [Eggert et al. 2003; Hedges et al. 2007a; Hedges et al. 2007b; Hedges et al. in review; Chapters 5 and 10].

Similar considerations apply to the problem posed by unknown defecation rates: in those situations where using published defecation rate data and the assumptions discussed in Section 4.5 are considered inappropriate, either (1) correct dung pile density data for dung pile decay rate but not for defecation rate, and use the resulting index of elephant population density to evaluate trends or (2) use another method such as the fecal DNA based capture–recapture methods mentioned above and described in detail in Chapters 5 and 10. For the first of these options to be appropriate, all subsequent dung surveys would have to be conducted at the same time of year as the first survey, and there should be no significant intra-seasonal variation in defecation rate. We suggest that the second option is superior.

4.8.3 Swamps and other inundated areas

Seasonally flooded areas can often be accommodated by appropriate timing of survey and dung decay monitoring work so that all fieldwork coincides with non-inundated periods.

Permanently flooded areas, however, pose real problems, as dung surveys cannot be conducted in flooded areas. Therefore, it likely that surveys in areas with significant areas (> 5%) of permanent swamp or similar landscape/vegetation types (which are used by elephants), will underestimate elephant density by an unknown amount. In such situations use one of the alternative survey methods discussed elsewhere in this manual (see especially Chapter 2).

4.10 References

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CHAPTER 5

Estimating Abundance and Other Demographic Parameters in Elephant Populations Using Capture–Recapture Sampling: Statistical Concepts

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5.1 Introduction

Capture–recapture (also known as mark–recapture or capture–mark–recapture) methods have a long history of application in wildlife biology, human demography and other fields [see Seber 1982; Wilson et al. 1996; Williams et al. 2002; Amstrup et al. 2005]. Capture–recapture methods were originally developed for situations in which it was possible to physically catch and mark animals with tags that permit individual identification (e.g., Silvy et al. 2005). However, the modelling of such data is very similar to that used when animals are identified from natural marks.

For naturally patterned animals such as Tigers (*Panthera tigris*) [Karanth and Nichols 1998, 2010], Leopards (*Panthera pardus*) [Henschel and Ray 2003], Snow Leopards (*Panthera uncia*) [Jackson et al. 2006], Jaguars (*Panthera onca*) [Wallace et al. 2004], Cheetahs (*Acinonyx jubatus*) [Marniwicke et al. 2008] and Hyenas (*Hyaena hyaena*) [Singh et al. 2010], individuals can be identified with certainty from such markings. Capture–recapture methods are frequently applied using photographs of individuals of such species [O'Connell et al.

2010]. Although elephants can be photographed and individually identified using a combination of natural characters (e.g., tusk shape and size, tail form, ear notches, etc.; Goswami et al. 2007; see Chapter 10 for details), the probabilities of misidentification present challenges because these marks are not as distinct or invariant as coat patterns. In addition, numbers of elephants (and elephant photographs) are likely to be much larger than for the carnivores listed above, and the increased numbers of photographs that need to be compared are expected to lead to greater uncertainties (see Chapter 10).

Another generally useful option for identifying individual elephants in replicated 'samples' drawn from wild populations is to use DNA obtained from their dung samples collected in the field. We term this genetic capture–recapture [GCR; for reviews see Mills et al. 2000; Waits 2004; Lukacs and Burnham 2005; Schwartz et al. 2007]. This approach has been successfully applied to elephants [Eggert et al. 2003, Hedges et al. 2007a, 2007b, in review] and can provide precise estimates of elephant population size as well as much other useful data about the populations in a relatively quick and cost effective manner [Hedges et al., in review]. Practical considerations for the conduct of dung surveys for the purpose of GCR are provided by Hedges and Lawson (2006) and in Chapter 10.

Standard capture–recapture models were developed assuming that individuals are always identified correctly. However, photographic and DNA samples can be characterised by uncertainties associated with individual identifications, and such uncertainties, when ignored, can lead to biased estimates of abundance and density [Mills et al. 2000, Creel et al. 2003, Lukacs and Burnham 2005]. Fortunately, model-based approaches to incorporating identification uncertainty into capture–recapture inference have been developed recently [Lukacs 2005; Lukacs and Burnham 2005; Lukacs et al. 2007; Yoshizaki 2007; Yoshizaki et al. 2009, 2011; Link et al., 2010].

Regardless of the exact origin of the detection data, capture–recapture sampling is based on multiple 'samples' consisting of detections of individually identifiable animals. These samples are assumed to have been drawn from the population of interest, which is of unknown size. The pattern of detections and non-detections for each animal across sampling periods during which the animal is known to be alive and present in the sampled population (e.g., because it was detected before and after those sampling periods) permit inference about detection probability, the probability of detecting an animal given that it is alive in the population of interest (see Chapter 1). As specified in equations (1.2) and (1.4), estimation of detection probability permits inference about abundance and various demographic parameters.

Thus, we advocate use of formal capture–recapture models for inference based on data from detections of individual animals at multiple sampling occasions. These models provide powerful tools that explicitly address the central problem of imperfect detections squarely, rather than ignore it or assume it away, as many other current approaches to wildlife monitoring end up doing. Furthermore, capture–recapture modelling and estimation methods have a long-established theoretical basis [Seber 1982; Williams et al. 2002; Amstrup et al. 2005; Royle and Dorazio 2008] and have undergone extensive validation via simulation studies and major field studies on a variety of species (see summary in Williams et al. 2002). Investigators also benefit from the extensive literature (including many useful syntheses), free software, helpful websites and list servers (Appendix 4). Because of these advantages, we strongly recommend approaches based on capture–recapture techniques to monitor elephant population size and trend and to study elephant population ecology.

5.2 CAPTURE–RECAPTURE MODELS

Capture–recapture models can be viewed as probabilistic expressions describing the processes that give rise to the observed capture–recapture data. Capture–recapture models are frequently classified according to requisite assumptions about population 'closure'. Closed population models are used when there are no gains to, or losses from, the population between sampling occasions.

We note that such losses or gains producing lack of closure can occur as a result of demographic processes (births and deaths) or movement (immigration or emigration). Because of this 'closure' assumption, closed capture–recapture models are generally applied to studies conducted over relatively short time periods, typically to estimate population size or density at a given point in time.

The basic data on which traditional capture–recapture inference is based are referred to as capture histories or detection histories. Each history is a row of K 1s (denoting capture/detection) and 0s (denoting no capture/detection), where K is the total number of sampling occasions. For example, a capture history of (0 1 0 1 1) indicates an individual animal that is caught in occasions two, four and five of a 5-period (K=5) study. These data are combined for all animals to form a capture history matrix or X matrix, as shown below.

In closed populations, every 0 in a capture history indicates a sampling occasion at which the animal was present but not captured. Probabilistic models are developed to describe the captures and non-captures as functions of capture probabilities. These capture probabilities are then used to estimate the number of animals in the population that were not captured (i.e., that exhibited capture histories of all 0s), and hence total population size. Traditional closed capture–recapture models (e.g., Otis et al. 1978, White et al. 1982, Chao and Huggins 2005a, b) such as those implemented in software programs CAPTURE [Rexstad and Burnham 1991] and MARK [White and Burnham 1999] thus provide estimates of abundance or population size.

For some purposes, it is useful to translate such abundance estimates into estimates of density, i.e., the number of animals per unit area. In the case of traditional capture–recapture approaches for closed populations, an additional estimate is needed of the area from which sampled animals are drawn. Conceptually, we would like to estimate density as the number of home range centres per unit area. However, capture–recapture abundance estimates are likely to include animals with range centres lying outside the sampled area that are nonetheless exposed to capture efforts in the sampled area. In traditional capture–recapture modelling, the effectively sampled area is estimated from ancillary spatial information about movement rates, such as data on capture locations or radio-telemetry data. Finally, density is computed by dividing the estimated population size by the estimate of sampled area (e.g., Wilson and Anderson 1985).

The majority of capture–recapture modelling has been based on such data indicating whether an animal was detected or not in the study area during each of the sample occasions. However, recent developments include spatially explicit capture–recapture models [Efford 2004; Efford et al. 2004, 2008, 2009; Royle and Young 2008; Royle et al. 2009a, b; Gardner et al. 2009, 2010; Royle and Gardner 2010]. Such models require not only an indication of whether or not an individual is detected at each occasion, but also the exact location of capture or detection (e.g., using a 2-dimensional coordinate system). For example, assume that the example capture history from above (0 1 0 1 1) represented captures on a 10 x 10 grid system, with each trap location specified by a row x column combination. If we included the information on capture location, we might rewrite this history as (0 [3,7] 0 [4,9] [4,10]). We have simply replaced each '1' in the capture history with a grid location specified in brackets (e.g., [3,7] indicates row three and column seven of the grid system). We still have a row of detection data for each individual, but the additional information on capture location permits direct inference about density using spatially explicit capture–recapture models.

Open capture–recapture models [Cormack 1964; Jolly 1965; Seber 1965; Pollock and Alpizar 2005; Nichols 2005] are used when there is potential for gains or losses occurring between sampling periods. Open capture–recapture models permit reliable estimation of apparent survival rates and recruitment that drive changes in the population. Recent developments for open models permit the direct estimation of rate of population change [Pradel 1996; Williams et al. 2002]. Gardner et al. (2010) and Royle and Gardner (2010) have developed initial open models for spatially explicit data. In addition to providing estimates of survival and recruitment, these models have the potential to allow inferences about changes in home range location and size to be drawn.

A combination of closed and open models, called the 'robust design' [Pollock 1982; Pollock and Kendall 1992; Kendall et al. 1995, 1997; Williams et al. 2002] is ideal for long term studies. In such studies, the population size (and density) are estimated each season (e.g., annually) using closed model data and then combined with open model data across seasons to obtain a reliable understanding of elephant population dynamics, even in the face of imperfect detections that are a major problem in field surveys of elephants. In such robust design studies there are two types of sampling occasions: the primary sampling occasions that occur once across each of the longer time intervals (say a year or so) when the population is open, and, the secondary sampling occasions that occur within each of the primary samples, during which population closure is assumed. The robust design permits inferences that are not possible with either closed or open designs alone (see Nichols and Pollock 1990; Kendall and Pollock 1992; Kendall et al. 1997).

The field data for all of these analyses are typically summarised in the form of capture histories of individually identified elephants based on physical captures, fecal DNA, photography or videography. In the following sections, we discuss basic concepts underlying both closed and open capture–recapture models for elephant population analysis.

5.2.1 Estimation of elephant abundance using closed capture–recapture models

5.2.1.1 Lincoln-Petersen 2-sample estimator

The Lincoln-Petersen estimator is simple to compute and provides an intuitive basis for understanding capture–recapture models and abundance (population size) estimation (also see Seber 1982, Williams et al. 2002). The estimator is based on a capture–recapture study conducted for two sampling occasions. However, in most field studies, it is advisable to obtain more than two samples.

We have noted above that, in general, capture histories provide a good way to summarise capture–recapture data. In 2-sample studies, only three capture histories can be observed: 10, 01, 11. Denote the number of animals detected in a 2-sample study that exhibit each of these histories as x_{10}, x_{01}

and x_{11}, respectively. Define the following statistics obtained directly from the collected data on captured (or identified) individual elephants as:

$n_1 = x_{10} + x_{11}$ = number of elephants caught, identified and released back on sampling occasion one,

$n_2 = x_{01} + x_{11}$ = number of elephants caught and released on occasion two, and

$m = x_{11}$ = number of elephants recaptured on occasion two which were previously caught on occasion one.

Define the unknown quantity of interest as:

N = total number of elephants in the sampled population.

Then define the following model parameters:

p_i = probability that an elephant exposed to sampling efforts in the sampled area is captured on occasion i, $i = 1, 2$ and

$p^* = 1 - (1-p_1)(1-p_2)$ = probability that a member of N is caught at least once during the study (so the probability of being caught at least once is 1 minus the probability of being missed [not caught] on both occasions).

Estimation proceeds by first estimating detection probabilities for the two sampling periods. This is accomplished by conditioning on the animals captured in one occasion and then finding what proportion of these were captured in the other occasion as well. The idea of 'conditioning' simply refers to using a set of animals as a starting point and focussing on them for the estimation. For example, in order to estimate p_1, we condition on the animals caught in period two (we know these were present at period one, because the population is closed) and find how many of these were caught in period one. We estimate p_2 in a similar manner and then estimate p using $\hat{p}_1$ and $\hat{p}_2$.

$$\hat{p}_1 = \frac{m}{n_2},\ \hat{p}_2 = \frac{m}{n_1}.$$

$$\hat{p}^* = 1 - (1-\hat{p}_1)(1-\hat{p}_2). \tag{5.1}$$

Recall that the canonical estimator (equation 1.2) is given as $\hat{N} = \dfrac{C}{\hat{p}}$, where C is the count statistic (number of individual elephants caught, in this case), and $\hat{p}$ is the estimate of the associated capture probability. If we base estimation on the count statistic from period one, then we obtain the following estimator:

$$\hat{N} = \frac{n_1}{\hat{p}_1} = \frac{n_1}{m/n_2} = \frac{n_1 n_2}{m}. \tag{5.2}$$

If we base estimation on the count statistic from period two, then we obtain the same estimator:

$$\hat{N} = \frac{n_2}{\hat{p}_2} = \frac{n_2}{m/n_1} = \frac{n_1 n_2}{m}. \tag{5.3}$$

Finally, if we base estimation on the summed count statistic from both periods one and two, then we again obtain the same estimator:

$$\hat{N} = \frac{n_1 + n_2 - m}{\hat{p}} = \frac{n_1 + n_2 - m}{1-(1-\hat{p}_1)(1-\hat{p}_2)} = \frac{n_1 n_2}{m}.$$

The final estimator of equations (5.2–5.4) is known as the Lincoln-Petersen estimator (e.g., Seber 1982). The following bias-adjusted estimator of Chapman (1951) is recommended for actual use, especially when sample sizes are small:

$$\hat{N} = \frac{(n_1 + 1)(n_2 + 1)}{m + 1} - 1. \tag{5.5}$$

An approximately unbiased estimator for the variance associated with the estimator of equation (5.5) is given by

$$\hat{\mathrm{var}}(\hat{N}) = \frac{(n_1 + 1)(n_2 + 1)(n_1 - m)(n_2 - m)}{(m + 1)^2 (m + 2)} \tag{5.6}$$

from Seber (1970, 1982). An approximate 95% confidence interval can be constructed for the estimate as:

$$\hat{N} \pm 1.96\hat{SE}(\hat{N})$$

This approximate confidence interval is based on the fact that $\hat{N}$ should be distributed approximately as normal with large sample sizes.

As a simple example of the Lincoln-Petersen estimator, consider a camera trapping study in an Indian wildlife reserve. Assume that 60 camera traps are placed along trails and at other suitable places (waterholes, mineral licks, etc.) and that photos of elephants are taken for two consecutive sampling occasions. The following statistics result from this study:

$n_1 = 7$ = elephants photographed on the first occasion,
$n_2 = 5$ = elephants photographed on the second occasion, and
$m = 2$ = elephants photographed on both occasions.

These statistics can be used to estimate capture probabilities and abundance.

Capture probabilities for each sampling occasion and for both sampling occasions combined are estimated as:

$$\hat{p}_1 = \frac{2}{5} = 0.40,$$

$$\hat{p}_2 = \frac{2}{7} \approx 0.29, \text{ and}$$

$$\hat{p} \approx 1-(1-0.40)(1-0.29) \approx 0.57,$$

based on equation 5.1. The Chapman estimator for abundance is:

$$\hat{N} = \frac{(7+1)(5+1)}{(2+1)} - 1 = 15,$$

with variance and standard error as

$$\hat{\text{var}}(\hat{N}) = \frac{((7+1)(5+1)\,(7-2)(5-2))}{(2+1)^2\,(2+2)} = 20, \text{ and}$$

$$\hat{SE}(\hat{N}) = \sqrt{20} \approx 4.47$$

We found that the estimated number of elephants in the sampled area is 15. We can compute an approximate 95% confidence interval on this estimate as:

$$\hat{N} \pm 1.96\hat{SE}(\hat{N}) \approx 15 \pm 8.76 = (6.24, 23.76)$$

Thus, the estimate is not very precise and there is a moderate degree of uncertainty associated with the estimate. This relative imprecision is to be expected with the small sample sizes that are likely to be characteristic of small isolated populations of elephants.

5.2.1.2 Multiple-sample closed capture–recapture models

Capture history data from studies with $K>2$ sample periods permit greater flexibility in modelling. Models for capture history data are based on parameters associated with the events that give rise to the capture histories. Models developed for capture histories from multiple or K-sample studies deal with three basic sources of variation in capture probability: time (sampling occasion), behaviour (trap response) and individual heterogeneity (for detailed treatment of the subject, please refer to Otis et al. 1978; White et al. 1982; Williams et al. 2002; Chao and Huggins 2005a, b). These probabilistic models thus incorporate different sets of assumptions about capture probability.

For example, model M_0 assumes that every elephant shows the same capture probability (p) for each sampling period (no variation in p). Model M_t assumes that capture probability varies by sampling occasion (p_i, $i = 1\ldots\ldots, K$). M_b assumes that capture probability differs for elephants that have (c) and have not (p) been caught previously. Consider capture history 101, indicating an elephant caught at periods 1 and 3 of a 3-period study. Example probability structures for capture history 101 under the above three models are given by:

M_0	M_t	M_b
$p(1-p)p$	$p_1(1-p_2)p_3$	$p(1-c)c$

Model M_h permits each individual in the sampled population to have a different capture probability. There are multiple approaches to estimating abundance under M_h (summarised by Chao and Huggins 2005a, b). One of these is known as the finite mixture model approach [Norris and Pollock 1996; Pledger 2000]. In its simplest form, this model attempts to model variation in detection probability by assuming two groups of animals with a different capture probability characterising each group. Define π as the proportion of animals in group one, characterised by capture probability p^1. Thus, the proportion of animals not in group 1 is given by $(1-\pi)$, and these animals have a different capture probability, p^2. Under this finite mixture model for heterogeneity, the probability structure for capture history 101 is

$$\pi\,(p^1[1-p^1]\,p^1) + (1-\pi)\,(p^2[1-p^2]\,p^2).$$

In addition to these models incorporating single sources of variation in *p*, there are models corresponding to all possible combinations of these sources of variation (M_{tb}, M_{th}, M_{bh}, M_{tbh}). Estimators for abundance are available under all models, although that for M_{tbh} requires ancillary data (on trapping effort). Most estimates are obtained iteratively, so computer programs are required for estimation. The program CAPTURE (Rexstad and Burnham 1991) computes estimates under all of these models except M_{tbh}, and the program MARK (White and Burnham 1999) can be used for these models as well (see Appendix 4).

The first step in the data analysis for a single season survey involves testing to see if the assumption of population closure during the survey was violated, using one or more of the 'closure tests' in programs CAPTURE or MARK. If the closure assumption appears reasonable, the analysis can proceed. If closure appears to have been violated seriously, then the investigator may have to use one of the open model analyses discussed below. The only disadvantage of using open models is that they do not permit robust estimation of abundance in the face of heterogeneity or trap response.

Model selection is based on goodness-of-fit and between-model tests. A discriminant function model selection procedure (based on simulated data) has been built into the program CAPTURE. Program MARK offers additional likelihood-based model selection tests and statistics, for example, Akaike's Information Criterion (AIC) (see Burnham and Anderson 2002).

Of all the estimators associated with the different models in CAPTURE, estimators based on model M_h tend to be relatively robust to deviations from underlying model assumptions. This is important because the model selection procedure in program CAPTURE requires large sample sizes in order to perform well. In the future, model weighting will likely be used to develop estimators [Stanley and Burnham 1998]. With model weighting, the final estimate is computed as the weighted mean of estimates produced by the various models in the model set, with the weights coming from AIC values or some other measure of the relative support for the different models (e.g., see Burnham and Anderson 2002).

At present, we know of no specific example of multiple-sample closed capture–recapture models for elephants. However, in order to better understand the kind of analysis that is possible, we describe Tiger camera trapping data from Kanha reserve in India. The area was sampled on 10 occasions during a survey period of 60 days. The total camera trapping effort was 803 trap-days and yielded 59 photo captures of 26 individual tigers. Capture histories for this study are presented in Table 5.1.

These capture histories were used with program CAPTURE [Rexstad and Burnham 1991] to compute statistics related to the closure assumption

TABLE 5.1 *Individual tiger capture histories from camera trapping conducted at Kanha Reserve, October–December 1995.*

Tiger identification code	Sampling occasion									
	1	2	3	4	5	6	7	8	9	10
KNT–101	1	0	0	1	0	0	0	1	1	0
KNT–102	1	0	0	0	0	0	0	1	0	1
KNT–103	1	1	0	1	0	0	0	0	1	1
KNT–104	0	1	1	0	0	0	0	1	1	1
KNT–105	0	1	0	1	0	0	0	0	0	1
KNT–106	0	1	0	0	0	0	0	0	0	0
KNT–107	0	0	1	0	0	1	1	0	0	0
KNT–108	0	0	1	1	0	0	0	0	0	0
KNT–109	0	0	0	1	0	0	0	0	1	1
KNT–110	0	0	0	1	1	0	0	1	1	0
KNT–111	0	0	0	1	0	0	1	0	0	1
KNT–112	0	0	0	0	1	0	0	0	0	0
KNT–113	0	0	0	1	0	0	0	0	0	0
KNT–114	1	0	0	1	0	0	0	0	0	0
KNT–115	0	0	0	1	0	0	0	0	0	0
KNT–116	0	0	0	1	1	0	0	0	0	0
KNT–117	0	0	0	1	0	0	1	0	0	0
KNT–118	0	0	0	0	1	1	0	0	0	0
KNT–119	0	0	0	0	1	0	0	0	0	0
KNT–120	0	0	0	0	0	1	0	0	0	0
KNT–121	1	0	0	0	1	0	1	0	0	0
KNT–123	0	1	0	0	0	0	0	0	0	0
KNT–124	0	0	0	1	0	0	0	0	0	1
KNT–125	0	0	0	0	0	1	0	0	0	0
KNT–126	0	0	0	0	0	0	0	0	0	1
KNT–127	0	0	0	0	1	0	0	0	0	0

and to model selection as well as model-based estimates of abundance. The closure test statistic from program CAPTURE provided no evidence that the closure assumption was violated ($z=0.57, P=0.72$). Thus, we concluded that the closed population models of CAPTURE were appropriate for these data and that we did not need to turn to models for open populations such as those of Pollock et al. (1990).

Based on the discriminant function model selection procedure, models M_0 and M_h were likely the most appropriate models for this data set. However, our *a priori* belief that there would be heterogeneity in capture probabilities among individual tigers because of social structure and unequal access to camera traps influenced our choice of the M_h model. Also, because of the robustness of the jackknife M_h estimator to deviations from model assumptions [Otis et al. 1978], we were more comfortable with the estimates from this model. Estimates under the above two models are presented below.

Capture probability estimate			*Abundance*		
Model	Per occasion	Entire study	$\hat{N}$	$\hat{SE}(\hat{N})$	95%CI
M_0	0.21	0.93	28	2.04	26-33
M_h	0.18	0.79	33	4.69	29-49

The estimates under these two models differ in predictable ways. First, in the presence of heterogeneity, estimates under model M_0 should be negatively biased and smaller than estimates under M_h [Otis et al. 1978; Pollock et al. 1990]. This expectation is reflected in the above estimates under the two models. Second, estimates under model M_h are expected to be much less precise than those under model M_0, and this expectation is reflected in the above estimates of standard error.

5.2.2 Estimation of elephant density using closed capture–recapture models

The above approaches (K-sample models implemented in program CAPTURE or MARK; the 2-sample Lincoln-Petersen estimator) provide estimates of abundance N and variance $var(\hat{N})$. Sometimes, we are also interested in elephant population density, $D = N/A$, where A is the area from which elephants are sampled. To estimate elephant density from the estimate of elephant abundance, we have to estimate the area that was effectively sampled during the survey. In the absence of hard boundaries to the survey area, elephants are likely to move in and out of the area actually covered by the DNA collection or photography such that the real area sampled is expected to be larger than the area physically covered by the sampling locations. Density estimation must attempt to deal with this uncertainty about area sampled [Williams et al. 2002].

The methods developed for estimating sampled area for standard trapping grids or trapping-webs cannot usually be used for elephant capture data because of the irregular geometry of the array of sampling locations (see, for example, Figures 10.1 and 10.3). In the early application of photographic capture–recapture sampling methods for Tigers and Leopards, Karanth and Nichols (1998, 2002) and others adapted an *ad hoc* approach, which used a buffer strip equal to expected 'half-home range length' around the polygon formed by the sampling locations in order to demarcate the sampled area. The mean of the maximum distance moved (MMDM) between any two photo-captures for each animal was frequently used to estimate the average home range length. However, this conventional closed model capture–recapture approach, which has several shortcomings, is now essentially rendered obsolete by the development of newer closed spatially explicit capture–recapture (SECR) models. We view the traditional approach as being useful only if one wants to compare current estimates to those obtained at other times or other places where new SECR models were not employed.

Spatial capture–recapture modelling approaches (e.g., Efford 2004; Borchers and Efford 2008; Royle et al. 2009a, b; Borchers 2010) have at least two distinct advantages over the 2-step approach to density estimation outlined above. First, SECR models can largely deal with the problems posed by individual heterogeneity in capture probabilities. Variation among individuals in capture probability can arise from the location of the individual animal's home range with reference to each sampling location (location of photo-captures, fecal DNA collection, etc.). SECR models estimate the numbers of centres of activity (one for each animal exposed to capture efforts) as latent variables, specifically modelling capture probability associated with any specific sampling location as a function of the distance between that location and each animal's activity centre. Thus, animals with activity centres that are distant from sampling locations (e.g., near the edge of the sampled area) have lower probabilities of being captured than animals with activity centres near sampling locations (e.g., near the middle of the sampled area). Spatially explicit capture–recapture explicitly incorporates such variation in capture probability associated with the spatial location of an animal and in doing so accounts for a major source of heterogeneity in capture–recapture studies. Secondly, SECR models present a formal single-step approach to simultaneously estimate abundance and density in a manner that is readily defended. Likelihood-based estimates of elephant densities can be obtained with user-friendly software program DENSITY [Efford et al. 2004] or from a Bayesian inferential approach to density estimation implemented in program SPACECAP [Singh et al. 2010]; see Appendix 4 for details of these programs. As noted above, the field data requirements for these spatially explicit analyses (SECR) are almost the same as for conventional capture–recapture surveys. The difference is that at each capture, the capture location (e.g., GPS location of where the image of the elephant or its dung sample was collected) is recorded and substituted for the usual “1” included in capture history data for traditional capture–recapture models.

5.2.3 Capture–recapture models for open populations

To understand elephant population dynamics more fully, in addition to abundance estimates obtained for each primary sampling period (say each year), several extra parameters such as survival, recruitment and temporary emigration (probability of an elephant not being present during some of the primary sampling periods) should also be estimated. In open population capture–recapture studies, the probability of observing a particular elephant capture history depends on several factors: the probability of capture, the probability of that individual surviving between primary sampling occasions and the probability of remaining within the sampled area. This survival is referred to as ‘apparent survival’ (ϕ), and its complement ($1–\phi$) combines losses due to death and permanent emigration.

The original Cormack–Jolly–Seber (CJS) model (first published by Cormack 1964) permits only estimation of apparent survival and capture probabilities. The Jolly–Seber model (JS), which includes an additional assumption of equal capture probabilities for caught and uncaught animals, can also estimate abundance [Williams et al. 2002]. New parameterisations of the Jolly-Seber model permit direct estimation of additional parameters of interest. For example, the ‘super-population’ approach of Schwarz and Arnason (1996) permits estimation of entry probabilities, corresponding to probabilities that animals enter the sampled population between any two sampling periods (see Goswami et al. 2007). This approach also includes estimation of the size of a super-population ‘N’, which is the total number of animals that are alive in the sampled area at some time during the study. The temporal symmetry models of Pradel (1996) include different parameterisations that permit direct estimation of (1) rate of population change, (2) seniority (probability that an animal alive at one sampling occasion was also alive during the previous sampling occasion) and (3) recruitment rate (new animals in one occasion divided by old animals the previous occasion). All of these parameterisations can be implemented using program MARK [White and Burnham 1999].

A problem with these earlier models for analyses of capture data is that they produce biased estimates of abundance in the presence of individual heterogeneity in capture probability and/or behavioral trap response. Fortunately, Pollock’s robust design largely overcomes this problem by integrating the sampling at two temporal scales: the primary sampling occasions separated, for example, by years and between which the population is assumed to be 'open' to gains and losses, and, within each of these, several secondary sampling occasions between which the population is typically assumed to be 'closed' to gains and losses [Pollock et al. 1990; Williams et al. 2002]. Robust design analyses estimate survival across the years using information similar to that used for CJS-type estimators. Most of the information used to estimate abundance each year is the same as that used by closed models. Thereafter, recruitment into the animal population can also be estimated by combining estimates of survival and time-specific abundance [Williams et al. 2002]. Kendall et al. (1995) proposed a likelihood-based robust design approach that simultaneously combines data from primary and secondary samples. Such joint modelling enables borrowing information across years, reducing the number of model parameters and increasing estimator precision.

As a result of all of these advances, even temporary emigration [Kendall et al. 1997; Schwarz and Stobo 1997] and transience [Pradel et al. 1997, Hines et al. 2003] can be estimated. We note that 'transience probability' can be viewed as the expected proportion (among all captures of new animals) of animals that have a near-zero probability of being recaptured in a subsequent primary period. Movement or migration of elephants is likely to be a function of study area size: in open study areas without hard edges, some individuals may be absent during some of the primary occasions simply because parts of their home ranges lie outside the trapped area. Furthermore, improvements in likelihood-based estimators that can incorporate individual heterogeneity in capture probabilities [Williams et al. 2002] have been very important for earlier studies on large mammals. Program MARK [White and Burnham 1999] offers a large suite of flexible models, including most of those mentioned above. Goodness-of-fit tests for robust design models are not fully developed. Thus, it is sometimes useful to conduct separate tests for the open and closed components of the full robust design model using such programs as CAPTURE [Rexstad and Burnham 1991], RELEASE [Burnham et al. 1987] or CloseTest (for the last two programs, see Williams et al. 2002 and Appendix 4).

At present, we know of no specific examples that exploit the robust design for drawing inferences about elephant population dynamics. In order to provide the reader with a concrete example of the kinds of analyses that are possible we note the paper by Karanth et al. (2006). These authors summarise the results of a 10-year robust design study of tiger population dynamics in Nagarahole National Park (India). They estimate survival, recruitment and abundance as well as parameters associated with temporary emigration and transience. This work clearly demonstrates the utility of approaches described here for assessing elephant population dynamics in future studies.

At the time of writing, spatially explicit versions of open models are being developed [Gardner et al. 2010; Royle and Gardner 2010]. We anticipate that these will soon be available to elephant biologists in an implementable manner.

5.3 Survey Design Considerations

An overview of the modelling and estimation approaches associated with photographic and DNA based capture–recapture data on elephants was provided above. A key issue in this regard is to design field surveys so that assumptions that underlie these capture–recapture modelling approaches are met fully or at least adequately. Serious violations of key model assumptions will render survey results valueless, however much effort is expended in the conduct of the surveys. Here we provide some general guidelines about temporal and spatial aspects of field capture–recapture sampling of elephant populations.

Temporal considerations: Duration of sampling periods A general objective in closed capture–recapture sampling of elephant populations is to keep sampling time 'short' relative to population turnover so that closed models can be used (in conjunction with secondary samples, in a robust design). However, the definition of 'short' will be situation-dependent and determined by the logistical considerations as well as elephant movement rates. It would be ideal to complete each season's (or year's) sampling within a period of 4–6 weeks, although given relatively slow turnover rates of elephant populations, periods of 8–12 weeks may also serve the purpose of meeting demographic closure (closure to births and deaths). Goswami et al. (2007) provide an example of a closed model study of this nature. In addition to demographic closure, animals moving in and out of the sampled area between sampling occasions also violate the closure assumption, unless such movement is completely at random (see Kendall 1999). For this reason, secondary samples within each year should occur at a season during which animals are relatively sedentary, as opposed to a season during which migration typically occurs.

In the context of assessing elephant population dynamics using open models, the vital rates of interest will include recruitment of calves into the population, recruitment of young into the breeder population, loss of tuskers to poaching and possible movement of animals from one subpopulation to another. Open models permit inferences about these parameters even in the face of transience. We believe the primary sampling periods must be at least 12 months apart, involving annual (or longer) surveys to get a reasonable handle on these parameters. Annual surveys are needed if focus is on population responses to weather or annual management decisions, whereas longer intervals between sampling (between primary periods) may be adequate for inferences about trends and average rates of survival and recruitment. Open models permit estimation of abundance, although such estimates are typically not robust to deviations from model assumptions [Williams et al. 2002]. Robust Design analyses [Pollock 1982; Pollock et al. 1990; Williams et al. 2002] permit robust estimation of temporary emigration. In general, we recommend the robust design for studies that require estimates of both abundance and the vital rates that bring about changes in abundance.

Spatial considerations Under traditional capture–recapture models for closed and open populations, a primary objective when considering space is that at each capture 'occasion', every individual elephant in the study area should have some non-negligible chance of being captured. Even models that incorporate heterogeneity (e.g., mixture models and other approaches to inference based on M_h) do not deal well with situations where capture probabilities approach zero for some individuals. Thus, investigators should seek to sample areas in such a way as to eliminate 'holes', i.e., areas within the overall sampled area within which some individual elephants may have zero or very low capture probabilities. In fact, the overall goal should be to achieve relatively similar capture probabilities for individual elephants in the sampled area.

The more recent SECR models for closed populations [Efford 2004; Borchers and Efford 2008; Royle et al. 2009a, b; Royle and Gardner 2010] are able to deal with potential holes in the sampled area. However, SECR models for open populations, including the robust design, will require that the same areas be surveyed during all of the primary occasions. If this is not the case, then it is not possible to separate inferences about population losses and gains from changes in the locations that are sampled.

Possible approaches in field surveys: process involved in obtaining captures of individual elephants Although stationary camera traps placed on trails where movement of target species is anticipated works very well for camera trapping carnivores (e.g., Karanth et al. 2010, 2011a, b), this is not a good way to obtain individually identifiable pictures of elephants in most cases. The camera (either triggered by an observer or, less commonly, by an automated device) needs to be used in day time to get good pictures (still or video) of elephant herds and individuals. Goswami et al. (2007) used either a vehicle or a stationary platform while photographing elephants. We note that the exact point process [Royle and Dorazio 2008] involved in the encounter between cameras and elephants differ for these two methods of photographic captures of elephants. In one case, both the elephant and the camera are moving around, in the second, the camera is stationary while the elephants are moving. In the case of collection of fecal DNA, the elephant is moving so its dung may be distributed over multiple locations, and the observer who picks up dung is moving as well. Assuming that both stationary traps and mobile investigators with cameras adequately sample the area of interest (e.g., there are holes) at each sampling occasion, these differences are not relevant to conventional capture–recapture analyses, but they would be to spatially explicit capture analyses. In the case of mobile investigators, the requirement for conventional capture–recapture to sample the area of interest on each sampling occasion means, for example, that the investigator cannot obtain photographs on one half of a study area one day and the other half another day and view these as two different sample occasions. Instead, an occasion must include sampling of the entire area of interest (so in the latter example, the two days would be combined to constitute a single sampling occasion). We believe that spatially explicit capture–recapture models can likely be adapted to deal with any of the sampling processes described above. Currently, spatially explicit capture–recapture models developed both under a classical likelihood-based statistical framework [Efford 2004; Borchers and Efford 2008] as well as under a Bayesian framework using Markov Chain Monte Carlo simulations [Royle and Young 2008, Royle et al. 2009 a, b] are available and have been implemented in software programs DENSITY [Efford et al. 2004] and SPACECAP [Singh et al. 2010] for estimating abundance or population size (see Appendix 4).

5.4 Conclusions

Photographic or DNA based capture–recapture sampling can provide a practical and rigorous way to estimate the abundance, density and other demographic parameters of wild elephant populations. These methods provide unbiased and more precise estimates (because there are fewer sources of variance) than many other methods including dung density based methods; they take less time to execute than dung density based methods (not least because there is no need for a lengthy pre-survey period monitoring dung decay rates) [Hedges et al. in review]; they provide more data about population parameters (sex ratio, genetic variability, effective population size, etc.); and they allow survival and recruitment rates to be estimated from repeated surveys. We strongly recommend the consideration and use of these modern sampling methods instead of the 'traditional census methods' that largely ignore critical issues of imperfect detections and thus produce biased estimates.

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CHAPTER 6

Estimating Distribution and Abundances of Elephant Populations from Sign Surveys at the Landscape Scale using Occupancy Modelling: Statistical Concepts

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6.1 Introduction: From Presence–Absence to Occupancy and Abundance Estimation

For logistical reasons, it is typically not possible to apply the intensive elephant monitoring methods of abundance estimation across large landscapes or regions, particularly in forested areas where aerial count surveys are impractical. Furthermore, in landscape scale surveys, investigators are typically compelled to rely on detection of signs left by elephants (tracks, dung, etc.) rather than visually detect or photograph elephants as required by sighting-based distance sampling and capture–recapture sampling methods, respectively (Chapters 3 and 5). Consequently, most sign survey methods (except dung based line transect surveys, see Chapters 4 and 9, which are not easily applied at the landscape scale) aim to estimate spatial distributions of elephant populations rather than their abundances.

Such landscape-scale studies of habitat occupancy are important for regional conservation planning. Moreover, habitat occupancy-related metrics are important for studies of elephant habitat selection [e.g., Linkie et al. 2006; Buij et al. 2007; Linkie et al. 2007; Martin et al. 2010]. However, some recent analytical advances in occupancy modelling methods show that under specific conditions even animal abundance can be potentially inferred from occupancy surveys, enabling estimation of abundance over large landscapes [Royle and Nichols 2003; Royle 2004; Stanley and Royle 2005; Conroy et al. 2008].

Most studies of elephant distributions, however, are still based on traditional 'presence versus absence' surveys that cannot separate true absence from non-detection ('false absences') of elephants. Because of imperfect detections traditional presence/absence surveys underestimate spatial range [MacKenzie et al. 2006]. The vital rates of occupancy dynamics, such as rates of local extinction and colonisation are also poorly estimated when using raw presence/absence data uncorrected for non-detection [Mackenzie et al. 2003, 2006]. Such false absences also have serious consequences for habitat models. For example, Tyre et al. (2003) and Gu and Swihart (2004) found that even modest levels of false absences caused estimates of habitat effects to be biased, particularly if detection probability varied between habitat types. Thus, inferences about the ecological values of different vegetation types can be highly misleading if detection probabilities are correlated with occupancy probabilities [Mackenzie and Royle 2005]. In a metapopulation analysis context, Moilanen (2002) concluded that false absences were a greater source of bias than inaccurately recorded patch sizes or presence of unknown habitat patches present within the study area.

Fortunately, the development of 'occupancy estimation' methods [MacKenzie et al. 2002; Royle and Nichols 2003; Mackenzie and Royle 2005; MacKenzie et al. 2006] has resolved this fundamental problem by permitting investigators to estimate probabilities of detecting animals, given their presence, using replicated surveys. Results of landscape level sign surveys using rigorous occupancy modelling have yielded promising results for Swift Foxes and Wolverines in North America [Sargaent et al. 2005; Magoun et al. 2007], Tigers [Hines et al. 2010; Karanth et al. 2011] and Four-horned Antelope in India [Krishna et al. 2008]. Such methods can clearly distinguish 'true absence' of elephants and other species from 'non-detection' given presence, and are thus valuable in contexts where more intensive abundance estimation methods are impractical.

Occupancy surveys can be viewed as being analogous to the capture–recapture surveys discussed in Chapter 5. However, here, instead of 'identified individual elephants' being captured, 'identified patches of elephant-occupied habitat' are detected by means of replicated field surveys.

Since elephant sign (e.g., dung piles and foot prints) are plentiful and easier to detect than the animals themselves, especially in forests, often it is advantageous to search for, detect and count 'habitat patches' containing such signs.

6.2 OCCUPANCY MODELLING AND ESTIMATION

6.2.1 Basic approach

Design of a survey to estimate the proportion of area occupied (habitat occupancy, spatial distribution) first requires division of the area of interest into sampling units. These units may be naturally defined, such as existing forest patches, or they may be defined by subdividing an area into equally sized grid cells (e.g., 10 km^2 each). Define the quantity of interest, occupancy, as:

$$\psi = N / S$$

where ψ = the probability that an individual site is occupied (it can also be viewed as the proportion of sample units occupied), N = number of sample units containing elephant sign and S = number of sample units in entire area. The proportion of sample units occupied can then be estimated as:

$$\hat{\psi} = \hat{N} / S \tag{6.1}$$

where S is known, but $\hat{N}$ (and thus $\hat{\psi}$) is not known and must be estimated.

As in most animal sampling situations (see Chapter 1), there are two sources of variation to consider in the estimation of N and ψ: spatial sampling and detectability. Spatial sampling is not an issue if all S cells are surveyed. Possible approaches to spatial sampling when all cells are not surveyed include simple random sampling, stratified random sampling and adaptive cluster sampling. Simple random sampling requires random selection of s cells for survey from the total number of S potential cells. Stratified random sampling first involves division of the entire area of interest into strata based on a factor such as habitat or expected elephant density. Then, simple random sampling of cells is used within each stratum. Allocation of samples to the different strata may be proportional to stratum size, proportional to expected elephant density, or according to some other scheme [Thompson 1992]. Adaptive cluster sampling begins with simple random sampling. The second step involves sampling of grid cells that border those cells in the initial sample that contained sign. The final step involves sampling all cells bordering cells sampled in the previous step that contained sign and continuing to sample in this manner until the cluster is surrounded by vacant cells [Thompson 1992; Thompson and Seber 1996].

Detectability (the problem of imperfect detection) is a source of variation in presence/absence surveys just as in surveys directed at animal abundance, because not all cells or sample plots containing elephant sign will be identified (because sign can be 'missed'). Elephant 'presence' is assumed to be determined because without error, such that detections of elephant sign are not confused with sign of other large herbivores and vice versa. Elephant 'absence' can reflect either true absence, or presence with non-detection. Different possible approaches exist for estimating detection probability, which we define as p = probability that elephant sign is detected, given elephant activity (hence possible sign presence) in the cell or sample plot.

In order to deal with non-detection, we again turn to the canonical estimator of the form $\hat{N} = C/\hat{p}$ presented in Chapter 1. However, now, C is the count of patches at which elephant sign was detected, p = the average probability of detecting elephant sign if elephants are present in a patch and $\hat{N}$ = the estimated number of patches that contain elephant sign.

The probability p of detecting elephant sign, given presence of elephants, is estimated from replicated field visits that are analogous to the different sampling occasions of a capture–recapture survey (Chapter 5). The detection data assume the usual X matrix structure (e.g., Table 5.1), with identified elephants being replaced by identified patches of occupied habitat, a '1' indicating that elephant sign was detected in that sample, and a '0' indicating non-detection.

The replicated samples are either temporally separated (different observers independently surveying a given patch) or spatially separated (the same observer surveying different parts of the patch). Since we know the total number of all patches, S, in the surveyed region (note that this is the key difference between occupancy and abundance estimation; in occupancy estimation the X matrix includes the sites at which no sign is detected), we can then go on to estimate the occupancy parameter ψ as:

$$\hat{\psi} = \hat{N}/S$$

where $\hat{N}$ is the estimated number of occupied sites. This two-step estimator was proposed by Nichols and Karanth (2002), whereas more recent work has focussed on the direct estimation and modelling of ψ [MacKenzie et al. 2006]. The occupancy parameter shown above can be viewed either as the probability that a randomly selected sample unit is occupied by elephants or the 'proportion of sample units occupied by elephants'.

6.2.2 Basic occupancy modelling

As noted above, in the case of temporal replication (repeat visits to sample units) the raw data for occupancy modelling are the rows of the X matrix, reflecting the detections or non-detections at each sampling occasion. For the estimation of occupancy in one season, the repeat visits are assumed to occur relatively close together in time to increase the likelihood of 'closure'. In the occupancy context, the closure assumption is that a site is either occupied by the species of interest, or not, for all of the sampling occasions within the season. The basic approach to estimation of occupancy from such data [MacKenzie *et al.* 2002, 2006] is very similar to that used in capture–recapture modelling for closed populations (Chapter 5). Specifically, a single row of the X matrix can be referred to as a detection history for a specific sample unit. Consider the detection history 0 1 0, indicating a site that is visited on three occasions with the species detected on occasion two, but not on occasions one or three.

The parameters required to model detection history data are:

p_{ij} = probability that elephant sign is detected at site i on occasion j, given that elephants are present at the site;

ψ_i = probability that elephants are present at site i.

Using the example of detection history given above for site i, h_i = 0 1 0, would be modelled as follows:

$$\Pr(h_i = 010) = \psi_i\,(1-p_{i1})\,p_{i2}\,(1-p_{i3})\,.$$

We know that the site is occupied, based on the detection in period two. The probability associated with this event (occupancy) is ψ_i. Given occupancy, we did not detect the species in periods one or three. The probabilities associated with these events are $(1 - p_{i1})$ and $(1 - p_{i3})$, respectively. The probability associated with the detection event in period two, is simply p_{i2}.

Now consider the more problematic detection history, 0 0 0. This represents a site at which elephants were not detected at any of the three occasions. Indeed, the entire problem of occupancy estimation can be viewed as one of estimating what fraction of these all-0 detection histories represent sites that were occupied. The probability associated with this history is:

$$\Pr(h_i = 000) = \psi_i\,(1-p_{i1})(1-p_{i2})(1-p_{i3}) + (1-\psi_i)\,.$$

In this case, we have written down two additive probabilities associated with two possible events. The first possibility is given by the left-hand term of the sum. The species may have been present (ψ_i), but was not detected at any of the sampling occasions $[(1-p_{i1})(1-p_{i2})(1-p_{i3})]$. The other possibility is simply that the species was not present $(1-\psi_i)$. In this case we need no detection parameters, because the species was not present and available to be detected.

Such probability modelling permits estimation of parameters of interest. Specifically, the raw data are the number of sites exhibiting each possible detection history, whereas the above modelling approach provides a probabilistic model for each history. Maximum likelihood then provides one approach to combining the data and the model in order to obtain estimates of model parameters, in this case the detection probabilities and the occupancy parameters [e.g., MacKenzie et al. 2002, 2006]. Such estimates are computed by programs PRESENCE [Hines 2008] and MARK [White and Burnham 1999]. In addition, Bayesian computational approaches can also be used to obtain these estimates [Royle and Dorazio 2008].

We note that in the absence of additional information, it is not possible to estimate a separate occupancy probability for each site. Thus, a single estimate of occupancy can be obtained for the group of sites representing a stratum of interest. However, if site-specific covariates thought to influence occupancy (or detection probability) are collected during the sampling (e.g., habitat variables, distance to water or distance to nearest human settlement) then it is possible to estimate occupancy probabilities specific to any set of covariates. This ability to directly draw inferences about determinants of occupancy is extremely useful and will be relevant to many ecological investigations [e.g., Krishna et al. 2008; Karanth et al. 2009, 2011].

6.2.3 Assumptions

The above single-season model is based on four assumptions. The first is that each sample unit is closed to changes in occupancy for all of the sampling occasions within the season, that is, a unit is either occupied or not for all of the sampling occasions. If sample units are small relative to an animal's daily range, then this assumption is likely to be violated, i.e., an animal will likely be present on some days and not on others. However, if the presence or absence of at least one animal is the result of a random process (at least one animal will be present on each sampling day with some underlying probability) then the resulting occupancy estimate will reflect animal usage of the sites rather than physical presence (see discussion in Mackenzie et al. 2006 and later in this chapter). If elephant sign is used as the basis for detection, the closure assumption may be more easily met, as sign (e.g., fresh dung) is available for detection for some period after deposition.

The second assumption is that the probability of occupancy is constant across sample units, or that variation in occupancy is modelled as a function of site-specific covariates. Substantial variation among units in occupancy beyond that accounted for via measured covariates will result in violation of this assumption. In this case, mixed models of the type discussed under closed capture–recapture modelling (Chapter 5) can be used [MacKenzie et al. 2006]. The third assumption is similar to the second and states that there is no un-modelled variation in detection probabilities. These probabilities can be modelled as functions of site-specific and time-specific covariates, providing many opportunities to adequately deal with this assumption.

The final assumption underlying the above modelling approach is that detections and detection histories at each location are independent. Violation of this assumption may only result in estimated variances that are too small, or it may produce biased estimates of occupancy, depending on the type of dependence. However, it will frequently be possible to either modify field procedures or develop models to deal with specific forms of dependence, if the form can be identified [see Mackenzie et al. 2006; Hines et al. 2010]. For example, in the case of animal sign detected by temporally repeated visits by the same investigator, it is likely that dependent detections will result, if investigators do not modify field procedures. Assume that an investigator detects fresh dung on the first sampling occasion. During a visit two days later, the investigator may recall the location of the previously detected dung pile and return to record another detection. This would clearly violate the assumption of independent detections.

At least two approaches are possible to deal with this problem. For example, the investigator could mark (even if only via a GPS) each dung pile detected at each sampling occasion and only record previously unrecorded dung at each sampling occasion following the first. Use of different observers on the different sampling occasions would (assuming sufficiently precise GPS locations are possible with the equipment available) also eliminate this problem. It is also possible to simply modify the modelling approach to deal with this non-independence. This can be accomplished by using a model that is similar to the trap-response model for closed capture–recapture modelling [Chapter 5]. Detection probabilities for sites at which detections have not occurred on a previous sampling occasion are allowed to differ from detection probabilities at sites on which previous detections have occurred.

An alternative approach to replication, more practical to the conduct of surveys of large mammals such as elephants, is by means of spatial replication. Instead of replicate surveys conducted at different times, replicate visits are made to different parts of the cell by the same observer. If these spatially replicated surveys are designed properly [Kendall and White 2009], they have great utility as demonstrated in the case of Tiger surveys recently [Hines et al. 2010; Karanth et al., 2011]. Even more importantly, the lack of independence between detections made on different spatial replicates, which was a thorny problem, has recently been explicitly addressed through the development [Hines et al. 2010] and application of new occupancy models [Karanth et al., 2011].

Assumptions underlie all ecological models, and consideration of them is important. As discussed above, there are usually multiple ways of dealing with potential assumption violations, including development of accommodating models and/or modification of field procedures. The goodness-of-fit test of MacKenzie and Bailey (2004) can be used to test for violations in underlying model assumptions and is available in program PRESENCE [Hines 2008; see Appendix 4].

6.2.4 Multi-season (multi-year) occupancy models

Closed models such as those discussed above are used to estimate occupancy and to provide inferences about ecological or management factors (e.g., habitat variables and human disturbance) that affect occupancy. In some cases, investigators will be interested in modelling changes in occupancy over time. One approach to such modelling is simply to use single-season models to obtain estimates of occupancy for each year of the period of interest. Various metrics expressing change in occupancy over time can then be computed from these year-specific estimates [MacKenzie et al. 2006].

MacKenzie et al. (2003, 2006) developed an alternative multi-season modelling approach that explicitly models the underlying dynamics. Under this approach, dynamics are decomposed into those dealing with the distinct processes of local extinction and local colonisation. A site that is occupied in season or year t may be either occupied in year $t+1$ or not. This process is governed by the local extinction probability, Pr(not occupied in $t+1$ | occupied in t) $= \varepsilon t$. A site that is not occupied in season or year t may also be either occupied in year $t+1$ or not. This probability is governed by the probability of local colonisation, Pr(occupied in $t+1$ | not occupied in t) $= \gamma t$. This decomposition of occupancy change into components associated with extinction and colonisation has two advantages. First, it relaxes an implicit assumption of the first approach described above, allowing the probability of occupancy in one year to differ between sites that were and were not occupied the previous year. Second, it permits us to focus directly on the processes associated with change. So, instead of modelling elephant occupancy as a function of human disturbance, for example, we can draw direct inferences about effects of disturbance on local extinction and colonisation, the primary determinants of future occupancy.

Data for multi-season modelling can again be expressed as detection histories. In this case, the histories follow the robust design format

[MacKenzie et al. 2003, 2006], reflecting the two temporal scales at which sampling occurs. For example, consider detection history: 01 00 11. This detection history covers three seasons or years or primary sampling occasions (the three groups of two numbers each), with two secondary sampling occasions occurring within each season or year or primary sampling occasion. So if the primary sampling occasions represent years, then the above detection history would represent a site at which the species was not detected on occasion one, but detected on occasion two, of year one (01); was not detected on either occasions of year two (00); and was detected on both occasions of year three (11). Modelling of multi-season data uses these detection histories, in conjunction with probability models, just as for the single season models. However, now the parameters required for this modelling include not only the occupancy and detection probability parameters, but also the extinction and colonisation parameters. These latter two parameters underlie the dynamic process of changes in occupancy and can be modelled as functions of site-specific and time-specific covariates, thus permitting strong inference about occupancy dynamics.

6.2.5 Survey design considerations

Various aspects of designing single-season occupancy studies are discussed in detail by MacKenzie and Royle (2005) and MacKenzie et al. (2006). We refer the reader to these sources and here focus on a few design issues likely to be of specific relevance to those modelling occupancy in elephant populations. If inferences about elephant distributions over a large area are of interest, then it may be useful to begin an investigation by defining a survey region of interest and then dividing it into a number of grid cells or discrete units (not necessarily of equal size) of potential habitat, depending on the area of interest. This step can be accomplished using landscape ecological tools (e.g., land cover or habitat maps, GIS and prior field knowledge).

If the cells / habitat patches are large in relation to expected maximum home range size, then elephant signs encountered in different habitat patches are likely to come from different individuals or clusters of individuals. The occupancy parameter in such cases should reflect elephant presence in the sample unit. If the home range sizes are large relative to patch or grid cell size, then a home range may cover several habitat patches, and the occupancy parameter can be considered as a measure of 'intensity of habitat use' [see MacKenzie and Royle 2005]. Hence, the choice of survey site (grid cell) size needs to be guided by knowledge of likely home range size in order to distinguish true occupancy from habitat use.

Although dependent on study objectives, our usual preference is for estimating occupancy by using 'large' cells in relation to home range size. The potential advantage of this approach is that some recent model development [Royle and Nichols 2003; Royle 2004; Stanley and Royle 2005; Conroy et al. 2008] might eventually permit linking of elephant abundance to occupancy under this scenario as discussed below.

In the combined survey protocols developed for the Wildlife Conservation Society's habitat occupancy studies for Tigers and Asian Elephants, cell sizes of about 200 km^2 were used in southwestern India and 400 km^2 in Southeast Asia, based on expected largest home ranges for these species. An example map of the study region, elephant habitat and pre-defined patches used in an occupancy survey of elephants in the Malnad-Mysore landscape in Karnataka, India, is provided in Chapter 11.

In theory, field surveys of occupancy can be conducted in a subset of all potential cells (e.g., relying on spatial sampling schemes such as random, stratified-random or adaptive sampling). Such incomplete spatial sampling still permits extrapolation of the results to non-surveyed areas and mapping of distributions. Such extrapolation and mapping are facilitated by modelling occupancy as a function of site-specific covariates associated with habitat variables, human disturbance and the like. Occupancy models, however, perform best when sample sizes are large, providing a reason for surveying all cells. Detailed recommendations about required sample sizes, including number of sample units and number of replicate visits per sample unit, are provided by MacKenzie and Royle (2005) and MacKenzie et al. (2006).

Design recommendations for multi-season studies include many of the issues relevant to single-season occupancy studies, as well as additional issues that are only relevant to longer studies that focus on dynamics. Bailey et al. (2007) discuss the program GENPRES, which was developed for use in investigating design issues for both single-season and multi-season occupancy models. This software permits the user to specify underlying design parameters, generate associated occupancy data, and then assess such issues as estimator bias and precision, as well as performance of model selection statistics, under various designs. For example, an issue that arises in the study of occupancy dynamics is whether to survey the same sites or sample units season after season or whether instead to increase geographic coverage by selecting different sites each year. If the focus is on dynamics and underlying rates of local extinction and colonisation, then it is nearly always best to survey the same set of sample units over all seasons (Bailey et al. 2007). This is just one example of the kind of design question that can be addressed using the GENPRES software.

6.3 Occupancy and Distribution Mapping

Exercises directed at mapping the spatial distribution of elephants have often ignored the issue of detection probabilities and assume that animals are detected with probability 1 (i.e., it is assumed that if elephants are present in an area their signs are always detected). Using occupancy survey methods [MacKenzie et al. 2006; Conroy and Carroll 2009], it is possible to estimate the number of occupied cells in which the species is 'missed' during sampling, but it is not possible to specify the exact location of missed-detection cells. In these instances, the reasonable approach to mapping, in our opinion, is to develop the map using shading that corresponds to probabilities of cells being occupied (e.g., the darker the cell the higher the probability that it is occupied by the species). Examples of this approach, including a map of the distribution of the Asian Elephant in India, are provided by Karanth et al. (2009).

An important design consideration for mapping the entire distribution of elephants in an area of interest is that of what type of spatial sampling to employ. We offer here a few suggestions and discussion points about survey designs for the purpose of mapping overall elephant distribution. If there is no strong clustering of elephants, then a systematic or uniform sampling pattern may be best. Our rationale is that systematic uniform sampling can minimise the average distance between un-surveyed cells and surveyed cells. In general, the farther an un-surveyed cell lies from surveyed cells, the more difficult it is to make inference about its possible occupancy status. If strong clustering does exist, then adaptive cluster sampling [Thompson 1992; Thompson and Seber 1996] may be worthy of consideration. In both cases, the ability to identify and measure cell-specific covariates that are determinants of occupancy will facilitate inferences about un-surveyed cells. For spatial variation in occupancy that is not accounted for by covariates, methods based on the spatial correlation structure of cell occupancy may be useful ('Kriging' is a method based on this idea, Thompson 1992).

The practical issues relating to mapping the distribution of elephants are covered more thoroughly in Chapter 11. We believe that such mapping exercises can be most useful when they are conducted in conjunction with an appropriate spatial sampling design. Under this approach, all locations need not necessarily be sampled, but the critical point is that all locations have known *a priori* probabilities of being included in the sample. This condition permits reasonable inference when only a subset of potential locations is surveyed. As noted above, we also emphasise the need to adequately characterise detection uncertainty, for example, using shadings associated with probabilities of occupancy, in the preparation of distribution maps.

Finally, as Martin et al. (2010) note, ideally one should account for detection probabilities when estimating transition probabilities among occupancy or abundance states [Yoccoz et al. 2001; Williams et al. 2002; Chapter 1]. However, the basic design of many historical large-scale elephant monitoring programs does not allow for the estimation of detectability. The types of models presented by Martin et al. (2010) can be used to reduce errors associated with detectability by at least assigning individuals to broad abundance classes rather than modelling uncorrected count data directly. By assigning observations to broad categories in this manner it is possible to reduce the impact of misclassification errors, which result from imperfect detection (e.g., false absences). Chapter 13 provides further discussion about the use of modern methods to revisit old problematic data sets. Nevertheless, we again strongly encourage biologists and wildlife managers to design monitoring programs that will explicitly consider both detection and sampling variation to avoid errors associated with these two sources of variations, which can result in unreliable inference [Yoccoz et al. 2001; Williams et al. 2002].

While there are as yet no published applications of occupancy methods to the study of detailed elephant–habitat relationships or management questions in Asia, the large-scale study of Karanth et al. (2009) based on historical records and expert opinion surveys of current distribution concluded that the elephant had a relatively restricted range, occupying about 25% of the Indian subcontinent. While in Africa the studies of Buij et al. (2007) and Martin et al. (2010) have used occupancy modelling approaches to address several management-related questions relevant to elephant distributions.

6.4 Linking Abundance to Occupancy

Royle and Nichols (2003) presented a model that links occupancy to abundance when variations in detection probability exist as a result of variations in location-specific animal abundance. The key requirement being that variation in abundance is the primary driver of the variation in detection probabilities among sites. In such a case, variation in site specific abundance can be modelled explicitly in terms of its effects on heterogeneity in detection probability. Exploitation of this linkage allows the model to estimate the underlying distribution of site specific animal abundances. Thus, the Royle and Nichols (2003) model allows estimation of abundance from repeated observations of detection or non-detection (i.e., 'occupancy surveys'). Applications of this model must assume some underlying distribution of animal abundance across space, typically a Poisson distribution, although other distributions used include the negative-binomial, which allows for over-dispersion in the data, and the zero-inflated Poisson, which allows for additional empty sites. However, the model requires a number of assumptions that are unlikely to be met in elephant surveys: individuals are distributed in space according to a known spatial process (e.g., Poisson process), all individuals are approximately equally detectable across the sample unit as above and detection of individuals is independent. Royle (2004) has extended the model further and these methods have shown promise in avian surveys; see, for example, the comparison of abundance estimates (and worked example) obtained for Blue Grosbeaks using the Royle–Nichols (2003) and Royle (2004) models and those obtained using distance sampling methods (Conroy and Carroll 2009: 90–91). Clearly, further work is needed and is indeed under way, driven largely by researchers working on Tiger populations [Karanth et al. unpublished data].

In addition to the methods discussed above, Conroy et al. (2008) proposes a two-phase sampling scheme and model in a Bayesian framework to estimate abundance for patchily distributed populations. Adopting the Conroy et al. (2008) approach to estimate the abundance of elephants across a large landscape would involve, in a first phase, estimating occupancy from surveys to detect elephant sign (typically dung piles) in all selected sites in the landscape, where selection would be typically of all sites available (if possible), or a random sample of sites (if not). In a second phase, for sites in which an adequate detection threshold had been achieved in the first phase, capture–recapture sampling would be conducted to estimate site-specific elephant abundance. Capture–recapture modelling would then be used to estimate abundance to inform both the spatial distribution of abundance and the abundance–detection relationship. This information will be used to predict abundance at the remaining sites, where only site occupancy data were collected. These abundance models of occupancy and their extensions offer a promising approach to estimating elephant abundances across large landscapes from sign surveys of habitat occupancy.

6.5 Analysis of Elephant Occupancy Data

Going beyond standard GIS software such as ArcGIS and other landscape-level analytical tools, occupancy modelling uses specialised software to process the data in a 'capture–recapture' framework. PRESENCE [Hines 2008], as well as several options available in MARK [White and Burnham 1999], can perform most of these analyses [Appendix 4]. However, for grasping the basic concepts of occupancy estimation methods, we believe that the simple formulae proposed by Nichols and Karanth (2002) are useful. An example occupancy analysis is also provided in Chapter 11. Finally, we note that Conroy and Carroll (2009) provide some useful worked examples.

6.6 References

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CHAPTER 7

Estimating Density and Abundance of Elephants from Sightings along Line Transects: Field Methods

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7.1 Introduction

Line transect sampling based on visual detections [Chapter 3] is a powerful approach to the estimation of densities of elephants, but only in areas with relatively open vegetation types so that the elephants are easily visible [Karanth 1988; Dawson and Dekker 1992; Karanth and Sunquist 1992; Hedges 1993; Varman and Sukumar 1995; Kumar 2000; Goswami et al. 2007; Wegge and Storaas 2009]. For African examples see, e.g., Jachmann (2002); also see Msoffe et al. (2010). Line transect surveys are typically conducted by investigators on foot, seated on domesticated elephants or from vehicles; the last of these can only be conducted in areas where cross-country driving along true straight lines is possible [Wilson et al. 1996]. As with any method, it is important to ensure that the key assumptions [Chapter 3] are adequately met for this method to be useful. Biological and logistical considerations often impose severe constraints on being able to execute theoretically justifiable line transect surveys in the field. In this chapter, we describe the best practices necessary to attain this objective and discuss some practical problems encountered in the field by investigators.

As in all other sample surveys, 'sample size' achieved plays a critical role in generating reliable estimates of the elephant densities. In the context of line transect surveys of elephants, where densities of clusters (groups of elephants) are estimated and thereafter animal densities derived by multiplying these estimates by cluster size estimates, the 'sample size' consideration will involve both number of detections of clusters (for the estimating detectability and cluster size) as well as number of the spatial replicates used (for estimating spatial variability in encounter rates).

Where densities of elephants are low, investment of even a large sampling effort (long distances walked) may yield a small sample size of visual detections, leading to a relatively poor density estimate. In such circumstances, we recommend use of fecal DNA based (or possibly camera/video trap based) capture–recapture methods [Chapters 5 and 10] or, if these capture–recapture methods are not applicable for some reason, dung counts along line transects [Chapters 4 and 9]. With regard to the last-mentioned of these methods, it is interesting and relevant to note here that White and Edwards (2000) estimate that one can expect to see in the order of 150 elephant dung piles for every elephant sighted in the rainforests of Africa. In this chapter, we focus only on visual detection based surveys, where elephant densities, habitat conditions and logistical considerations permit the application of this specific method.

7.2 MEETING LINE TRANSECT ASSUMPTIONS IN FIELD SURVEYS

Reliable density estimates from sighting-based line transect sampling depend on satisfying five key assumptions [Chapter 3]:

1. An adequate number of spatial replicates (transect lines) are located randomly with respect to the underlying distribution of the elephant clusters.
2. Elephant clusters (groups) on or close to the transect line are detected with certainty.
3. Elephant clusters are detected at their initial location and these detection events are independent between clusters.
4. Measurements of distances, angles and cluster size are made accurately.
5. Elephant cluster sizes are recorded accurately.

We emphasise here that the 'cluster' results from geometric configuration of elephants visible from the transect line and, thus, may or may not include the entire social group of elephants present. We recommend that a pilot survey be carried out to check for potential field problems in meeting these critical assumptions.

7.2.1 An adequate number of line transects are located randomly with respect to the distribution of the elephants

As noted in Chapter 3, placement of line transects using a well-defined survey design alleviates the need to assume that elephants in the population being sampled are randomly distributed in the study area (an assumption that is unlikely to be true). Random placement (or systematic placement with a random start point) of an adequate number of transect lines (spatial replicates) helps reliable inference. The automated survey design feature implemented in the program DISTANCE [Strindberg et al. 2004; Thomas et al. 2010] offers several useful options that meet various logistical constraints [see Chapter 3]. Strindberg et al. (2004) recommend at least 20–40 replicate lines.

7.2.2 Elephants on the transect line are detected with certainty

Violation of this key assumption underestimates the proportion of clusters counted and negatively bias estimates of elephant density [Chapter 3]. Steady uniform pace of movement along the transect line, and balanced search effort over the area faced by surveyors is essential. Well-trained and alert observers ensure elephants in the sampled strip are not missed needlessly. We recommend a team of two observers, with the first person concentrating on the search by scanning an arc of 30° straddling the transect line. The second observer should concentrate on the remaining area, but quick communication between the two is essential. The observers should wear cryptic clothing (we prefer camouflage) and footwear that permits silent passage (Figure 7.1).

7.2.3 Elephants are detected at their initial location

Ideally, observers should obtain the distance, angle measurements and an estimate of the elephant cluster size before they are noticed by the elephants in order to ensure there is no responsive movement away or towards the observers. Responsive movement by elephants can be reduced significantly if observers move as silently as possible (transect lines could be maintained periodically by trimming the vegetation where essential, to enable the observers to move quietly in a single file), listen intently to breaking of branches, rumbles, snorts and other cues typically associated with elephants, detect fresh elephant signs such as dung, feeding and rub marks, and do not smoke or use perfumes. The direction of travel along transects should take wind direction into account as far as possible as moving into the wind will help reduce the chance of elephants moving away before they are seen. Such superior field-craft will help reduce the chance of elephants becoming alarmed and disturbing other elephants as they run away.

Active detections (which are preferred) occur when observers detect elephants before the animals become aware of them. Passive detections occur if the elephants move in response to the observers. Recording sighting distances (and sighting angles), as recommended in this chapter, also helps in identifying the elephants' locations before they move.

More than two observers will increase the amount of disturbance without adding significantly to the chances of detecting elephants.

7.2.4 Measurements made are accurate

In elephant surveys, the sighting distances and sighting angles (from the observer to the assumed centre of the cluster) should be recorded accurately and precisely. As a prerequisite, the transect line should be clearly marked so that observers know its exact alignment on the ground at all times. It is acceptable to temporarily move off the transect line by a few metres to get around obstructions (e.g., dense thorny vegetation, fallen trees and deep gullies). However, all measurements should be taken from the actual transect line.

Appropriate equipment such as binoculars, optical or laser range-finders, and compasses are critical to improve the accuracy and precision of observations and measurements. We do not recommend use of GPS compasses to measure sighting angles as these are not sufficiently accurate. Before each survey, range-finders should be calibrated against known distances (e.g., a marked staff at a known distance) and then recalibrated periodically throughout the overall survey period. When recording sighting distances to elephants, it is often helpful to take the measurement to a stationary feature in the environment at or close to the point of interest, which can greatly improve measurements (although sometimes several measurements must be made and aggregated if vegetation or other obstacles hinder a direct measurement or the distance is greater than the capability of the measuring instrument being used). While we generally recommend against 'guessing' distances we recognise that in some situations (e.g., intervening vegetation or the distance being measured being too large or too small for the range finder), this might be required. If such estimates are made on many occasions, observers should practise estimating these distances collectively before the survey so that their estimates match and are accurate. Untrained observers tend to be poor at estimating distances by eye (Alldredge et al. 2007). The observers should continue to check the accuracy of their estimates (against objects at known distances) throughout the survey period. Particular care should be taken to avoid to rounding of distances, say to the nearest five or 10 metres, which is a well-recognised problem (Buckland et al. 2001) that causes difficulties while modelling the data.

Sometimes, detections may occur after observers have passed the elephants. Such 'back sightings' should also be recorded. After such a detection, when one observer is measuring and recording data, the second observer should continue to scan ahead in order not to miss any other elephants.

The field protocol should clearly define what is meant by an elephant cluster. As a rule of thumb, we suggest treating all elephants within a 30-metre radius as practical way to define a cluster. Sighting distances from the observer are then recorded to the visually assessed geometric centre of the cluster. Thus observers need to be trained in the field to visualise the shape of clusters of elephants and to identify their geometric centres.

The sighting distance, r, should be recorded to the centre of the cluster. Two compass bearings should be measured: the first to the centre of the cluster, θ_1; and the second along the transect line, θ_2 (see Figure 3.2). The perpendicular distance, x, can then be calculated, during the data entry stage, as $x = r \sin(\theta_1 - \theta_2)$. Compass bearings must be measured with a sighting compass (Figure 7.2). Laser binoculars can take both distance and angular measurements accurately, although they are too expensive for most surveys. To avoid heaping and rounding errors, the distance and angle measurements should be accurately read and exactly recorded. It is particularly important to avoid recording any sighting as being exactly on the line (a common problem) unless the centre of the cluster of elephants really is exactly on the line (extremely rare). Sometimes a cluster of elephants straddles the line—even in this case, the geometric centre of the cluster is rarely exactly on the line.

7.2.5 Elephant cluster sizes are recorded accurately

Sighting-based line transect surveys provide an estimate of the density of elephant clusters, which is then multiplied by the mean cluster size to derive elephant population density. In some situations, size bias can occur (with cluster sizes closer to the line being estimated more accurately that those farther away [see Buckland et al. 2001]). As explained in Chapter 3, there are several analytical approaches to deal with this problem, although only accurate counting of cluster size is relevant to field practice. Observers must practice counting the individuals in elephant clusters rapidly and accurately before conducting the actual survey. If it becomes apparent that the cluster is larger than that initially counted, after the observers have moved on for example, it is advisable to record the extra individuals detected and if possible recalculate the geometric centre of the cluster and re-measure the sighting distance and angle (making a note that this redefined cluster is not a new cluster but the one recorded earlier).

7.3 COLLECTION OF ADDITIONAL AND ANCILLARY DATA DURING SURVEYS

Observers should record the age and sex classes of all elephants seen if possible. Other information that is commonly recorded is the observers' location along the line (say to the nearest 50 or 100 metres—this is facilitated by placing distance markers along the line or through the use of a topofil such as a HipChain) and, if required by your monitoring/study objectives, simple habitat type information. Changes in habitat type should be recorded as they occur or every *x* metres depending on your objectives.

We note that ecologists and managers are often interested in the spatial variation of elephant density within surveyed area, in addition to getting one overall estimate of abundance and density [Royle and Dorazio 2008]. They may also want to extrapolate or predict elephant densities at un-surveyed locations, using density data from sampled locations. Classical distance sampling approaches such as those described here, which are based primarily on design-based inference, cannot achieve these objectives. However, recent advances in inference related to distance sampling [Hedley and Buckland 2004; Royle et al. 2004; Royle and Dorazio 2008; N. S. Kumar, unpublished data] do address this issue effectively using analytical approaches that are outside the scope of this manual. However, if such are the survey goals, additional data on the spatial location of detections and habitat related covariates, should be gathered along with the standard line transect data described here. If such cases, it is essential not to weaken or dilute the quality of the basic line transect survey data. Therefore, if such risk of data dilution exists, it may be necessary to gather the additional covariate data needed from either separate field surveys or from GIS / remotely sensed data sources.

Similarly, the pursuit of other natural history or personal goals like measuring human impacts, making vegetation measurements, observing details of animal behaviour, bird watching or recording bird song while conducting line transect surveys seriously undermines data quality and should be strictly avoided.

7.4 LINE TRANSECT SURVEY DESIGN

When planning a line transect survey in a new area, always do a pilot survey first—otherwise your first real survey will effectively become a pilot survey! The pilot survey will allow you to estimate the total line length required given the elephant cluster encounter rate found during the pilot survey and the desired level of precision [see Chapter 3]. Indeed, the pilot survey may show you that it is not possible to achieve the desired level of precision given constraints on resources and time and so you may need to adopt another method or redefine your monitoring objectives. Finally, a pilot survey will help you assess the logistic and other challenges presented by the survey area and may help you identify strata, depending on the extent of your pilot study.

Once you have completed the pilot study, you should also manually plot a histogram of the recorded distances (or better still, use program DISTANCE to do this) and check for problems such as a spike at zero distance, rounding of angles or true distances, or evidence of evasive movement of elephants prior to detection (Figure 3.4).

Ideally, the more spatially replicated lines in the survey design there are, the more likely it is that you will obtain a representative sample and a reliable estimate of variance. However, elephant habitats can often be difficult to access, and so there are often high costs and logistical problems associated with having large numbers of spatial replicates. The number of spatially replicated transects one can establish will depend, therefore, on the terrain, resources (manpower, money, etc.) and time available to the survey team. The logistical problems associated with difficult to access areas can potentially be reduced by stratifying the study area by ease of access (or cost) as explained in Section 3.4. Stratification can also improve the precision of your estimates and provide information about elephant abundance or density in different management units, habitat types, etc. [Chapter 3]. However, a word of caution is appropriate here: stratification will only give modest increases in precision unless the differences in density among strata are large and there is also the danger that too little effort will be allocated to, or too few elephants seen in, a low density stratum, so that reliable estimation of abundance in the stratum cannot be achieved [Buckland et al. 2001].

Ideally, for each density estimate we generate, there should be at least 15–20 but preferably 25 spatially replicated transect lines [Buckland et al. 2001; Thomas et al. 2010]. If your goal is to obtain one density estimate for the entire area, then these replicate lines should sample the entire area. On the other hand, if you need separate density estimates for subareas such as management units, habitat types or other strata, then you must have at least 15–20 but preferably 25 or more replicates in each stratum. Smaller numbers of spatial replications may require estimating the variance of encounter rates theoretically rather than empirically (see Chapter 3 for a fuller discussion). This is generally considered a less desirable option than generating variance estimates empirically from your data [Buckland et al. 2001].

Walking at a speed of about 1.5 kilometres per hour usually enables efficient detection of elephants. Assuming that an observer can employ about 2–3 hours of concentrated sampling effort in a field session then transect lines of 3–4 kilometre length can be covered during each session. If there are to be 25 transect lines (spatial replicates) then about 75–100 kilometres of transect line may have to be established (also see Section 3.4.5 on estimating the total line length necessary for a given encounter rate—ideally derived from a pilot study—and target level of precision).

In conclusion, we strongly recommend that investigators rely on sampling and survey designs considerations [Chapter 3], avoid over-stratification, establish true transect lines that meet modelling requirements and follow the practical considerations explained in the next section.

7.5 Establishment of Transect Lines: Practical Considerations

Ideally, observers should be able to walk along a predetermined line transect route without disturbing any vegetation. However, in reality, because of vegetation density and other obstructions, this sometimes cannot be done. Crashing noisily through the bush will make elephants move from their initial location and thus violate the assumption of no undetected, responsive movement [Sections 3.3.3 and 7.2.3]. Failure of this key assumption can seriously bias the survey results. Therefore, transect lines may have to be physically marked and maintained minimally to permit quiet passage of survey personnel. The transect line width should just be enough to allow passage of survey personnel, but not wide enough to attract or repel elephants or artificially inflate the detection probability along the line itself [see Chapter 3]. The position of the line should be marked clearly at regular intervals using paint, brightly-coloured flagging tape or metal tags (Figure 7.3). In open woodlands and grasslands, it may be possible to use only paint marks without cutting any vegetation.

All too often, biologists use roads, stream banks, animal trails, firebreaks and other convenient linear formations as 'transect lines'. In almost all cases, these features represent specific microhabitats that elephants are attracted to or repelled from. Furthermore, such formations almost always will be non-random with respect to vegetation types, topography, and other factors that may be influencing elephant spatial distribution. Use of such non-random 'psuedo-transect lines' leads to violation of a major assumption [see Section 7.2.1] and severely biases density estimates (Varman and Sukumar 1995).

Permanently marked transects make it easier to conduct monitoring to detect changes in elephant abundance over time. Furthermore, variation in density estimates contributed by spatial location is eliminated from successive estimates giving investigators greater statistical power for detecting density variations [Buckland et al. 2001, 2004]. If the same transect lines are used repeatedly, however, elephants may either start using the lines as their paths or, in heavily hunted areas, avoid these deliberately. Such behaviours will violate the assumption of transect lines being placed randomly with respect to elephant distribution. If elephants are attracted to the transect lines then density will be overestimated; if they are repelled, it will be underestimated [Buckland et al. 2001]. Furthermore, cleared permanent lines in some cases may allow access to hunters and other illegal intruders [White and Edwards 2000; Buckland et al. 2001]. If there is a suspicion that elephants are responding behaviourally to the transect lines, then it is worth doing a separate study involving strip transects of elephant dung [see Chapters 4 and 9] at fixed perpendicular distances from the affected lines to determine the extent of the problem. Finally, it is desirable to wait for the effects of the disturbance created by cutting of transect to dissipate before conducting the actual survey.

7.6 Season and Timing of Surveys

If some elephants are migratory in the population of interest, the season in which surveys are conducted is determined by whether the investigator wants density estimates when most of the elephants are in the area or outside of it. The ideal season, from a practical point of view, is the one when footfall noise is lowest and visual detectability is highest. These are somewhat contradictory demands where marked wet and dry seasons exist and leaf fall is linked to dry seasons.

Greater visibility and lower footfall noise results in larger number of detections for a given survey effort: more clusters of elephants are encountered because a wider strip is sampled and evasive movement is reduced. As a result, sample sizes are larger and counts more accurate. Overall, conducting line transect surveys in the dry season, but after a few early showers, is often the best practical approach.

Typically, elephants are more mobile around dusk and dawn, and tend to rest during the hotter parts of the day. Therefore, they are more detectable (a wider strip can be sampled) around dusk and dawn. In other words, the shape of the detection function [see Chapter 3] will change, although the underlying densities do not change. We recommend surveys during early mornings and late afternoons to take advantage of elephant activity patterns.

Sometimes, a survey may have to be abandoned due to the onset of darkness, bad weather or elephants being aggressive. In all such cases, the location where the survey was abandoned and the actual distance covered should be recorded, so that sampling effort is computed accurately and, if required, a subsequent survey along the same transect can be resumed from the same point.

7.7 OTHER PRACTICAL CONSIDERATIONS

Before starting a line transect survey, the survey leader should record the following data: date, time, weather conditions and the surveyors' names. A specimen field data form is shown in Appendix 1. Such forms can be printed in bulk preferably on waterproof paper for use by multiple survey teams.

A survey may have to be abandoned due to the onset of darkness, bad weather—and in the case of potentially dangerous species like elephants—due to the animals literally blocking your path or behaving aggressively. We recommend that if rain persists for more than 15 minutes, the survey be abandoned because the rain may affect elephant behaviour and/or observer efficiency, thus affecting the probability of detecting elephants. In all such cases, the location where the survey was abandoned and the actual distance covered should be recorded, so that the sampling effort is recorded accurately on the data form. This is also useful for resuming the survey at a favourable time later on, if necessary.

Immediately after the survey, data forms should also be checked for possible data entry errors. The memory of a particular detection fades rapidly ('was that elephant cluster I saw at 7.20 am at 30 metres or 300 metres?'). All data should be carefully and flawlessly entered into a computer. Backup hard copies of datasheets and all electronic data files should be made regularly and carefully, and these should be stored in separate locations to the original datasets [see Appendix 3]. The data are now ready for analysis using methods described earlier [Chapter 3] and in Buckland et al. (2001).

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CHAPTER 8

Aerial Survey Methods

Simon Hedges and Timothy O'Brien

8.1 Introduction

Standardised aerial survey methods have long been established and are widely used for (African) elephant surveys, primarily in southern and eastern Africa; for a list of aerial surveys classified according to data quality, see Blanc et al. (2007). Accordingly, there are a number of standard texts covering the design and analysis of aerial surveys both for elephants and for other species [Norton-Griffiths 1978; Douglas-Hamilton 1996; Mbugua 1996; Jachmann 2001]; in addition, the CITES MIKE program has published a set of *Aerial Survey Standards* [Craig 2004]. Aerial surveys permit rapid counts of elephants and are particularly well-suited to surveys of large areas (of non-concealing elephant habitat types). For obvious reasons, aerial surveys are not appropriate for elephant populations living in tropical forests hence have been little used in Asia [but see Olivier and Woodford 1994].

Typically, aerial surveys are conducted as either total counts or sample counts [Douglas-Hamilton 1996; Mbugua 1996]. In most situations, including very large landscapes, aerial sample counts are usually the only viable aerial survey method. Aerial sample surveys can be conducted using line transect, strip transect, block or quadrat techniques. For aerial surveys of terrestrial mammals, including elephants, strip sampling has typically been the preferred option because navigation is easier and there is no need to search for block or quadrat boundaries [Wilson et al. 1996]. As we have already indicated, there are several standard texts on the conduct and analysis of aerial surveys including strip sampling methods. To avoid 'reinventing the wheel', we refer the reader to these texts and note that they are freely available online [Section 8.6]. We primarily restrict ourselves, therefore, to a short discussion of aerial line transect surveys, which we believe warrant further testing for elephant surveys, and a short review of the potential 'opening-up' of forested areas to aerial surveys as a result of developments in infrared scanner technology.

8.2 Limitations of Conventional Aerial Survey Methods

Aerial surveys tend to suffer from low precision because they are based on the instantaneous distribution of animals, and so the variation between transects is usually very high [Barnes 2001]. This low precision makes trend detection difficult when surveys are repeated infrequently [Ferreira and van Aarde 2009]. In addition, and more seriously, aerial surveys tend to produce underestimates of animal abundance. For example, only 29% of a known-size black rhinoceros population was counted from the air [Goddard 1967]; an aerial count of eight African large herbivore species returned only 23% of known numbers [Spinage et al. 1972]; and only 56% of known numbers of Indian rhinoceros were detected in an aerial survey [Caughley 1969]. These underestimates generally result from the several sources of bias that afflict aerial surveys. Some of these sources of bias can be avoided with a proper survey design, but others are less tractable.

Typically, detectability problems represent the most important source of bias in aerial techniques [Norton-Griffiths 1978; Jachmann 2002]. Detectability is influenced by the density of the vegetation, by the size and colour of the animals, by group size, by the animals' reaction to the aircraft, by light conditions (including weather conditions) and by operational factors, such as the aircraft's height above the ground during the survey and by search rate. As Jachmann (2002) notes, several sources of bias can be avoided with a proper design, i.e., (i) insufficient coverage of the survey area, when parallel flight lines are set too far apart (a problem for total count methods); (ii) attempting visual estimation of large herds, when photography should be used, and; (iii) double-counting of animals as a result of poor navigation. Other sources of bias are more difficult to avoid and include: (i) observer-related biases, which relate to the quality of the observers in terms of eyesight, experience and ability to concentrate for the duration of the flight; (ii) sighting probability bias, which relates to the lower probability of sighting single animals and small groups of animals, which can be minimised by keeping operational factors, such as search rate and height above ground level within reasonable limits; (iii) visibility bias, which is related to animals not 'available' to the observers because they are concealed by other animals, are cryptically coloured, or are obstructed by rock, trees and the like. For most animal species, the combined observer, sighting probability and visibility biases lead to undercounts [Jachmann 2002]. In the past, several methods have been proposed to eliminate bias from aerial counts [Caughley and Goddard 1972; Caughley et al. 1976; Cook and Jacobson 1979; Grier et al. 1981; Caughley and Grice 1982]. Unfortunately, most of the proposed methods are impractical and/or expensive [Barnes et al. 1986]. However, one notable exception is the double-count technique, which is both feasible and theoretically sound (it is based on capture–recapture statistics; see Chapter 5). The double-count technique involves two observers making simultaneous counts of the target species, independently and without collusion [Caughley 1974]. From the numbers of animals seen by the front observer only ('marked animals'), those seen by the rear observer only ('captured animals'), and those seen by both observers (marked animals that are 'recaptured'), a correction factor can be derived [Magnusson et al. 1977; Cook and Jacobson 1979; Grier et al. 1981; Caughley and Grice 1982; Graham and Bell 1989; Caughley and Sinclair 1994; Jachmann 2001]. This double-count technique probably corrects most observer biases and possibly a small proportion of sighting probability bias, but not visibility bias; furthermore, correction factors apply to a single animal species, for a particular count only [Jachmann 2002]. Nevertheless, these correction factors are routinely used in aerial counts throughout Africa and the accuracy of the resulting estimates is rarely questioned. This is quite a serious problem because the inaccuracies may be large and thus Jachmann argues that unless the problem of variable undercounting bias is solved, 'aerial techniques are of limited use for most animal species'. Not surprisingly, a number of researchers and surveyors have attempted to address the problem, primarily by applying line transect methods to aerial surveys [e.g., Burm and Griffin 2000].

8.3 AERIAL LINE TRANSECT SURVEYS

The major advantage of using distance sampling is that the variable strip width [see Chapter 3] can potentially provide estimates free of some of the biases to which conventional aerial sample counts are prone. Typically, aerial line transects are conducted by means of attaching 4–5 marker poles ('wands') to the wing struts of a fixed-wing aircraft, thus creating 5–6 distance intervals, with the last interval having an infinite distance

[Jachmann 2002]. An alternative is to use a clinometer to obtain the angle of declination ϕ to the centre of the elephant group as it passes abeam (where 0° is at the horizon and 90° is directly below the aircraft) and the altitude of the airplane h and by applying the formula $x = h/\tan\phi$ [see Chapter 3 and Fig. 3.2]. However, these approaches risk violation of an assumption critical to obtaining reliable estimates, that is, the probability of detection on and near the transect line should be one [Chapter 3]. The assumption is violated because observers may not be able to see the line, many animals move away in response to an overhead aircraft, and because the basic technique does not allow for correction of observer and sighting probability biases near the transect line: these violations result in a lower than expected number of animal detections on or close to the line, leading to undercounts [Jachmann 2002]. One 'high-tech' option, which we believe should be used more often, is to mount downward looking video cameras on the aircraft to allow for detections on or near the line to be noted after the survey flight.

Other problems include inaccurately assigning animals to a distance interval due to aircraft's rolling movements and the difficulty of determining when animals are perpendicular to the moving aircraft, which also results in errors when assigning groups of animals to different distance intervals. For these reasons, helicopters have typically been preferred for aerial line transect surveys. Unfortunately, for a given level of effort, a helicopter survey is roughly 5–6 times more expensive than a survey with a small fixed-wing aircraft because it is slower and it costs more per hour [Jachmann 2001, 2002]. In addition, as Jachmann also notes, because terrestrial line transect surveys [Chapter 7] are cheaper than helicopter surveys, they are probably to be preferred over aerial surveys for small- to medium-sized study areas (< 5000 km^2); also see Msoffe et al. (2010), who argue that terrestrial sighting-based line transect surveys are more accurate, more precise and less expensive than aerial surveys at least for smaller areas.

8.4 Infrared Aerial Surveys for Forested Areas*

Barnes (2001) noted that thermal (infrared) imaging had been tried successfully with deer in the UK [Gill et al. 1997] but he argued that it was unlikely to work in the dense foliage of tropical forests. Barnes added that 'experiments with thermal imaging from the air showed that it was impractical, a conclusion already predicted from theory [Prinzivalli 1992]'. However, there have been significant advances in infrared detector

* We are indebted to George Wittemyer for a review of infrared aerial survey methods that he prepared for the CITES MIKE program in late 2010, which greatly helped in the writing of this section.

technology over the last decade or so which arguably allow aerial surveys of areas previously precluded because of the nature of their plant cover [Roberts et al. 2006; Burn et al. 2009]. Small military-grade infrared sensors can apparently detect an elephant at 500 metres through partial canopy cover, but International Traffic in Arms Regulations (ITAR) restrictions apply to international movements of such technology. More seriously, given the nature of tropical forests in areas such as Central Africa, near closed canopy (canopy cover after 85%) causes significant signal disruption that would be likely to limit a sensor's ability to distinguish an elephant from another heat source. Further experiments and trials are still required, therefore, to assess the sensors' abilities to distinguish elephants from other heat sources. Another important limitation is that the swath width of an IR sensor is less than that used in typical direct sighting based aerial surveys (c. 10–30 metres at 90 metres height), therefore a greater number of infrared transect flights are required to cover a particular area than in conventional aerial surveys. Moreover, it seems likely that high daytime temperatures in Central Africa would preclude the use of infrared monitoring systems except at night. However, flying and landing at night is more hazardous for the pilots, especially in the forest belt of Central Africa. There is, therefore, growing interest in the use of Unmanned Aircraft Systems (UASs) or 'drones' [Jones et al. 2006; Watts et al. 2010]. Advances in UAS technology are occurring rapidly, in large part because of the interest of the military.

The advantages of using UAS for aerial surveys include the ability to conduct low speed and low altitude flights with minimal risk, lack of pilot and observer fatigue issues (see discussion of observer-related biases above), and reduced per flight costs compared to conventional aircraft (although initial investments are high if the UASs are purchased rather than hired). The cost of UAS systems, their carrying capacity and flight range depend on the size of the system. The smaller fixed-wing UASs can be hand-launched and can be landed using nets, parachutes and the like, as well as landing strips. Ranges of such UASs, which are powered by electric motors, are typically around 120–150 km, and currently cost $70,000–80,000. Helicopter UASs are of course also similarly versatile in their launch and landing requirements, but currently their range is only some 50–60 km. Medium sized petrol motor driven units have payloads of up to 23 kg, ranges in hundreds of kilometres, and cost between $250,000 and $2 million; they also require landing strips. Larger UASs have longer ranges (>1000 km) and are similar to manned aircraft in respect of take-off and landing requirements. Costs for such units start at about US$1.5 million. As such, flight-time of a large UAS would presumably have to be rented from an agency that owned UASs.

In addition to cost, other disadvantages, at present, include the difficulty of transporting the rather fragile UASs to remote areas (such as the forests of Central Africa or northern Burma). UASs also require highly trained individuals to operate them and, at least in the case of the smaller UASs, these operators have to be on site. The larger UASs can be controlled remotely using satellites, but at increased expense. Moreover, two further major constraints on the use of UASs for elephant surveys exist currently: the Federal Aviation Authority (FAA) in the US restrict the use of small UASs within US airspace to within sight of ground control stations in response to concerns regarding potential for mid-air collisions. Since the US is the major market for UASs, such regulation is limiting development of longer range small UASs. Other countries' equivalents of the FAA may also place similar limits on the use of UASs in elephant range States. On a more serious note, the sale and transport of UASs with advanced flight guidance systems (which are required to facilitate the precise flight paths needed by aerial survey methods) is governed by the ITAR since a UAS fitted with an explosive payload could become a ready-made cruise missile! Nevertheless, NOAA has addressed this constraint by asking the US military to transport the UASs across international boundaries for international surveys. Similar military arrangements would presumably be required to transport UASs used for elephant surveys, rather adding to the logistic and administrative burden on those planning the surveys.

In conclusion, while increasingly promising, these infrared aerial survey methods need to be further developed before they can be recommended for surveys of elephants in closed canopy forests. Most importantly, at present, infrared sensors will detect only elephants in areas of semi-open forest canopy.

8.5 Occupancy, Double Sampling and Two-Phase Adaptive Sampling

Because many aerial surveys in the open vegetation types of eastern and southern Africa have been carried out over very large landscape (> 5000 km^2) using the Systematic Reconnaissance Flights (SRF) methods of Norton-Griffiths (1978) and employing systematic sampling surveys, there are options to use the standard aerial survey sampling design to obtain unbiased estimates of occupancy [MacKenzie et al. 2006; Chapter 6] that may complement current population estimation methods. Recent advances in occupancy analysis permit incorporation of spatial sampling replicates that are not randomly selected where we may expect spatial autocorrelation or Markovian dependence [Hines et al. 2010; Chapter 6].

A standard aerial survey design using methods of Norton-Griffiths (1978) first divides the study area into sampling units that are treated as subunits along a transect. For example, an investigator may choose 100 subunits of 2x5 km^2 and fly 10 transects through each of 10 sets of subunits. The result is a set of 50 km transects sampling a middle strip of the subunits. The observers record elephants seen within the strip as well as covariate data. This data collection process lends itself to an occupancy analysis in which subunits are treated as spatial replicates. Subunits can be grouped into sampling units covering an area that approximates the home range size of an elephant or elephant group over a time period appropriate to the length of the survey (perhaps a month). Covariates of land use and vegetation type are typically collected during aerial surveys and may be used to improve estimates of occupancy. Vegetation type exerts a large influence on the ability to detect elephants and may be used as a covariate affecting detection probability. Models may be built with and without incorporating Markovian dependence to evaluate its importance to the detection process.

Because elephants are patchily distributed, some patches can have large numbers of elephants whereas others may have few or none. This spatial variation drives down the precision of population estimates based on standard analytical approaches [Jolly 1969a; Jolly 1969b; Norton-Griffiths 1978]. Two related, alternative population estimation strategies include double sampling [Williams et al. 2002] and two-phase adaptive sampling [Conroy et al. 2008]. Double sampling, as the name implies, involves collecting two samples, one sample being a subsample of the other. Aerial counts (*xi*) are conducted using standard methods, while more accurate counts or population estimates (*yi*) are made on a subset of the aerial samples. The relationship between *xi* and *yi* can be determined using ratio or regression methods and used to predict *yi* for the larger landscape. If the abundance estimates and counts are highly correlated, then the precision of population estimates based on the predicted values of abundance can be significantly improved.

Two-phase adaptive sampling is an extension of double sampling methods. In the first phase, an occupancy survey is conducted across all sites or across a randomised sample of sites. In the second phase, a subset of occupied sites is sampled for abundance using some population estimation procedures (capture–recapture methods or distance sampling). The occupancy phase allows estimates of the rate of occupancy and of the number of occupied sites. In the second phase, if a detection threshold is achieved, defined as a minimum number of detections in *z* replicated sampling units, then an abundance estimation procedure is conducted. The detection and abundance data are used to model the detection probabilities in occupancy using an abundance covariate model. The population estimation model is used to estimate abundance at the sites where abundance sampling is conducted and to estimate the relationship with detection probability associated with occupancy. Finally, an abundance–detection model is used to predict abundance at the remaining sites that were not sampled for abundance (also see Chapter 13). Conroy et al. (2008) believe their approach should lead to a cost-effective method of estimating abundance of patchily distributed species over large areas in a statistically rigorous fashion.

The methods discussed here have yet to be applied to aerial surveys of elephants although initial evaluations are currently underway.

8.6 References

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CHAPTER 9

Estimating Absolute Densities of Elephant Populations Using Dung Counts Along Line Transects: Field Methods

Simon Hedges, Fiona Maisels and Stephen Blake

9.1 Introduction

As we saw in Chapter 4, there are three essential components to a dung density based population survey: estimating dung pile density, estimating dung decay (dung disappearance) rates and estimating defecation (dung production) rates. The first two of these components requires dung piles to be classified into stages based on their state of decay, and so unambiguous dung classification systems are also of great importance. The method of estimating dung pile density recommended in this manual is the line transect [Buckland et al. 2001; Buckland et al. 2004; Chapters 3 and 4]. In recent years, a number of so-called 'recce' (reconnaissance) survey methods have been used in conjunction with line transects in an attempt to reduce the logistic challenges posed by line transect based methods and improve the precision of dung count surveys [Walsh and White 1999; Chapter 4]. The recce transect method, as described by Walsh and White, involves walking along a 'path of least resistance' through the forest and counting all dung piles found but not measuring perpendicular distances to these dung piles. The recce data set can then, in theory, be used to estimate dung pile density providing the functional relationship between encounter rates on recces and line transects was derived from a subset of recces matched with line transects. However, the use of these recce methods to estimate elephant dung pile density is considered problematic (Section 4.2.2) and is not therefore recommended in this manual.

9.2 Dung Pile Classification

9.2.1 Standard dung pile classification systems: the S system used by the CITES MIKE program and an experimental alternative

Whenever we aim to estimate elephant density from estimates of dung pile density it is necessary to calculate dung pile decay rates. This is done by monitoring the decay of freshly dropped dung piles until they disappear. Dung count based surveys rely on the field workers responsible for monitoring decay rates and those counting dung piles along transects being able to consistently classify dung piles into the appropriate classes. It is very important therefore that the dung pile classification system adopted is robust and simple to use. The S system, introduced in Chapter 4 and described in detail below (Table 9.1; Section 9.2.2), is a relatively simple system that appears—from extensive field trials in Asia—to be robust, and it is also the system that was adopted for the CITES MIKE program's *Dung Survey Standards* [Hedges and Lawson 2006]. For a discussion of the problems with previously-used systems see Section 4.3.

While the S system seems to be an improvement on earlier dung pile classification systems, getting field teams to apply it consistently is still a challenge and requires considerable investment of training time. Thus there is still a need for a simpler and unambiguous system of classifying dung piles. An attractive possibility for such a system is to use dung height [Walsh and White 2005; Kuehl et al. 2007; Sections 4.3.5 and 9.2.3].

9.2.2 Using the S system: practical guidance

Following Hedges and Lawson (2006), we recommend that the MIKE program's S system for classifying dung piles (Box 9.1; also see Section 4.3.4) is used for all dung count based survey work. However, we also recommend that the experimental height-based system introduced in Section 4.3.5 and described in Section 9.2.3 be used simultaneously in all dung surveys in order to better evaluate the two systems.

The following points should be kept in mind when using the S system:

- During field work, dung piles should be recorded as belonging to stage 'S1', 'S2', 'S3', 'S4' or 'S5' as appropriate; dung piles should not be recorded as, say, 'present' if in classes S1 through S3 and 'disappeared' if in classes S4 or S5. This classification into 'still present' and 'disappeared' is done during the analysis phase (see Sections 4.3.5 and, especially, Section 9.3.3 and Box 9.1).
- When testing to see whether a bolus is still coherent, gently touch and rock it to see whether the whole bolus moves as one (it is still 'coherent') or whether the bolus is in fact already split into more than one fragment (it is no longer 'coherent'). See Box 9.1 for the definition of 'coherent'.
- Look to see whether termites or ants have hollowed-out boli. Such hollow boli will crumble easily when examined but, when examining such boli to determine if they are coherent, it may be necessary to break them open to see if they held together by mud or fecal material (Plate 9.1).
- If the number of boli is unclear, note the range of possibilities. For example, write '5 to 7 boli' or 'at least 4 boli'.
- Often knowing the exact number of intact boli will not be important (see Box 9.1 for the definition of 'intact'). For example, if a dung pile has five boli, of which two are definitely intact and another may be intact, do not waste your time trying to decide whether the third bolus is intact because the dung pile will be in stage S2 regardless of whether two or three boli are intact.
- You must, however, take the time to examine the boli carefully if finding out how many intact boli that remain will determine which stage the dung pile is in. For example, if a dung pile has five boli, of which four are definitely not intact and the fifth may be intact, then deciding whether that fifth bolus is really intact or not will determine whether the dung pile is in stage S2 or S3. If it is impossible to be sure, give both possibilities and make a note of the reason(s) why you are unsure.
- You must also be careful when deciding how many dung piles the boli you find come from. Sometimes you will find two or more dung piles close together. In such cases you will need to look at the size and appearance (colour and shape) of the boli, as well as how degraded they are (e.g., whether they have fungus growing on them) and how far apart they are (Plate 9.3). These observations will help you decide which boli belong to which dung pile and how many dung piles are present. The number of boli should also be used as a guide: most elephant dung piles contain 3–8 boli, so if you find 15 boli it is very

likely that they belong to at least two dung piles. If it is impossible to be sure, give all possibilities and make a note of the reason(s) why you are unsure.

9.2.3 An experimental height-based dung pile classification system

As we have already discussed, there is a need for a simple-to-use unambiguous dung pile classification system and dung height has been suggested as a suitable criterion on which to base such a system [Walsh and White 2005; Kuehl et al. 2007]. We recommend therefore that all dung count based surveys collect dung height data during the dung decay monitoring and density estimation phases of work.

We suggest defining dung height as *the vertical distance between the ground on which a dung pile is sitting and the maximum diameter of the tallest dung bolus in the pile*. We suggest using calipers to measure dung height (diameter). We recommend using bolus diameter as a measure of height of elephant dung to avoid confusion between bolus height and bolus length; e.g,. if a bolus is standing on its end it may appear taller than it would if it were lying on its side. Confusing bolus height and length can result in the 'undecaying' problem mentioned in Section 4.3.3, and will require care in the field.

The decay threshold (i.e., the height below which a dung pile is considered to have decayed) is defined *post hoc* from the decay monitoring and transect data sets. Clearly, a large (tall) threshold height will increase the number of dung piles crossing the threshold (i.e., 'decaying'), which will decrease the length of the dung decay monitoring period required, including the pre-survey periods required by so-called retrospective methods (Sections 9.3.2; also, see Sections 4.4.1, 4.6.2.4 and 4.6.2.5). However, increasing the threshold height increases the number of dung piles which are excluded because their initial height is below the threshold height (e.g., those from infants and juveniles or the 'cow pat' type of dung pile that can result when elephants eat a lot of very moist fresh foliage and/or fruits and which is very common in some African forests). Increasing the threshold height also reduces the number of dung piles included in the transect data set (because a larger proportion will have 'decayed'). Hence, the essential problem is to optimise the efficiency gain accrued by reducing decay time with the precision loss resulting from smaller sample size [Kuehl et al. 2007].

Box 9.1 *The MIKE S system for dung pile classification [Hedges and Lawson 2006]*

Stage	Definition
S1	All boli are intact (see notes below).
S2	One or more boli (but not all) are intact.
S3	No boli are intact, but coherent fragments remain (fibres are held together by fecal material, see notes below)
S4	No boli are intact; only traces (e.g., plant fibres) remain; no coherent fragments are present (but fibres may be held together by mud, see notes below).
S5 (gone)	No fecal material (including plant fibres) is present.

Notes:

A bolus is 'intact' if: (1) its shape and volume is plausibly the original shape and volume and (2) it is coherent *and* can be handled without crumbling.

A 'coherent' fragment is defined as a fragment (consisting of plant fibres embedded in a matrix of other fecal material) that does not crumble/break-up when handled. Plant fibres held together by mud *do not* count as coherent fragments.

The fecal material must be handled before a dung pile is classified:

When examining boli to determine whether they are coherent, it may be necessary to break them open to see if they held together by mud or fecal material.

When examining fragments, they should be passed from one hand to the other and rubbed gently between the fingers to determine whether the fibres are truly coherent or whether they separate easily (Plate 9.2), *but no attempt to pull them apart or crush them should be made*.

It is important to remember that dung piles are not necessarily in stage S1 when freshly dropped by elephants, they can be in stage S2 (or even stage S3): this is very common in some African forests.

It is recommended that all survey leaders produce a sheet of annotated drawings and/or photographs to help teams correctly identify 'intact' boli.

9.3 Monitoring Dung Pile Decay Rates

9.3.1 Preamble

Please remember that throughout this manual, the term 'decay' not 'disappearance' is used to refer to the disappearance of dung piles irrespective of the means by which the process occurs. For example, dung piles that have been washed away by water, destroyed as a result of insect activity or trampling and scavenging by animals or broken-down as a result of bacterial processes, are all considered to have 'decayed'.

As discussed in Chapter 4, most dung density based surveys to date have used dung pile decay rates from other places and/or from other periods, which can introduce large biases and result in large over- or underestimates of elephant density. In addition, even when dung pile decay rates have been monitored specifically for the survey in question, in almost all cases, monitoring has been initiated *at the same time* as the dung surveys themselves: this approach, termed the 'prospective' approach by Laing et al. (2003), can lead to significant biases [Buckland et al. 2001; Marques et al. 2001; Laing et al. 2003]. Two alternative methods, called 'retrospective' approaches by Laing et al., were developed independently by Hiby and Lovell (1991) and by Marques et al. (2001), Buckland et al. (2001: 186–187) and Laing et al. (2003). Based on the discussion in Chapter 4 and following Buckland et al., Laing et al. and the MIKE program's *Dung Survey Standards* [Hedges and Lawson 2006], we recommend that all elephant dung surveys use a retrospective method of estimating dung pile decay rates. This means that it is essential to conduct decay experiments prior to every survey, at every survey site.

Since retrospective approaches aim to integrate spatial and temporal heterogeneity in dung pile decay rate over the period preceding the survey, implementing them in an unbiased manner requires that the monitored dung piles are distributed representatively across a study area, not simply around a convenient location such as a national park ranger station or research centre [Buckland et al. 2001: 186–187; Laing et al. 2003; Hedges and Lawson 2006; Kuehl et al. 2007; Chapter 4]. Recent elephant survey work that has used a retrospective approach (e.g., Hedges et al. 2007; Hedges et al. 2008; Hedges et al. in review), demonstrates their applicability but they do require a large investment of time and other resources and may not be practicable in large areas with poor access and/or in areas with very low elephant density (see Chapter 4). If it is simply not possible to apply a retrospective method in a spatially unbiased manner in your area, then do not use a dung density based methods and select one of the alternative survey methods discussed elsewhere in this manual (Chapter 2; also see Section 4.6.2.5).

9.3.2 Practical guidance

9.3.2.1 Design and timing of decay rate monitoring

- All dung surveys should use the retrospective method of Laing et al. (2003) for estimating dung decay rates. The prospective steady-state approaches to incorporating decay rates into elephant density estimation should *not* be used because they lead to biases of unknown magnitude and direction (Chapter 4).

- Use of the Laing et al. retrospective method means that dung decay monitoring work has to be conducted *prior* to every survey, at *every* survey site.
- If possible, you should use local knowledge or the results of previous studies at the site to estimate the time, *t*, it takes for elephant dung piles to completely disappear. Begin monitoring dung pile decay rates *t* months prior to the mid-point of the line transect survey, e.g., if a 3-month-long line transect survey were planned for June/July/August 2012 and *t* is 4 months, decay rate monitoring should be initiated in March 2012. If there are no data on dung pile decay rates for the site, start monitoring dung pile decay a minimum of 10 months prior to the mid-point of the line transect survey (Section 4.6.2.4).
- Plan for a minimum of six equally-spaced visits to the survey area, with the final visit timed to coincide with the midpoint of the line transect survey. For example, if *t* is 4 months and the line transect survey is planned for June/July/August 2012, the six equally-spaced visits will need to be made at 24-day intervals and will take place on the following dates: 17th March, 10th April, 4th May, 28th May, 21st June and 15th July. Obviously, in practice, it is unlikely that you will be able to find and mark all the necessary fresh dung piles in one day and so the visits should be timed so that they centre on the dates above.
- During each visit, locate and mark a minimum of 20 fresh dung piles (see Section 9.3.2.2 for guidance on how to identify fresh dung piles). There is no need to classify the dung piles—indeed you should not classify them using the S system at this point, although you should measure dung height if you are testing that system (Section 9.2.3). It should be noted that, at present, there is some uncertainty about the number of dung piles that need to be monitored: 20 dung piles per visit may be too many and 10 may be adequate (Chapter 4). However, given this uncertainty, the precautionary principle suggests aiming for a minimum of 20 per visit (or 120 for the whole study): survey work is expensive and so you do not want to discover that 60 is inadequate after the survey is completed!
- The procedures for searching for fresh dung piles should aim to ensure that representative samples of the survey site's major vegetation types, rainfall zones and topography (slope) zones are obtained. Ideally, this would involve conducting a designed survey, for example, one comprising several strip transects or dung search blocks, randomly or systematically placed within the study area, to ensure that landscape/

vegetation types (habitat types) are sampled in proportion to their occurrence [Buckland et al. 2001; Laing et al. 2003]. In practice, however, it may be necessary to search for fresh dung piles in three or four areas selected to encompass as many of the factors likely to affect dung decay rates as possible. Before adopting this latter approach, it is essential to consult with a statistician familiar with dung-based surveys or other appropriate expert.

- If the method for searching for fresh dung piles in representative areas given immediately above is simply not feasible, consideration should be given to methods that rely on finding a concentration of elephants (and therefore fresh dung piles) and establishing dung decay monitoring experiments by moving dung piles to representative areas. Unfortunately this ‘find and move’ approach is problematic. For example, if dung decay plots are established by moving freshly dropped dung piles to locations chosen to represent, say, different vegetation types, the decay rates in these plots may in fact not be truly representative of decay rates in those vegetation types (see Section 4.4.2.1). A further concern here is the density of dung piles in the plot: for example, artificially high densities of dung piles may attract unusual numbers of dung beetles and lead to higher than typical decay rates. Even if dung piles are moved to random locations those locations may not be representative of areas where elephants defecate. Another issue is the loss of dung piles to flooding: plots are unlikely to be established in areas close to rivers for example, but dung piles will be dropped in such areas. Thus, the use of plots or other ‘find and move’ approaches is likely to underestimate the decay rates of dung piles. This is why, for unbiased estimation, the ideal is to locate freshly dropped dung piles by systematic searches of landscape/vegetation types (habitat types) sampled in proportion to their occurrence, and then monitor the dung piles *in situ* (Sections 4.4 and 4.6.2.4). However, we recognise that the ideal may not be obtainable at all sites. If ‘find and move’ approaches have to be used, advice must be obtained from suitably experienced statisticians or other wildlife population monitoring experts.
- Great care should be taken to avoid over-representation of dung piles dropped at waterholes and saltlicks, or on logging roads or major elephant tracks. These dung piles are likely to decay at un-representatively slow rates because of their exposure to sunlight, which dries them out [White 1995], or at unrepresentatively high rates because of the effects of trampling by animals [Hedges and Meijaard in prep.]. The teams should therefore search off trails in the adjacent forest whenever they find evidence of fresh elephant presence.

9.3.2.2 Field methods for dung pile decay rate monitoring

- For a list of equipment needed for dung pile monitoring work, see Appendix 2.
- Search for and mark fresh dung piles only. Fresh is defined for the purposes of this manual as meaning dung piles dropped within the previous 48 hours (see also Chapter 10). It is important to remember that fresh dung piles may not be intact; they can be in stage S2 (or even stage S3) when found. Fresh dung piles are identified by their appearance. They will be moist throughout, making then dense (heavy). They will usually feel slimy to the touch and the outside of intact boli or pieces of boli are shiny and look wet. Flies will often be present and the dung pile should smell of elephant dung, not fungus or earth. Secondary evidence of fresh dung is also sometimes provided by the presence of obvious recent elephant footprints and possibly damage to vegetation (e.g., plants pushed-over or trampled/eaten).
- If time, human resources and dung pile abundance permit, the criteria for identifying 'fresh' dung piles should be studied; and if, for example, observations show that the criteria classify dung piles as 'fresh' if they are up to four days old, fresh sign should be considered to be 2 days old (the average age of signs identified as fresh) for the purposes of analysis, as recommended by Laing et al. (2003).
- Dung piles should not be fenced-off or otherwise protected from trampling, disturbance, etc. They should be left as they were found.
- We suggest that both the standard datasheets (Appendix 1) *and* field notebooks should be used to record decay rate data as described below (having a notebook in addition to a datasheet allows you to record additional information which will help you relocate the dung piles; see Box 9.2).
- The number of boli per marked dung pile should be recorded.
- Do not classify the fresh dung piles using the S system when they are first found; however, the vertical distance between the ground on which a dung pile is sitting and the maximum diameter of the tallest dung bolus in the pile should be recorded for all marked dung piles. Dung piles should <u>only</u> be classified using the S system during the final visit (to avoid excessive handling of dung piles, which may affect decay rates); the maximum diameter of the tallest dung bolus in the pile should also be recorded for all marked dung piles during the final visit.
- All fresh dung piles located and included in the monitoring program should be identified using a unique reference number, which is recorded on the appropriate datasheet, in the team's field notebook,

Box 9.2 *Data to record while monitoring dung pile decay rates*

(a) Essential location data for monitored dung piles

What to record	Where to record it
Reference number	On a nearby tree (using red or day-glo pink paint), on orange or day-glo pink flagging tape (using a permanent black marker pen) and on the datasheet (in black waterproof ink), as well as in the field notebook.
GPS location data (UTM data)	On the datasheet and in the field notebook.
Distance and compass bearing from tree with red or pink painted reference number	On the datasheet and in the field notebook.
Distance and compass bearing from the orange or day-glo pink flagging tape	On the datasheet and in the field notebook.
If necessary, a simple sketch map showing the location of all boli in a dung pile (e.g., if the dung pile is scattered over several metres)	In the field notebook.
A description of the location (to help you find the general area again)	In the field notebook.

(b) Other data to record for each dung pile

In addition to the reference number and location data discussed above, the following data should be collected for each dung pile:

- The date the dung pile was found
- The number of boli in the dung pile
- Whether the dung pile was found on a trail or not
- Whether the dung pile was moved (e.g., whether it was moved off a trail into the surrounding forest)
- The vegetation type including undergrowth type
- Slope (degrees)
- Altitude (metres above sea level)

See the example datasheet (Box 9.5).

and on a marker (a metal or plastic stake) that is pushed into the ground next to the dung pile (Plate 9.4). Indelible-ink pens or paint will be needed to write on these stakes. Bamboo or wooden stakes should be avoided if possible, as they tend to rot or be consumed by termites.

- The reference number should *also* be written in red paint or other bright colour paint on a nearby tree (Plate 9.5) and the number of paces (or measured distance using a metal tape) and compass bearing from the tree to the dung pile (Plate 9.6) recorded in the team's field notebook. Finally, the dung pile should *also* be marked by tying suitable coloured flagging tape (e.g., orange or day-glo pink) marked with the reference number to a suitable nearby branch and recording in the field notebook the number of paces (or measured distance using a metal tape) and compass bearing from the tape to the dung pile (Plate 9.7). This duplication of effort may seem like 'overkill' but, as experienced dung surveyors will know, metal or plastic stakes get kicked out by elephants or covered by dung piles, ants cut flagging tape from trees and red paint fades and can be hard to see in the gloom of the forest. Furthermore, because only a relatively small number of dung piles are monitored during these experiments a failure to relocate even a few dung piles can have serious implications for the quality and utility of your data.
- In addition to the location data described above, the GPS location of the dung pile *must* be recorded on the datasheet. The position should be taken when the GPS position has stabilised.
- A general description of the dung pile's location (e.g., 'approximately 2 km downstream of Ban Thalang, 200 m west along major elephant trail, on the river's north bank') should be recorded in the team's field notebook as this will speed a team's ability to relocate dung piles.
- Locations of all marked dung piles should be clearly recorded on datasheets so that the dung piles can be easily found again. This should be tested to see whether the directions recorded (bearing and number of paces from recognisable landmarks or GPS points) are adequate.
- Monitoring teams *must* make a duplicate set of datasheets as a backup. One copy should be left at the base camp. If possible, an electronic version should be made each night at base camp, and sent in an email to the survey- or site-coordinator, to ensure that each day's data is backed-up at a different location.

9.3.2.3 Classification of dung piles during decay rate monitoring

- We recommend that you use logistic regression methods to estimate the probability of dung pile decay as a function of time [Laing et al. 2003]. This method only requires a single follow-up visit, timed to coincide with the mid-point of the line transect survey period, to establish whether the dung piles are still present or have disappeared (Chapter 4).

- However, since it is vital that all marked dung piles can be re-located, if time permits dung piles that were marked in previous visits should be checked during all subsequent visits to (1) ensure that the teams can relocate the dung piles and (2) to check that the metal or plastic stakes, paint and flagging tape are still present (and to replace these if necessary).
- Nevertheless, it is more important for the teams to search for additional fresh dung piles during a site visit than it is for them to return to previously marked dung piles. Priority should therefore be given to searching for new dung piles.
- Dung piles should only be classified during the final visit (to avoid excessive handling of dung piles, which may affect decay rates). The final visit should be timed so that it falls in the middle of the line transect survey period that will be used to calculate dung pile density (Section 9.3.2.1).
- It is vitally important that the criteria for determining whether dung piles have decayed used during the decay monitoring experiments are the same as those used in the transect surveys to estimate dung pile density. In both cases, the S system described in Section 9.2.2 should be used as well as the experimental height-based system described in Section 9.2.3.
- If possible, the same people who will conduct the survey should be responsible for monitoring decay rates. This is to try and ensure consistency of classification between decay monitoring experiments and surveys. Where it is not possible to use the same people, regular checks of consistency between teams should be conducted.
- In any case, dung decay rate monitoring programs should be designed so that testing inter-observer consistency of dung pile classification is possible, and such testing should be conducted during each visit.

9.3.3 Estimation of dung pile decay rate

The basic method of estimation is described by Laing et al. (2003). However, considerable simplifications to their calculations are possible. In essence, the data required to estimate mean time to decay are, for each marked dung pile, its age on the date of the last visit and its status, which is coded as '1' if the dung pile has not yet decayed (i.e., it is deemed to be 'still present') and '0' if it has decayed. Assume for now that no other covariates have been recorded.

Logistic regression techniques can then be used to estimate probability of decay as a function of time, and possibly of other covariates, and the mean time to decay is estimated from this function together with its standard error (SE) and coefficient of variation (CV). Free software is available for making these calculations easily (see Appendix 4) and an example of data coding and entry is provided in Box 9.3.

9.4 DEFECATION RATES

9.4.1 Background and recommended approaches for incorporating defecation rates

As discussed in Chapter 4, defecation rate data for all elephant taxa are scarce and obtaining defecation rate data from wild elephants, particularly in forests, is difficult and potentially dangerous—although it can be done with the aid of experienced trackers. In theory, captive elephants can be used to produce the necessary data, but in practice, there are serious concerns about the influence of a typical captive elephant's diet on its defecation rate. In light of these problems, we described and provided detailed justification for three options in Chapter 4; to summarise:

- **For areas where defecation rate studies have been conducted** Use the defecation rate data given in Table 4.4.
- **For forests in weakly seasonal areas with no available defecation rate data** Assume that (a) defecation rates do not show significant seasonal variation in forests [Merz 1986; Tchamba 1992; Theuerkauf and Ellenberg 2000; Tyson et al. in review], and (b) that a rate of 18.07 defecations per 24 hours (with standard error of 0.068918) is appropriate for forest sites in weakly-seasonal areas of Asia and Africa (see Chapter 4). Even in the relatively non-seasonal equatorial regions, extreme climatic conditions, such as those caused by El Niño Southern Oscillation (ENSO) events, may have a pronounced effect on defecation rates of elephants in forests. Therefore surveyors of forest elephant populations who wish to calculate the number of elephants from dung surveys should only count dung dropped during typical climatic conditions if they want to apply the 18.07 defecations per 24-hour rate recommended here (or make comparisons between years as per the second option recommended below).
- **For strongly seasonal areas with no available defecation rate data and for forest areas where the assumptions made above are considered inappropriate** There are no data from strongly seasonal areas of Southeast Asia, but such data do exist for southern India, and these data (together with those from Africa) suggest that defecation rates in these areas show major seasonal variation because of the large variations in the protein, fibre, and moisture content of elephant food stuffs [Guy 1975; Barnes 1982; Dawson 1990]. For such seasonal areas,

Box 9.3 *Example of the data coding and entry methods necessary if you wish to use the free software for estimating dung pile decay rates and the Laing et al. (2003) method recommended in this manual*

- Decide which dung pile decay classification stages (e.g., S1–S3) are considered to be 'still present' (or what height a dung pile needs to exceed to be considered 'still present') and give all those 'still present' dung piles a STATE code of 1; all other dung piles (e.g., those in stages S4 and S5) are then considered to have 'decayed' and are given a STATE code of 0.
- Enter the data into an Microsoft Excel worksheet in two columns headed DAYS for the age of the dung piles in days, and STATE for the decay status, for example:

DAYS	STATE
50	0
50	0
50	1
50	0
50	0
42	1
42	0
…	…
etc.	etc.

- Start the Genstat computer program and read in the data from the Excel spreadsheet prepared as above.
- Select Open from the File menu and the select Files of type: Other Spreadsheet Files.
- Open the Excel workbook and select the worksheet containing the data.
- Click Next twice.
- By moving columns with the -> and <- buttons, ensure that the Selected Columns are just DAYS and STATE.
- Click Finish and then OK.
- Next load the program in the file mean decay.gen.
- Select Open… from the File menu and select Files of type: Genstat Files.
- Open the file mean decay.gen.
- Select the window with the mean decay.gen program file and run the program by clicking Ctrl-W.

The estimated mean decay time, its standard error (SE), and coefficient of variation (CV) should be at the bottom of the Genstat's output window. Note that the Excel file Dung decay example.xls contains some fictitious data to allow you to familiarise yourself with the method (Appendix 4).

and for forest areas where the assumptions made above are considered inappropriate, we suggest that dung count data be corrected for dung pile decay rate but not for defecation rate, and that the resulting index of population density be used to evaluate trends. For this approach to be appropriate, all subsequent dung surveys would have to be conducted at the same time of year as the first survey, and there should be no significant intra-seasonal variation in defecation rate. (For example, in areas where the elephants' eat large amounts of fruit during certain periods of the year, dung count surveys should be conducted outside the period of fruit availability. This is because it is suspected that eating large quantities of fruit increases elephants' defecation rates.) Providing these conditions are met the indices of population density produced may be treated as direct analogues of absolute population density [Hedges and Tyson 2002; Tyson et al. in review]. Another and probably superior alternative would be to use fecal DNA based capture–recapture methods (Chapters 5 and 10), rather than the dung density based methods described in this chapter.

9.4.2 Recommended methods for defecation rate studies

Additional studies of elephant defecation rates are needed in both strongly seasonal environments and more constant ones. There are good opportunities for more work in both Southeast Asia and South Asia (including the tracking of captive elephants when they are foraging in the wild). If possible, at least one forest site in Southeast Asia (Myanmar or Thailand) or one forest site in South Asia (India or Sri Lanka) should be selected for further defecation rate monitoring studies in strongly seasonal environments. At least one more site in a stable forest environment outside of Indonesia would also be desirable to supplement the work conducted by WCS in Sumatra [Hedges et al. 2003; Tyson et al. in review]. At least one additional study in the forests of Central Africa would also be desirable.

If you have the opportunity to conduct a defecation rate study the following recommendations should be kept in mind:

- Tame captive elephants, such as those held in timber camps, can be used for defecation rate studies, but it is important that the animals are allowed to feed on a natural diet by foraging freely in typical wild elephant habitat.
- Defecation rate data should not be collected for the first three days of any study in order to ensure all foodstuffs consumed prior to the beginning of the period of feeding on a natural diet have passed through the animals' digestive systems [W. Karesh pers. comm.].

- Monitoring should be conducted over continuous 24-hour periods to account for diurnal/nocturnal variation in defecation rates [Ananthasubramaniam 1992; Tyson et al. in review] and unexpected peaks in defecation.
- Extrapolation from short observation periods (< 1 week in length) may be unrepresentative and should be avoided.
- Data should be collected from both male and female elephants, and if possible from juvenile animals as well as sub-adults and adults (2 adults, 2 sub-adults and 2 juveniles is the recommended minimum).
- Moving tame captive elephants into wild elephant habitat in order to study defecation rates is potentially dangerous to the health of the wild elephants, since diseases and parasites may be introduced to the wild population. Therefore, suitably qualified veterinarians should check the health of the captive animals before they are allowed to feed in areas with wild elephants.
- Obtaining defecation rate data from wild elephants in forests is difficult and potentially dangerous, and should only be attempted with the aid of experienced trackers (see, e.g., Tchamba 1992; Theuerkauf and Ellenberg 2000; Nchanji et al. 2008). Even with skilled field personnel, there is a risk of error if the team misses defecations produced by the target group, or includes old dung piles or dung from non-target elephants (Plate 9.8).
- Estimating defecation rates from observations at waterholes or saltlicks may produce biased data and should not be used [e.g., in Mudumalai Wildlife Sanctuary in southern India, Watve (1992) found a defecation rate at waterholes and saltlicks of 1.15/hour, while in the forest it was only 0.66/hour].
- For all studies, whether involving captive or wild elephants, it is essential to seek advice from people with experience of monitoring elephant defecation rates during the planning stage.

9.5 Estimation Of Elephant Age From Dung Dimensions

9.5.1 Preamble

As already discussed in Chapters 2 and 4, knowing the age structure of elephant population is very helpful for estimating the impact of legal and illegal killing, captures or changes in habitat extent and quality. Furthermore, the dimensions of dung boli (diameter and circumference) have been shown to be related to elephant size, and thus age, and to provide a practical way of assigning age classes to elephants when this cannot be done more directly (Chapter 4).

Elephant dung bolus circumference and diameter are relatively easy to measure during line transect (and other) dung surveys, and fortunately neither diameter nor circumference change appreciably with time for those boli which remain intact [Reilly 2002; Tyson et al. 2002; Hedges et al. 2003]. Reilly suggests that measurement of the greatest diameter is simpler and more precisely measured in the field. However, comparisons of bolus circumference and diameter showed that the coefficient of variation for circumference was always smaller than that for diameter [Tyson et al. 2002; Hedges et al. 2003].

It would appear, therefore, that valuable data on population age structure can be collected relatively easily during dung surveys if dung dimensions are measured and following Hedges and Lawson (2006), we recommend that dung boli circumferences are recorded during all line transect based dung counts. However, the following limitations should be kept in mind:

- Where the sex of an elephant that produced a dung pile is unknown, and where sexual dimorphism in the population is pronounced, there is a problem in assigning an age class from dung size. For example, a dung pile with mean bolus circumference of 45 cm may be from a mature female or from a sub-adult male Sumatran elephant [Tyson et al. 2002; Hedges et al. 2003]; but this can be overcome by collecting fecal DNA samples from the measured dung piles (see Chapters 5 and 10).
- Small boli may be overlooked (or have higher decay rates) potentially leading to underestimation of the number of juveniles in the population [Jachmann and Bell 1984].
- To reduce errors due to seasonal effects (e.g., dietary changes that might affect dung form), inter-year comparisons of age structures based on bolus size should only use data collected in the same season of each study year.

Possible methods for dealing with these potential problems are discussed in Chapter 4 (see also the discussion of size-biased sampling in Section 4.3.5).

9.5.2 Collection of dung circumference data

- Measuring of dung boli circumference should be part of the routine procedure carried out when dung piles are encountered during line transect surveys.
- For all dung piles found with intact boli (for definition of 'intact' see Table 9.1 and Section 9.2.2) the circumference of the three largest intact boli should be measured. If only one or two intact boli are present in a dung pile they should still be measured.

- The maximum circumference of each bolus should be measured to the nearest centimetre using a flexible plastic tape measure; boli may need to be inspected carefully to make sure the correct axis is measured (Plates 9.9 and 9.10).
- If resources permit, fecal DNA samples should be collected from all measured dung piles (see Chapter 10).
- The location of each dung pile should be noted by recording the distance along the line transect (from the HipChain reading, see Section 9.6.3.6).
- Dung circumference data should be recorded along with the line transect data on standard datasheets (see Appendix 1 and Box 9.5).

9.6 Estimating Dung Pile Density using Line Transects: Survey Design and Field Methods

9.6.1 Introduction

As we saw in Chapters 3 and 4, for those areas where elephant dung piles are more easily detected than the elephants themselves (e.g., because the elephants live in concealing habitat types such as dense forests, at low density, or move away before they can be seen) line transects provide an efficient method for estimating elephant density from dung pile density, dung decay rates and defecation rates. The general principles behind estimating density from line transect data are explained in detail in Chapter 3 and in Buckland et al. (2001; 2004), and so we focus here on matters specific to dung count based surveys.

As with sighting based line transect surveys (Chapter 7), it is important to ensure that the key assumptions of line transect sampling (Chapter 3) are adequately satisfied during dung count based surveys. In this chapter, we describe how you can attempt to achieve a balance between the theoretical needs of line transect sampling and some of the practical problems that occur in the field.

9.6.2 Meeting line transect assumptions in field surveys

Obtaining valid abundance or density estimates using dung count based line transect sampling depends on satisfying the following key assumptions (Chapter 3):

1. An adequate number of line transects are located randomly with respect to the distribution of the elephants.
2. Dung piles on the transect line are detected with certainty.

3. Elephant dung piles are identified correctly and measurements made are exact.

These assumptions can be relaxed in some situations, as discussed in Chapter 3, but it is preferable to design and conduct surveys to meet the assumptions whenever possible.

9.6.2.1 An adequate number of line transects are located randomly with respect to the distribution of the elephants

As we saw in Chapter 3, by locating the line transects according to a well-defined survey design there is no need to assume that elephants in the population being sampled are randomly distributed in the study area (an assumption that is unlikely to be true). Random (or systematic with a random start point) placement of an adequate number of line transects (the exact number depends on the variability in elephant density over the region of interest: 25–30 replicate lines is a reasonable recommendation, but sometimes 15–20 lines may suffice) by means of the survey design algorithm in program DISTANCE [Thomas et al. 2010] helps ensure valid statistical inference.

9.6.2.2 Dung piles on the transect line are detected with certainty

Distance sampling along line transects is based on the assumption that all dung piles are detected if they are on the line (i.e., if the perpendicular distance from the line is zero). If this assumption is not met, because dung piles whose centres are on or very near the line are missed, then our estimates of elephant abundance or density will be negatively biased as the proportion of groups counted will be underestimated. Thus it is very important to design surveys to satisfy this assumption. Steady rather than erratic progress along the transect line, a balanced search effort that ensures the transect line is well covered but not at the expense of greater distances, and using observers with a well-developed search image for dung piles (especially dung piles that have broken-down during the decay process) will all help ensure that dung piles on or near the line are not missed. There may also be advantages to experimenting with a double observer system [e.g., see Jenkins and Manly 2008]. It is therefore clear that line transect surveys require well-trained and highly motivated field staff who understand the importance of the assumption [Milner-Gulland and Rowcliffe 2007]: this also applies to the next assumption.

9.6.2.3 Elephant dung piles are identified correctly and measurements made are exact

For a dung count based survey, the perpendicular distance from the transect line to the centre of a dung pile should be recorded. Since elephant dung piles consist of a number of boli ('balls' of dung), which can often be scattered over considerable distances and intermingled with boli from other dung piles in the same general locality, it is important to take great care when deciding which boli belong to which dung piles. Once the dung piles and their constituent boli have been identified, the geometric centres of all dung piles have to be determined so that the perpendicular distances from the line transect's centre line to the centres of the dung piles can be measured. A steel measuring tape should be used to make all measurements because fabric tapes stretch. Detailed practical guidance on these critical points is provided in Section 9.6.3.6.

9.6.3 The need for pilot surveys

When planning a line transect survey in a new area, always do a pilot survey first – otherwise your first real survey will effectively become a pilot survey! The pilot survey will allow you to estimate the total line length required given the dung pile cluster encounter rate found during the pilot survey and the desired level of precision (see Chapter 3). Indeed, the pilot survey may show you that it is not possible to achieve the desired level of precision given constraints on resources and time and so you may need to adopt another method or redefine your monitoring objectives. Finally, a pilot survey will help you assess the logistic and other challenges presented by the survey area and may help you identify strata, depending on the extent of your pilot study. Previous survey reports, topographic, vegetation or landscape ecology maps, expert opinion and local knowledge can all help stratify a survey area and should all be considered at this stage.

Once you have completed the pilot study, you should also manually plot a histogram of the recorded distances (or better still, use program DISTANCE to do this) and check for problems such as a spike at zero distance and rounding of distances (Figure 3.4).

9.6.4 Line transect survey design

9.6.4.1 Design essentials

A valid random (or systematic with random start point) survey design as described in Chapter 3 is necessary if we are to be able to extrapolate from what we estimate from the transect lines to the study area as a whole. Without such a design one needs to assume that elephants in the study area are randomly distributed, which is unrealistic, or one has to resort to model-based inference, which relies on the possibility of fitting an unbiased model to the survey data. Thus, the simplest and most robust option is to use a random survey design. Fortunately, program DISTANCE has an easy-to-use automated survey design feature [Thomas et al. 2010], and this was described in Chapter 3.

As we discussed earlier, surveyors in the past often used roads, firebreaks and other convenient linear formations as transect lines, but this should be avoided absolutely in order to avoid seriously biased estimates of elephant density.

As we also discussed in Section 7.3, the more spatially replicated lines in the survey design there are, the more likely it is that you will obtain a representative sample and a reliable estimate of variance. However, we recognise that elephant habitats can often be difficult to access, and so there are often high costs and logistical problems associated with having large numbers of spatial replicates. The number of spatially replicated transects one can establish will depend, therefore, on the terrain, resources (manpower and money), and time available to the survey team. The logistical problems associated with difficult to access areas can potentially be reduced by stratifying the study area by ease of access (or survey cost) as explained in Section 3.4. Stratification can also improve the precision of our estimates and provide information about elephant abundance or density in different management units, habitat types, etc. (Chapter 3). Note that it is also possible to post-stratify (e.g., by habitat type) after the survey has been completed.

If possible, you should stratify the site by expected elephant density (because such stratification will help improve the precision of your population estimates). This means that you should divide the site into two or three areas: low- and high-density strata; or low-, medium- and high-density strata. The results of a pilot survey should allow you to define these strata (Chapter 3; Section 9.6.3). Stratification by distance from areas of human activity is sometimes a sensible alternative (particularly if one has few data on elephant distribution in advance of the survey) because elephant density often varies with distance from human disturbance, especially in African forests [Barnes et al. 1997; Blake et al. 2007; Blake et al. 2008; Yackulic et al. 2011], and distance from human disturbance is also often inversely correlated with ease of access (and thus survey cost). Another useful alternative is stratification by habitat type as one might expect both elephant density and the probability of detection to change by habitat type. In addition, if you have been forced to adopt a 'find and move' approach to dung decay rate monitoring and have selected a number of different habitat types in which to search and locate dung piles (Section 4.4.2.1) then it may be appropriate to stratify by habitat-type (but you should seek advice from suitably experienced statisticians or other wildlife population monitoring experts). However, stratification by habitat type is only possible if spatially explicit information on habitat is available (i.e., you have maps or satellite images showing habitat types) and the habitat types are not too fragmented and inter-digitated so as to make stratification by habitat type impossible at the design stage. In these situations, a possibility would be to use the record of habitat type changes along your line transects (see Box 9.4) to stratify your survey: i.e., one would then have a measure of the amount of survey effort spent in each habitat type, which would allow post-stratification by habitat type during the analysis stage. Again, seek advice from suitably experienced statisticians or other wildlife population monitoring experts.

Finally with regard to stratification, it is appropriate to repeat the words of caution in Chapter 7 here, since they apply to dung based as well as sighting based surveys. Specifically, stratification will only give modest increases in precision unless the differences in density between strata are large; and there is also the danger that too little effort will be allocated to, or too few dung piles seen in, a low density stratum, so that that reliable estimation of abundance in the stratum cannot be achieved [Buckland et al. 2001].

As we saw in Chapters 3 and 7, for each density estimate we generate there should ideally be at least 15–20, but preferably 25 spatially replicated transect lines (and in some cases, we have found that 30–40 replicates are needed) [Buckland et al. 2001; Thomas et al. 2010]. If your goal is to obtain one density estimate for the entire area, then these replicate lines should sample the entire area. On the other hand, if you need separate density estimates for subareas such as management units, habitat types, or other strata, then you must have at least 15–20 but preferably 25 or more replicates in each stratum (again, in some cases, we have found 30–40 replicates to be necessary per stratum). Smaller numbers of spatial replicates may require estimating the variance of encounter rates theoretically rather than empirically (see Chapter 3 for a fuller discussion). This is generally considered a less desirable option than generating variance estimates empirically from your data [Buckland et al. 2001].

Using the method introduced in Section 3.4.5, one can estimate the total length of transect required for a given encounter rate and a desired precision. The general approach to estimating the total line transect length necessary to achieve a specified coefficient of variation from a pilot survey is given by Buckland et al. (2001) and also discussed in Chapter 3. For the case of a dung count based line transect survey, the relevant equation is:

$$L = (b/\{cv_{\mathrm{t}}(\hat{E})\}^2).(L_o/n_o),$$

where

L = estimate of total line length to be surveyed to achieve target coefficient of variation, BOX 9.4 *Data to record along line transects*

	Record distance from start point using a topofil	Record GPS location
All elephant dung piles found (see below for further detail)	Yes	No
Any sightings of elephants	Yes	Yes
Any elephants heard	Yes	Yes
Any elephant carcasses found	Yes	Yes
All elephant trails that cross the line transect route	Yes	No
Any logging roads that cross the line transect route	Yes	No
Any crop fields or other agricultural activity encountered	Yes	Yes
Poachers' camps	Yes	Yes
Any other signs of human activity (e.g., snares, gun cartridges)	Yes	No
Any saltlicks	Yes	Yes
Any streams or small waterholes	Yes	No
Any ponds or lakes	Yes	Yes
Transitions between major vegetation types (habitat types)	Yes	No

Why take GPS reading for some things and not others?

Dung piles: it is far too time-consuming to take GPS readings for every dung pile and in any case unnecessary as the topofil indicates how far along the line any dung pile was found and GPS fixes will not be used in any analyses.

Animal trails or logging roads crossing transects: same as for dung piles (too time-consuming, precise coordinates are not needed and the information is available from topofil readings).

Streams: as for dung piles and animal trails.

Transitions between major vegetation types: again the topofil reading provides the location data at an appropriate resolution; waiting for a GPS reading at every vegetation type transition will waste lots of time for no gain in useful data. However, taking waypoints for carcasses and poachers camps enables these to be relocated by the protected area authorities or other wildlife authority.

b = dispersion factor (= variance inflation factor),
cv_t = target coefficient of variation,
$\hat{E}$ = density estimate,
L_o = total length of all pilot transects combined, and
n_o = total number of dung piles found on all pilot transects combined.

Estimation of b poses some difficulty from a short pilot survey, however Eberhardt (1978) provides evidence that b would typically be between 2 and 4 independent of n; and Burnham et al. (1980) provide a rationale for values of b in the range 1.5–3. Both Burnham et al. (1980) and Buckland et al. (2001) recommend the use of $b = 3$ for planning purposes. However, as more people do pilot surveys followed by dung surveys, we should be able to estimate better values of b for elephant dung counts. For example, Yaw Boafo and Nandjui Awo used $b = 3$ in their recent survey of Sapo, and their results suggest it may not have been the best value to use (R. Barnes pers comm. 2010).

As an example, let us consider a short pilot survey where $L_o = 40$ km and $n_o = 160$ dung piles (an encounter rate of 4 dung piles / km). If we use $b = 3$ and a target coefficient of variation for our dung pile density estimate, $cv_t(\hat{E})$, of 0.10 (= 10%), we see that the estimated total line length that we will need to survey to achieve the target coefficient of variation is.

$$L = (3/(0.1)^2).(40/160) = 75.0 \text{ km.}$$

Whether conducting a survey with 75 km of line transects is achievable with the resources available and within the necessary time frame for the site in question must then be assessed. The results of the pilot survey will also inform this decision since they will allow one to estimate the time required to conduct surveys in the site.

More generally, using a value of 3 for the dispersion factor (variance inflation factor), b, as suggested for planning purposes in Chapters 3, together with a target coefficient of variation (CV) of the estimated dung pile density estimate of, say either 25% or 10% allows us to estimate total line length for a range of encounter rate values (Table 9.1). For example, for an expected elephant dung pile encounter rate of one dung pile per kilometre obtained from a pilot survey, an estimated 300 km of line transects would be required to obtain an estimate of dung pile density with a CV of 10 percent.

Walking at a speed of about one kilometre per hour (1 km/h) generally enables efficient detection of elephant dung piles: faster speeds can be problematic (see Section 9.6.3.2). However, from practical experience in forested areas of often difficult terrain, it usually takes about one full day to conduct a 1–2.5 km long dung transect (when one factors in the time taken to get to the transect, the time spent categorising dung piles, making

TABLE 9.1 *Total survey effort in kilometres of line required for line transect surveys to achieve a desired coefficient of variation (CV) with different dung pile encounter rates*

CV	Encounter rate (dung piles / km)									
	1.0	2.0	3.0	4.0	5.0	6.0	7.0	8.0	9.0	10.0
25%	48.0 km	24.0 km	16.0 km	12.0 km	9.6 km	8.0 km	6.9 km	6.0 km	5.3 km	4.8 km
10%	300.0 km	150.0 km	100.0 km	75.0 km	60.0 km	50.0 km	42.9 km	37.5 km	33.3 km	30.0 km

measurements, etc.); indeed sometimes two full days will be needed for such transect lengths. Thus, as an example 300 km of line transects may require about 300 team/days (or in other words, 100 days of actual survey work would be required by three teams working simultaneously in order to complete the target transect length).

The automated survey design component of program DISTANCE should be used to design the surveys taking into account the total survey effort required to meet the target level of precision, the desired number of strata, the effort to be allocated to each stratum and the time and resources available. DISTANCE permits the selection of a design from among a number of different possibilities and the exploration of the design properties given the logistical constraints for the survey in question [Thomas et al. 2010; see Chapter 3]. Note that to use program DISTANCE to design the survey one has to define the survey area in a spatially explicit manner by means of a GIS shapefile. [An additional related resource is available online from the IUCN/SSC Primate Specialist Group, see http://apes.eva.mpg.de/fr/pdf/guidelines/IUCN_SGA_Monitoring_Section_3_Survey_design_For_web.pdf.]

The type of survey design generally recommended for dung count based surveys is a 'systematic segmented line transect design' with a random start point. This design involves placing a systematic arrangement of track-lines across the study area and then locating line transects of specified length systematically along it to meet the desired total length requirements of the survey.

The line transects should be short, typically 1–2.5 km in length, unless another line length has been recommended for your site by suitably experienced statisticians or other wildlife population monitoring experts. The line length should be long enough to reduce the probability of too many transects having no dung piles and thus depends on the encounter rate of elephant dung in the pilot study. We note, too, that many wildlife surveys in, for example, Central African forests will also include other wildlife species such as apes as the cost of accessing these areas is so high. In such situations, the pilot study may have shown that even if a 1-kilometre transect length is sufficient for elephant dung, it will not be enough for ape nest density estimation, and thus the length of each transect will have to be increased.

There is often a temptation to design a survey with a small number of long transects because that is more efficient from the logistic point of view. But from the statistical point of view (i.e., the power of the design to detect changes in elephant abundance), it is more efficient to have a large number of short transects. Many surveys in Central Africa have, in recent years, had transects 2–4 km long, with a minimum of 30 transects per stratum, to increase precision as far as is possible within logistical constraints.

Although a 'systematic segmented line transect design' provides transect segments that are occasionally less systematically placed than those given by 'systematic segmented grid sampling' designs, it tends to spread segments over a greater range of any potential density gradient. This provides for good spatial coverage of the site and is likely to yield a representative sample leading to a more precise estimate of dung density. The random start point used in the design permits a standard analysis using design-based estimators (see Chapters 3) and analysis of this type of systematic design generally proceeds as if the position of each transect were randomly selected. Although theoretically this is an issue, in practice systematic spacing of transects provides better spatial coverage within the survey area, and therefore an improvement in density estimate compared to randomly placed transects.

Ideally, to achieve greater precision one should orientate transect lines parallel to any gradients of density, so that any variation in encounter rate is maximised within transects and minimised between them. So, for example if one suspects that density decreases with increasing distance from a habitat feature such as a major river then transects would be placed approximately perpendicular to the river. If one suspects that density increases with increasing distance from a road, and if it is likely that the road has a stronger influence than water features, then transects should be placed perpendicular to the road. If both are suspected to have equal influence, then transect orientation should take both into account.

In conclusion, we strongly recommend that elephant surveyors: (1) apply the sampling and survey design considerations explained in Chapter 3; (2) exercise caution in developing any strata (and avoiding over-stratifying); (3) use the automated survey design feature of program DISTANCE to experiment with the properties of different designs given the logistical constraints for the survey in question; (4) establish their own transect lines over the area to be surveyed and never follow existing trails, roads, etc.; and (5) follow the guidelines for establishing transects presented below.

9.6.4.2 Survey design implementation problems related to map inaccuracies

In some areas, the best maps currently available are of relatively poor resolution: in some countries in Central Africa for example, paper maps typically have a scale of 1:200,000 and were generated from aerial photographs taken in the late 1950s or 1960s. Even when satellite images are available they too often suffer from a lack of geo-referencing accuracy, and cloud cover often obscures large areas. These errors can lead to a number of problems for implementing a survey design:

- Imperfect geo-referencing of paper and digital maps may result in there being a geographical (positional) 'shift' between features on the maps and their actual position on the ground. In reality, then, survey locations (transects) will sometimes not be where they 'should' be according to the maps used to design the survey. A 'shift' of 200 m is not unusual, and occasionally a 'shift' of a kilometre has been recorded.
- Imperfect knowledge of the terrain (e.g., rivers may have changed course since the maps were made or, more commonly, cloud cover in the original aerial photographs resulted in a complete lack of detail for large parts of the map) can result in transects being in inappropriate or impossible to survey places.
- Inaccurate or out-of-date information about land use including roads and human settlements can also result in transects being in inappropriate or impossible to survey places.

The combination of these problems can lead to considerable problems for those executing the surveys in the field. Transect routes may, in reality, fall in swamps, span large rivers, fall half way up a 75° rock-face or even be completely outside the study area. Most of these problems will not be encountered until the survey teams are on the ground and so a set of easy to follow and unambiguous protocols must be developed to account for these issues. Whatever is decided will bias the survey design by some unknown amount, but without perfectly geo-referenced maps that include all swamps, all inaccessible areas and other relevant landuse types, these problems are unavoidable. Some suggestions for addressing these problems are given below (also see Section 9.6.3.4).

a. *Transect location is outside the study area*

   If the start point of the transect is inside the study area but it crosses the area's boundary along its length, the transect should be 'reflected' back at a 30% angle when the boundary 'edge' is reached (Figure 9.1).

b. *Transect location wholly outside study area or wholly within a swamp*

   If a transect is wholly outside the study area or wholly within a swamp, the transect should be relocated to the closest estimated location with dry land within the study area (Figure 9.2). Selection of the closest area will be somewhat subjective because field teams will not have an accurate way to find the closest locations, however from maps and the lay of the land, they should have an idea of which direction the closest area of dry land may be found.

9.6.5 What is the best time to conduct dung counts?

You should aim to conduct both the pilot survey and the subsequent formal survey during the same time of year (or at least during the same season) in order to minimise the problems caused by seasonal elephant movements. For example, the geographical limits of high- and low-density strata may vary significantly between wet and dry seasons in some areas. In addition, dung

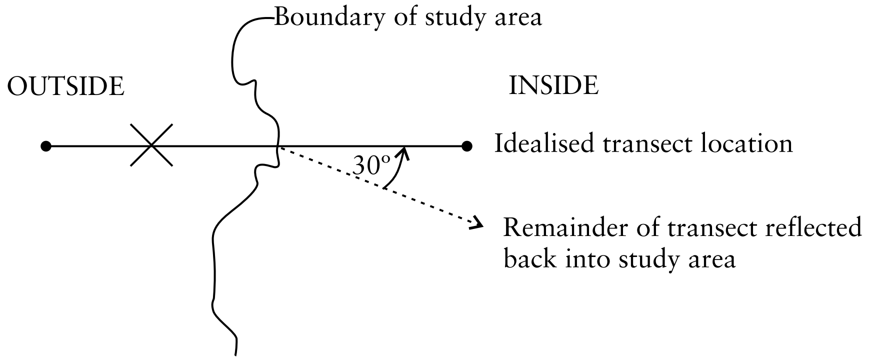

Figure 9.1 *Suggested protocol for those situations in which a transect location falls partly outside the study/survey area. Reproduced with permission from Blake et al. (2002).*

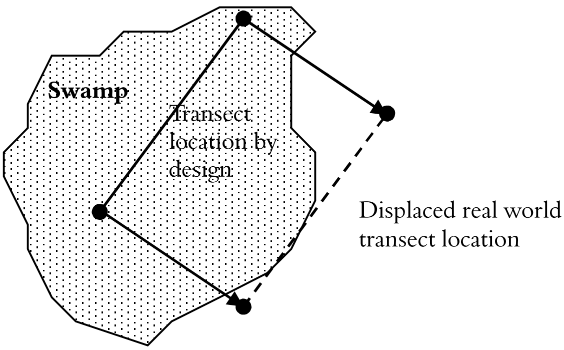

Figure 9.2 *Suggested protocol for those situations in which a transect location falls wholly in a swamp. Reproduced with permission from Blake et al. (2002).*

count based surveys should not span seasons in order to avoid problems caused by possible significant seasonal changes in elephant defecation rates. If the area is seasonally inundated, then conduct the count in the dry season (see Section 4.8.3). In areas where the elephants eat large amounts of fruit during certain periods of the year, dung count surveys should be conducted outside of the periods of fruit availability.

9.6.6 Conducting the survey: step-by-step field methods

9.6.6.1 Locating the survey tracklines and line transects

- Once a survey design has been decided upon, the survey teams should be provided with a database of coordinates for both the start and end point of tracklines (the lines linking the line transect sections, see Chapter 3) and the line transects themselves, as generated by program DISTANCE. These coordinates should be uploaded into appropriate GPS units and used for navigating in conjunction with site maps, aerial photographs or satellite images. We have found it useful to superimpose the transects and waypoints (both appropriately numbered) onto a topographic base map of the survey area, preferably at 1:25,000–1:100,000 scale, with any additional known roads or village data added (in a GIS), and a grid of latitude/longitude lines (at one-minute intervals, if possible) or UTM grid superimposed. The resulting maps can then be printed and laminated (back-to-back maps save weight) see Plate 9.11.
- Each team should carry two GPS units, one for daily use, and one as a backup. (Use long-life batteries for the GPS units; although expensive, they are more cost-efficient; see Appendix 2.)

9.6.3.2 Team composition, walking speed and other general considerations

- Ideally, each line transect team should be composed of a minimum of four people: one cutter, who cuts a straight-line path through the forest; a navigator, who directs the cutter; a dung pile spotter; and a data-recorder. Everybody should keep their eyes open for dung piles and tell the data-recorder if they see any. (However, if you are combining dung pile surveys with searches for other species of interest, e.g., ape nests or terrestrial mammals, there should be a dedicated spotter for these other species who does not look for dung piles). Depending on circumstances, these jobs can be rotated provided everybody knows how to use a sighting compass (in Asia, we have typically conducted elephant-only surveys and the jobs are rotated, in Central Africa we have typically conducted multi-species surveys and the jobs are not rotated). Two cutters may be needed in very dense forest to prevent people from becoming exhausted.

- 1–2 field assistants who will help transport essential survey equipment and with measurement taking, such as perpendicular distances, may also be added to the team if necessary. During transect cutting and surveying the rest of the team (porters) should remain at the start point and follow on some time later so as not to get in the way of the survey team.
- When starting a line transect survey the data recorder should, record the following data on the transect data sheet: date, time, transect number, weather conditions, and team members' names. A specimen blank field datasheet for use in line transect surveys is shown in Appendix 1: this form or a similar one can be duplicated (ideally on waterproof paper) for use on line transect surveys. An example of a completed datasheet is provided in Box 9.5. A list of the equipment required is given in Appendix 2.
- The data-recorder carries a topofil (such as a HipChain; Plate 9.12) to record the distance walked from the start point as well as a GPS.
- GPS coordinate data should be recorded at the beginning and end of every line transect. In addition, GPS coordinate data should be recorded every 500 m along the transect. [In some areas, there are problems obtaining GPS signals. For example, in our Lao surveys, the teams often had trouble locating the end of the transect due to forest cover (although with modern GPS antennae this is a rapidly declining problem). In the Lao case, we suggested walking a straight line to a more open area on a compass bearing and using the hip chain to measure the distance.]
- Line transects will be of varying length (but will typically be 1–2.5 km) depending on the pilot study results and the resulting survey design adopted (see Chapter 3 and Section 9.6.4).
- The speed at which the teams complete the transects will depend on terrain, the vegetation types encountered, and the amount of elephant dung found. However, as a guide, transects in forested areas should not be conducted faster than 1 km per hour. For grasslands, the maximum speed should be 45 minutes per 1 km. At speeds faster than these, it is likely that the teams will miss a significant number of dung piles.
- Data collection on transects should be suspended during rain, very overcast conditions or other times of poor visibility. Furthermore, transect data should not be collected too soon after dawn or too close to dusk due to poor visibility. In equatorial Africa, we recommend that observations are not started before 0700h, and are stopped at 1530h.

9.6.3.3 Cutting the line transects

- At least two people are needed to cut a line transect: a cutter and compass person who directs the cutters (Plate 9.13). Following the methods described by White and Edwards (2000) and Hedges and Lawson (2006) the compass person should cut a straight pole and trim the top end so that it is absolutely flat. The bottom end should be cut to a point. It can then be firmly planted into the ground without physical contact by the compass bearer to keep it upright. The stake should be long enough so that the compass is at eye level. The sighting compass is then placed on the stake and oriented so that one can sight through it without touching it (Plate 9.14). Guided carefully by the compass bearer, the cutter then traces a path away from the compass person cutting the absolute minimum amount of vegetation necessary to show the trajectory of the transect and to allow the team to follow in single file. The compass person must carefully monitor the cutters. Each time a cutter deviates from the path the compass person should immediately call out a correction (left or right). The compass person must be very strict, as cutters often tend to deviate along animal trails or around dense thickets: this will result in biased data. When it becomes difficult to see the cutter, the compass person should tell them to stop and then move forward to where they are waiting (the cutter must not move in the meantime). It is useful if the cutter wears a white, red or pink cap, so that the compass bearer can see him or her for as far as possible through the undergrowth before moving forward. The further the distance between the cutter and the compass bearer, the more accurate is the bearing. *A well defined, straight transect is crucial for good data collection and to reduce biases.*
- It is important also to emphasize that the cutters should cut just the minimum needed to allow the team to walk through the forest. They should not make a path or trail. Cutting too much vegetation along the route is bad because it damages the forest, it allows poachers easy access, it wastes time and energy and falling branches and leaves can cover dung piles making them difficult to see.
- Once the path is identified and the compass person has moved on to where the cutter is waiting, the rest of the team should move forward while looking carefully for dung piles.
- Since it is likely that the transects will form the basis of a long term monitoring program at each site their start point should be well marked using metal plates. During the transect survey, the compass bearing must be accurately recorded and GPS fixes <u>must</u> be taken at the start and end of every transect.

9.6.3.4 Dealing with obstacles along line transect routes

- Remember, it is essential to maintain a completely straight line. The following guidelines for dealing with obstacles should be used.
  - Thickets: cut through these obstacles (see below for dense bamboo thickets).
  - Water bodies such as flooded grasslands that you can wade across: continue the transect as normal, but record the distance on the topofil at the point you begin wading and at the point where you stop wading. This is because we need to know that, for example, 300 m of the transect route was under water when calculating dung pile encounter rates (Plate 9.15).
  - Water bodies such as deep pools, lakes, etc. that you cannot wade: record the GPS location at the edge of the water, identify a landmark the other side of pool/lake, break the topofil thread, walk around the pool/lake to the landmark, record the GPS location at the landmark, then, using the GPS, calculate the distance across the water body that should have been walked, and record how far this was, then continue as normal. Importantly in this case, because the thread has been cut, the extra distance after the water's edge is included in the total distance on the HipChain counter.
  - Dense bamboo thickets that you cannot cut through: record the GPS location at the edge of the thicket, identify a landmark on the other side of the thicket, break the topofil thread, walk around the thicket to the landmark, record the GPS location at the landmark, calculate the distance across the thicket that should have been walked (as above for water bodies), then continue as normal.
  - Major rivers, gorges, cliffs, etc. that you cannot cross: continue the transect up to the hazard, record the HipChain distance and the GPS location and stop the transect. Then, if possible navigate around the hazard and continue the transect as soon as conditions are suitable; if this is not possible move on to the next transect (or go to camp for the night depending on the time of day).
  - Only when conditions become very dangerous (such as on treacherous slopes) should survey teams abandon a transect line. Elephants under heavy hunting pressure in hilly or mountainous terrain often seek refuge in inaccessible areas such as steep slopes. Therefore if these areas are systematically avoided, the abundance estimate will be negatively biased.

9.6.3.5 Data to record along line transects

- Record all of the things listed in Box 9.2, and make a note of the distance from the start point of the transect using a topofil or a GPS as appropriate.

9.6.3.6 Finding and recording elephant dung pile data during line transect surveys

- It is important to keep the number of observers (the people who look for dung piles and, if appropriate, other species) the same for all transects in a survey and for repeat surveys in the same areas. This is necessary to ensure that encounter rates are not artificially inflated or deflated due to fluctuations in survey effort. Thus, all team members can look for dung piles and draw the attention of the data-recorder to any dung piles seen (as is common in Southeast Asia); or, as is more common in Central Africa, the number of observers is restricted to two people: one for dung and other terrestrial objects and one for primates/primate nests.
- For all elephant dung piles found along the transect the distance (in metres) from the start of the transect should be recorded. A topofil (HipChain) is used to measure these distances.
- The spotter and data recorder should examine the dung piles found to determine whether the boli present form one or more than one dung pile. Use the number and size of the boli (especially their diameter), their colour and general appearance, and the distance between the boli to guide your decision (see Section 9.2.2 for further detail). A standard approach to dealing with uncertainty in the number of defecation 'events' (dung piles) needs to be agreed, as this can be a significant source of error according to Jachmann and Bell (1984). The best strategy for dealing with this problem if and when it occurs is to note that the boli found may be from more than one defecation event, and record appropriate distances to the centre points of the probable defecation events. This allows for the transect analysis to be conducted assuming one defecation and then repeated assuming two defecations, for example.
- The perpendicular distance from the line transect's centre line to the centre of the dung pile must be measured using a stick and a steel measuring tape not a fabric measuring tape as these stretch. Remember, it is important to measure the distance from the transect line to the centre of all dung piles in order to minimise the tendency to round perpendicular distances, and especially the rounding of distances to zero for dung piles found near the centre line.

- The topofil thread is used both to measure transect length and to mark the centre line. A common mistake is to count all dung piles which lie in the cut area as being on the transect line and so record a perpendicular distance of 0 cm. This is wrong and will seriously reduce the quality and utility of the data you collect, usually to a degree to which it is impossible to estimate dung density. In fact, a dung pile with a perpendicular distance of exactly 0 cm is extremely rare. For example, of 120 dung piles in Birougou NP in 2007 and 915 dung piles in Lope National Park in 2008, none were exactly on the centre line; so none had a perpendicular distance of 0 cm.
- Next, decide where the centre of the dung pile lies. The easiest way to do this is to create the smallest possible square or rectangular box around the dung pile, using straight canes or metal measuring tapes, and then find the midpoint of the diagonal, X, which will also be the geometric centre of the dung pile (Figure 9.3). In training and supervising dung survey teams, we have found it helpful to encourage the teams to use the measuring tapes to form 4-sided rectangular 'boxes' around dung piles as aid to determining centre points (Plates 9.18 and 9.19). Be aware that elephants often defecate whilst walking and so dung boli

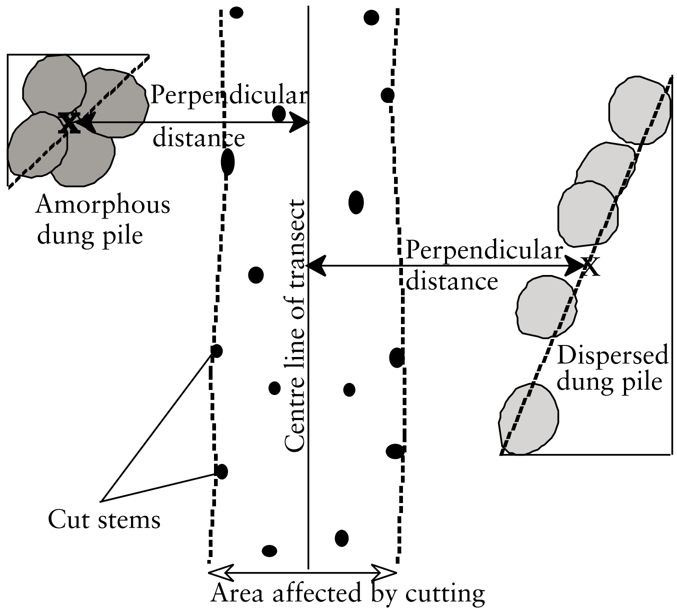

Figure 9.3 *Measuring the perpendicular distance from the centre of a dung pile to the centre line of a transect: X is the centre of the dung pile from which the perpendicular distance to the transect's centre line should be recorded. Reproduced with permission from White and Edwards (2000).*

may be spread over several metres (sometimes 10s of metres). You must look carefully at tracks, and the size, age and composition of the boli to help you determine which dung boli belong together (i.e., form one defecation 'event' / dung pile). Our experiences in Indonesia, the Lao PDR, Malaysia and elsewhere suggest that regular quality-checking and re-training are likely to be necessary to ensure that surveyors continue to correctly identify dung piles, locate centre points properly and measure perpendicular distances accurately.

- Next, identify the point on the transect line from which the perpendicular distance should be measured. This can be done by using a compass set at 90° to the direction of travel and then moving until the dung pile lies on this line. The assistant should cut a slim straight stick of about 1.5 metres long, and position it, upright, at the centre point of the dung pile (Plate 9.16). He or she should hold the tag end of the tape measure. The second observer should run the tape measure up to the topofil, which is kept taut by the first observer (Plate 9.17).
- Once you have located the two points between which the perpendicular distance should be taken, use a steel tape measure to measure it and record the distance in centimetres to the *nearest centimetre*.
- Once the perpendicular distances have been recorded, for all those dung piles having intact boli (Section 9.2.2 and Box 9.1) the circumference of the three largest intact boli should be measured; if only one or two intact boli are present in a dung pile they should still be measured (see Section 9.5). It is important to measure boli after measuring and recording perpendicular distances, to do so before might result in erroneous perpendicular distances if the dung piles are moved during classification.
- Then, the dung pile height should be recorded to the *nearest millimetre* by measuring *the vertical distance between the ground on which the dung pile is sitting and the maximum diameter of the tallest dung bolus in the pile*.
- Finally, the dung pile should be classified into decay stages using the S system (Section 9.2.2). It is important to classify dung piles after measuring and recording perpendicular distances and dung bolus diameters and circumferences; to do so before might result in erroneous perpendicular distances (if the dung piles were moved during classification) or difficulty in measuring diameters/circumferences (if boli were broken during classification).
- A common question is what to do with dung piles detected while surveyors are 'off the line', for example those dung piles which are not detected while surveyors are on the transect line but which are seen while off the line examining a dung pile that was visible from the line. Despite the recommendations to the contrary which are sometimes

made, these dung piles should be treated as any other pile and the perpendicular distances from these piles to the transect line recorded as normal. There is nothing in line transect sampling methodology that assumes that objects must be detected from the transect line (Buckland 2000; Buckland et al. 2001). There are however two potential problems. Firstly, dung piles could be detected at ever increasing distances when examining the more distant piles, which would be wasteful of survey time as these distant piles will contribute little or nothing to the final density estimate. However, this is easily dealt with by setting a maximum distance from the transect line beyond which dung piles are ignored. This maximum distance can be selected from existing line transect data for the area in question or from the pilot study [Buckland 2000; Buckland et al. 2001]. Secondly, there is a potential for bias if many additional dung piles are detected in the vicinity of the original pile. As Buckland et al. explain, this would have the effect of widening the effective strip width in areas of high dung pile density, as additional 'off-line' detections are more likely to be made in such areas. If a constant effective strip width is assumed, this will lead to the overestimation of that width and thus underestimation of dung pile density. The best strategy to overcome this problem is probably to treat all 'off-line' dung piles as 'normal' dung piles (i.e., measure perpendicular distances to these piles) but note whether they were seen off-line. This will allow the frequency of such sightings to be assessed, and thus the likely impact of 'off-line' dung piles on the estimates of dung pile density (if necessary by conducting two analyses of the transect data, one including 'off-line' piles and one excluding them). If 'off-line' piles are found to lead to underestimates of dung pile density they can be excluded from the final analyses.

9.6.3.7 Data collection and management for line transect survey teams

- Record the line transect data on approved datasheets (Box 9.5; Appendix 1). Print these datasheets on waterproof paper or keep them in watertight (ziplock) bags. Alternatively, in some situations we have found that waterproof notebooks are better than datasheets because they are very durable and protect the data well when they are closed. If using such books, before each day's survey (or even each week's survey) the team leader should ensure that appropriate columns have been added to the note book (see Appendix 1). Cybertrackers or some other form of electronic field data-logger may be also be used if these have been shown to be reliable at your site.

- When the teams return to the base-camp/office after each survey trip the team leaders must enter the data into a computer, back it up, and photocopy or scan the original datasheets. The original datasheets should be filed in the appropriate office. Photocopies should be filed at another site. An alternative to photocopying or scanning datasheets is to take digital photographs of the datasheets. The photos should be checked for readability before collating as PDF files and sending to others. The entered data (in Excel or a similar computer program) should be (i) copied to at least two other hard drives at the base, and kept in separate buildings and (ii) sent by email to the site- or survey-coordinator at the end of each field mission, to ensure that if disaster strikes a site a backup copy still exists off-site.

9.6.7 Data analysis and reporting

- Once the line transect surveys and decay rate monitoring work (and defecation rate work if attempted) are complete, calculate dung pile density from your line transect data using the program DISTANCE [Thomas et al. (2010); see Chapter 3 and Buckland et al. (2001) for advice]. We recommend you use the latest version of DISTANCE because, for example, DISTANCE 6 has a more accurate method of estimating some parametres than DISTANCE 5, and so on (see Appendix 4). Make certain that you use the same division of dung piles into those deemed to be 'still present' [typically those in stages S1, S2 or S3 (or those above the selected diameter threshold)] and those deemed to have 'decayed' [typically those in stages S4 or S5 (or those below the selected diameter threshold)] that you used in the dung disappearance experiments.
- Use the free software listed in Appendix 4 (and available from the authors) to calculate dung pile decay rates using the method of Laing et al. (2003); Section 9.3.3)
- When you are happy with your dung pile density estimates, use program DISTANCE to convert your dung pile densities into estimates of elephant density using your estimates of dung pile decay and defecation rates together with their standard errors as multipliers (as explained in Chapter 3).
- Write a report and submit it with together with copies of all your data and the associated computer files to the relevant authorities, donors and other interested parties.

Box 9.5 *Line transect datasheet showing an example of typical field data*

Date (dd/mm/yyyy): 09/12/2004				Line transect number: Example 1	
General description of location: 2 km from Ban Thalang along road to Poong Ta-ee, Nakai Plateau (Lao PDR)					
Start point (UTM): 18534500, 12040000					
Finish point (UTM): 12345678, 12345678					
Compass bearing: 045°				Distance at finish (m): 1 km	
Start time: 0725h				Finish time: 1030h	
Team members' names: Noy, Sithisack, Kanya, Teu					
Distance from start (m)	Dung pile data				Other notes
	Number of boli	Perpendicular distance (cm)	Decay Stage	Circumferences (cm)	
0					Closed canopy forest
26	6	348	S2	N/A	
48					Stream
145	5	29	S1	42, 44, 44	No DNA sample taken; GPS location 87654321, 87654321
256					Animal trail
259					Transition from closed canopy forest to secondary scrub
265	6	79	S3	N/A	
276	5	223	S2	N/A	
335	4	65	S2	35, 36, –	Only 2 intact boli; No DNA sample taken; GPS location 87654321, 87654321
489	6	243	S2	N/A	

491					Animal trail
656					2 snares, old, broken
899	5	3	S2	29, 31, 28	No DNA sample taken; GPS location 87654321, 87654321
924					Transition from secondary scrub to closed canopy forest
999					Very old elephant carcass found, photos # EX1, EX2, EX3; GPS location = 12345678, 12345678
1000					End of transect (still in closed canopy forest)

Most of the datasheet should be self-explanatory. A couple of points of clarification are however included below:

- The 'Line transect number' is that on the survey locations map and the GIS for the site.
- For the 'General description of location' the team leader should write something like 'approximately 2 km from Ban Thalang to Poong Ta-ee road, Nakai Plateau'.
- The 'Other notes' column should be used for recording transitions between vegetation types, elephant sightings, carcasses, etc.

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CHAPTER 10

Estimating Abundance and Other Demographic Parameters in Elephant Populations Using Capture–Recapture Sampling: Field Practices

K. Ullas Karanth, N. Samba Kumar, Varun R. Goswami, James D. Nichols and Simon Hedges

10.1 Introduction

In Chapter 5, we saw that sampling animal populations by repeatedly 'catching' identified individuals can generate capture histories. From these histories, capture frequency statistics and estimates of capture probabilities can be derived. Estimates of capture probabilities permit us to estimate the abundance and density of animals in the surveyed area, as well as other demographic parameters such as survival rates, without 'catching' all the individuals in the population.

In traditional capture–recapture surveys (also known as mark–recapture or capture–mark–recapture surveys), target species such as rodents, birds and fish are usually marked using artificial tags or bands. However, it is not usually possible, or desirable, for us to catch and mark elephants in this manner. Fortunately, currently there are two possible ways of 'catching' individual elephants non-invasively: 'photographic captures' and 'DNA-based captures'.

In the photographic capture based method, elephants can be identified using characters such as the length and shape of their tusks, the patterns of notches in their ears, their tail morphology, natural body markings and the like [Douglas-Hamilton 1972; Moss 1976; Whitehouse and Hall-Martin 2000; Goswami et al. 2007; Plates 10.1 and 10.2]. This ability to identify individuals allows us to use photographs obtained from manual and automatic still cameras or videography to conduct capture–recapture surveys of elephants. We note that some investigators do not maintain photographic identifications, but simply observe and record natural marks in field notes [Turkalo and Fay 1995]. We do not recommend such record keeping and identification methods, although analytical issues (see Chapter 5) are similar to those for identifications made from photographs or DNA.

However, in contrast to species such as tigers and other conspicuously marked animals, it is difficult to obtain elephant photographs and assign individual identities when standard 'camera traps' are deployed [see O'Connell et al. (2011) for a comprehensive treatment]. Typically camera trap photographs will only show a part of an elephants' body and, because elephants are group-living animals, investigators may find one animal obscuring another [Hedges et al. in prep.; Plates 10.3 and 10.4]. Nevertheless, traps that employ still or video cameras can sometimes be deployed at clearings, waterholes, mineral licks and travel routes favoured by elephants by positioning them specifically to obtain useable whole-body images [Varma et al. 2006]. In addition, in some relatively open and accessible environments, it is practical to manually photograph or videograph elephants simply by driving, riding captive elephants or walking along predetermined survey routes; and, by waiting at waterholes, salt licks and other such places that elephants visit frequently including *bais* (forest clearings with water outlets; see Blake 2002) [Goswami et al. 2007; Morley and van Aarde 2007]. Conceptual issues of survey design are covered in chapter 5.

In the fecal DNA capture method, individual elephants are identified from DNA extracted from the cells in fragments of intestinal skin shed when defecating. Thus dung samples are collected to a prescribed survey design and DNA extracted for amplification at several microsatellite loci. Other molecular markers, such as single nucleotide polymorphisms, may also be used [Eggert et al. 2003; Fernando et al. 2003; Eggert et al. 2005; Kongrit et al. 2008]. Matching genotypes are considered to have come from the same individual and are classified as captures or recaptures of these individuals [Kohn et al. 1999; Lukacs and Burnham 2005; Chapter 5].

In addition, fecal DNA survey methods can generate data on genetic diversity within, and gene flow among elephant populations [Eggert et al. 2008; Ahlering et al. 2011]. There are now a number of examples of fecal DNA based capture–recapture surveys of elephant populations from Africa [Eggert et al. 2003] and Asia [Hedges et al. 2007a; 2007b; in review]. These methods are described in the CITES MIKE Program's *Dung Survey Standards* [Hedges and Lawson 2006]. Moreover, the cost of such DNA based surveys is now low enough that they will often be less expensive than traditional dung density based surveys (Chapters 4 and 9) besides being quicker and more informative [Hedges et al. in review].

The above methods (photographic and DNA based) allow us to 'catch' individually-identifiable elephants repeatedly in samples drawn from a wild population. These data can then be used with the powerful capture–recapture analytical techniques presented in Chapter 5 to estimate abundance (population size) and other demographic parameters.

10.2 SAMPLING OCCASSIONS AND DURATION OF CAPTURE–RECAPTURE SURVEYS

All capture–recapture surveys must include two or more sampling occasions. Under closed models, these sampling occasions (or periods) occur relatively close together in time and are used to estimate abundance. In open population models, the time intervals between sampling periods is typically long, as gains and losses to the population are expected to occur. Abundance, survival and recruitment can all be estimated using open models. Under the robust design, sampling occurs at both of these time scales. Secondary sampling periods occur over a short time within each primary sampling period, and primary periods are separated by longer time intervals. Abundance, survival, recruitment and additional parameters (e.g., associated with temporary emigration) can be estimated using the robust design. Under the robust design, the time intervals that separate the secondary periods (i.e., the length of the primary sampling period) are guided by both theoretical and practical considerations. Recall that the critical parameters that are estimated, based primarily on capture histories across secondary sampling periods, are capture probability and the number of elephants present in the surveyed area (abundance). The models that provide the most precise estimates of these quantities assume that the elephant population is 'geographically closed' and 'demographically closed' throughout the primary period (Chapter 5). Therefore, the survey should ideally last for only a few days, as the closure assumption is more likely to be met for such short periods (but see below) although some violation of this premise may occur due to logistical and other reasons.

Selection of the number of secondary sampling occasions represents an interesting design trade-off. More secondary sampling periods provide a better ability to discriminate among competing models of capture probability. In addition, a larger number of periods typically results in greater precision for abundance estimates. However, the more secondary occasions, the greater the likelihood of violation of the closure assumption and of variation among occasions in capture probability. Inclusion of this variation in modelling (e.g., including time as an important source of variation) can lead to lower precision and robustness [Otis et al. 1978; Seber 1982; Williams et al. 2002; Amstrup et al. 2006; Link and Barker 2010]. Nonetheless, having more than two sampling occasions is better than having just two. Thus we see that even if we have enough detectors (dung samplers or photographic devices) on hand to cover the entire survey area in just one day, 10 days of sampling will be required to attain 10 sampling occasions. Usually, observers and cameras are not available in such large numbers and one sampling occasion—each 'round' of sampling across the entire survey area—takes more than one day, depending on the area covered and the sampling design used (Chapter 5). More generally, it will be seen that deciding on the duration of the primary sampling period involves a trade-off between increased model discrimination ability and, sometimes, estimator precision (these are good) and potential violation of the closure assumption and need to model occasion-specific variation (these are bad).

For example, in a photographic capture–recapture survey of adult male Asian Elephants over a 176 km^2 study area in Nagarahole and Bandipur National Parks, the primary sampling period of Goswami et al. (2007) spanned 80 days and comprised 10 secondary samples. Given the wide-ranging behaviour of the species, the length of the primary sampling period resulted in a violation of the closure assumption and prompted the authors to use an open population model for abundance estimation. An alternative approach would have been to reduce the number of secondary samples such that geographic closure was achieved. Later analyses suggest that eight sampling occasions spanning a primary sampling period of 60 days were sufficient to meet the closure assumption and obtain closed population estimates of elephant abundance [V.R. Goswami unpublished data].

In a fecal DNA-based capture–recapture survey of elephants in Thailand, Manopawitr et al. (2008) used sampling locations spread across 960 km^2 of rugged terrain. Each sampling occasion took two weeks to complete—the maximum recommended in the MIKE *Dung Survey Standards* [Hedges and Lawson 2006]—and the entire survey period was about 10 weeks in length. However, because elephants are relatively long-lived animals (compared to, say, rodents), assuming demographic closure for 45–70 days of sampling is not unreasonable. Nevertheless, surveys of the primary sampling period should be completed within the same season (e.g., surveys should not straddle a wet/dry season transition period) in order to minimise the risk of significant movement of elephants into or out of the surveyed area. Such movements can easily result in violation of the closure assumption and also lead to questions about exactly what abundance is of interest (if this changes during the survey period).

Sampling an area continuously over months and years certainly violates the closure assumption. It is incorrect to apply closed capture–recapture models to the data from these open populations, unless smaller periods are partitioned, during which closure might be reasonably assumed. Data can also be partitioned into discrete primary periods for use with open models. Although open models can be used to estimate abundance, associated estimators are not as robust as closed model estimators. This is particularly true for elephants whose social organisation pattern almost certainly results in heterogeneity of capture probabilities among individuals. While closed capture–recapture models can deal with such heterogeneity, open models cannot. Therefore, open models will yield estimates of elephant abundance that are certainly much better than simple counts of total individuals caught, but are less reliable than estimates based on closed models.

10.3 Conducting Photographic Capture–recapture Surveys

10.3.1 Choice of photographic method

Elephants can be photographically captured for subsequent analyses under traditional or spatially explicit capture–recapture models. In one photographic capture method, the investigators move around and/or wait at likely locations, and manually photograph individual elephants that they encounter. In camera trap photography, automated cameras are mounted at likely locations to capture elephants that trigger the sensor connected to the cameras. In this chapter, we have called the former 'observations' and latter type 'camera trap photography'.

The successful use of both these photographic capture methods relies fundamentally on a survey design that maximises probabilities of capturing elephants (Chapter 5). This means that surveyors should identify locations for photo-capturing elephants with the aim of maximising capture probabilities: such locations should not be randomly selected since elephants regularly use the same travel routes and do not move randomly through space. If locations for photographing or camera-trapping elephants are selected truly randomly, zero or near zero probability of 'catching' any elephants may be the result (refer to Chapter 5).

The selection of good observation or camera-trapping locations depends on careful reconnaissance of signs (e.g., tracks, dung piles, rubbing marks and feeding signs) indicating regular past use. The convergence of major elephant trails (sometines referred to as circles), salt-licks, valleys, *hadlus* or *gadde* (swampy areas with grass), entry and exit points around forest and village boundaries, resting sites (sometimes referred to as 'guest houses'), specific fruiting trees and water sources that attract elephants are all indicators of potentially good locations. Consulting local naturalists, hunters and other people familiar with the area is very useful in this context.

In the case of observers photographing elephants, animals that are accustomed to human presence will be relatively easier to 'capture' than those in areas of intense human–elephant conflict where they may be wary and display either flight-or-fight behaviour. Similarly, in open forests where elephants are more visible and less aggressive they are easier to photograph. Even in dense forests, identifying open areas frequented by elephants as mentioned earlier will improve the chances of getting usable photographs. Wherever vehicles are used to approach elephants, a relatively silent automobile will permit closer approach.

If automated camera traps are used, the immediate area around the trap should not be modified, and traps should be placed unobtrusively. Often, a simple trick like throwing a few branches around a trap can prevent it from being obvious. Masking tape of a dull colour should be used on metallic or brightly coloured parts. All loose cables should be hidden under leaf litter or soil. Taking these precautions is wise because elephants are known to be curious and inquisitive of changes and new objects/odours in their surroundings and this might lead to either 'trap happy' or 'trap shy' behaviour [Otis et al. 1978]. From our experience, the scent of humans and flashes from the camera may sometime provoke elephants to damage camera traps. Furthermore, with respect to camera flashes, studies have shown that photographs obtained of elephants during day are far more useful than photos taken at night due to limitations of flash photography [Varma et al. 2006].

In order to obtain photographs of both flanks, two camera traps should be positioned about 7–10 metres apart, on either side of the elephants' anticipated path. Even if the elephants walk along the edge of the path, usable pictures can still be obtained. For active infrared camera traps [O'Connell et al. 2011], the electronic beam should be set 100 cm above the ground to be tripped by adults as well as calves. Remember that the goal is to obtain photographs of the elephants' sides so that features such as head, ear, tusk and tail are clearly visible. Consequently, full, broadside pictures are the most useful.

10.3.2 Spacing and number of observations or camera trap locations

In any capture–recapture sampling design, the spacing of observations or camera trap locations is dictated by the biology of the species. Recall that when using traditional capture–recapture models, our goal is to have no 'holes' in which an animal present in the surveyed area has zero or near zero probability of being captured (Chapter 5).

Home ranges of elephants can be as small as 14 km^2 or >10,000 km^2 (see Sukumar 2003 for a review). There can also be very large variation among individual range sizes even within a population. So, in areas where some of the elephants have significantly smaller home ranges, ensuring that there are 3–5 observation or camera trap locations within the smallest home range size will ensure reasonable exposure probabilities for all individuals. If home ranges are of the order of 10–15 km^2 size, observations or camera trap locations can be about 2–4 km apart; while for areas where elephant home ranges cover several hundred square kilometres, the observation or camera trap locations can be 5–10 km apart or more (Figure 10.1); also see Karanth and Nichols (2002: 144–146). These distances are mentioned only as broad guidelines. Specific local knowledge about elephant movement

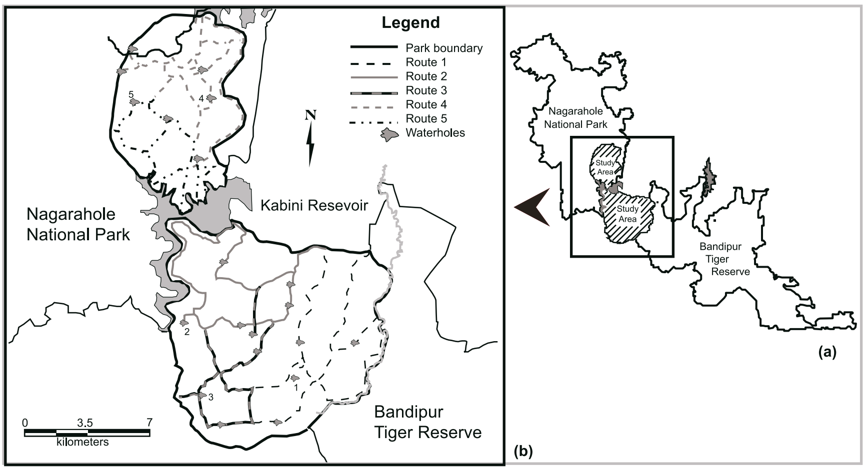

Figure 10.1 *Survey design for observer-based photographic capture–recapture of elephants involving drives along predetermined routes and 'waits' at selected water holes to 'capture' individual elephants. Map reproduced from Goswami et al. (2007).*

patterns and expected home range sizes are very helpful when deciding on locations for observations or camera traps.

If at least 3–5 observation or camera trap locations are within an individual elephant's home range, then it is more probable that it will be 'caught' in multiple sampling occasions. Thus it is generally better to have more such locations rather than fewer. On the other hand, however, the goal is to catch a large number of individuals in the overall study area of interest: the sample size in a capture–recapture survey is the number of individuals caught [Otis et al. 1978; White et al. 1982]. With a limited number of suitable observation locations or camera traps at our disposal, the investigators can potentially increase the number of individual elephants exposed to capture by increasing the space between trap locations to sample a larger area that holds more elephants. Consequently, trap spacing becomes a compromise between these two conflicting needs.

The survey area should ideally be reconnoitered first to locate and map all suitable observation or camera-trapping locations. Thereafter, the investigator can experiment on a map, using different combinations among these locations, to select locations that provide the best compromise among factors explained in Chapter 5. In this process, factors such as the availability of manpower, transport, equipment, terrain and the nature of any road or trail systems, will all influence how and where cameras can be deployed and checked. When deciding on observation or camera-trapping locations, other local factors such as possibility of equipment damage by animals, vandalism and theft also should be considered.

10.3.3 Data collection forms and protocols

Meticulously following field protocols is important to prevent all sorts of inaccuracies creeping into the identification of individual elephants and their photographic capture-history records. We make the following recommendations:

1. Camera traps have to be checked as frequently as possible, to verify proper working and to record any tripping that occurred previously.
2. A standard data form (Appendix 1) rather than field notebooks should be used to record observation or trapping data. Unique identification numbers should be given to each individual camera and each tripping unit for easy troubleshooting when these pieces of equipment malfunction. A unique ID and GPS location should be recorded for each of the observation points/locations.
3. Individual elephants must be identified correctly from photographs of both sides. If single-side photographs are used instead, approximately half the data will be lost as the inference will be based on left sides or

right sides only. Given all the other major expenses and efforts involved in camera trapping elephants, this approach is inefficient.

4. It is important to uniquely number each film roll (assuming film is used) before it is loaded. Such identification is critical for accurately fixing the location and the sampling period for each elephant capture event, and for matching left and right profiles of individuals for accurate identification. Because commercial laboratories mix up several rolls of film investigators must ensure that each film roll contains a photograph of a unique identifying label. If identities of rolls are mixed up during processing, capture history data can be seriously flawed. It will also be impossible to match left and right profiles of specific individuals. If the processed negative film is reversed while printing, the elephant identifications will also be erroneous. Even if elephant images are captured using digital cameras or video, similar care must be taken to ensure there is a foolproof system to correctly identify the location, date and time of each capture of any individual elephant.
5. When camera traps are checked, particular attention should be paid to battery status, number of remaining exposures, alignment of the sensor and the proper mounting of the camera. Other information, such as the date, time and other trap-settings must be set correctly. The date and time of each exposure as read by the camera's data-back should be carefully double-checked with those recorded by the sensor device.
6. All cameras and related equipment should be kept free of dust and moisture and maintained as recommended in the manufacturer's manual. Fresh batteries should be used in the cameras and tripping devices to ensure that data are not lost unnecessarily.

10.3.4 Identifying elephants from photographs

Following Goswami et al. (2007), we recommend that the following types of photographs be taken of elephants during 'observations': (1) frontal pictures with the head down, showing tusk and ear morphology; (2) profile pictures for both flanks to ascertain tusk angle (with respect to the ground) and tail length, and to identify scars, warts and other marks on the body; (3) clear side or frontal pictures of both ears; and (4) a close-up picture of the tail to identify the brush type (Plates 10.1 and 10.2). In addition, we also recommend the use of schematic diagrams using a template and a combinatorial key of the characters used for identification to complement the photographs [Goswami et al. 2007, Box 10.1]. Finally, each encounter with its associated set of photographs should be given a unique sighting ID and the geographic location of the sighting recorded using GPS. A similar procedure should be adopted for camera trap photographs.

Box 10.1 *A combinatorial key used to identify elephants photographed in Nagarahole and Bandipur National Parks in 2006 [Goswami et al. 2007].*

Character	Categories
Age class in years	< 15 / 15–20 / 20–25 / 25–30 / 30–35 / 35–40 / 40+
Shoulder height in feet	< 7 / 7–9 / 9+
Presence of tusks	Absent / Both / Right only / Left only
Tusk arrangement	Parallel / Convergent / Splayed
Tusk angle*	Straight ahead / Intermediate / Downward pointing
Tusk length in feet*	> 3 / 2–3 / 1–2 / < 1
Tusk thickness*	Thick / Normal / Slender
Ear fold*	Absent / L-shaped / U-shaped
Ear lobe shape*	L-angular / V-acute / U-rounded
Ear tear*	Yes / No
Ear hole*	Yes / No
Tail length	Below ankle / Below knee and above ankle / Below penis sheath and above knee / Stump (above penile sheath)
Brush type	Absent / Inside only / Outside only / Both-discontinuous / Both-continuous
Presence of tumors	Yes / No
Presence of scars	Yes / No

* Data to be recorded seprately for right and left ears and right and left tusks.

For identification of individual animals, variations in the tusks, ears and tail, as well as height and age classes (refer to Box 10.1 and Plates 10.1 and 10.2 for details), are most helpful. However, individual identification on the basis of all of these morphological features would necessitate comparisons across a fairly large set of physical attributes. For example, Goswami et al. (2007) used a combination of 16 morphological traits to distinguish individual tuskers. Given that elephants are often elusive, partially hidden and difficult to observe, recording such a large number of physical features poses practical challenges in the field. In addition, individual identification protocols can be somewhat descriptive and thus dependent on the field and communication skills of individual researchers. Goswami et al. (in prep) have evaluated an objective, automated approach to elephant identification and suggest that the overall set of morphological traits may be confined to characters (e.g., variations in tusk shape and size) unlikely to change during a typical survey. It is also worth noting that computer programs have been developed to assist individual identifications from photographs of a number of species, [Hiby and Lovell 1990; Kelly 2001; Karlsson et al. 2005; Hastings et al. 2008; Hiby et al. 2009]. In future we expect that similar programs could be adapted for use with elephants.

After all the photo-captured individual elephants are identified and given unique ID numbers, every capture of each individual elephant is assigned to a particular sampling occasion (Chapter 5) based on the date and time of the capture event. We strongly recommend maintaining a formal database of all capture records. From this database, the capture histories of each individual elephant can be placed into a capture history matrix format quite easily. In the standard '1 or 0' capture history matrix format, each animal detected at least once represents a row in the matrix, with each column representing a sampling occasion and containing either a 1 (indicating capture on that sampling occasion) or a 0 (indicating no capture). Table 10.1 shows an example of a capture history matrix constructed from observations in Nagarahole National Park and Bandipur Tiger Reserve (India) in 2006 [Goswami et al. 2007].

As described in Chapter 5, these capture histories are then used to estimate elephant population size either using conventional capture–recapture models (implemented in programs CAPTURE or MARK) or spatially-explicit capture–recapture models (implemented in programs DENSITY or SPACECAP). While a capture history matrix like that shown in Table 10.1 will suffice for analysis using programs CAPTURE or MARK, the spatial location of each individual capture event should also be included for spatially explicit analyses. An example file of capture histories suitable for spatially explicit analyses is included in Annex 10.1.

10.3.5 Choice of camera and camera trap equipment

A recent publication [O'Connell et al. 2011] provides details and evaluation of currently available camera trap equipment. We also provide some brief guidelines below.

Camera trap equipment has two components: the camera and a tripping device that fires it. Many ordinary electronic cameras can be hooked up to tripping devices to get usable pictures of elephants. However, such cameras must have auto-focus and auto-wind features and internal circuitry that permit shooting several pictures in succession using flash (if needed), without exhausting the battery. The camera should be able to withstand exposure to moisture, extreme temperatures and rough field use, and its

TABLE 10.1 *An example of a capture history matrix constructed from observations of elephants in Nagarahole National Park and Bandipur Tiger Reserve (India) in 2006 [Goswami et al. 2007]*

CAPTURE HISTORY MATRIX FOR INDIVIDUAL ELEPHANTS

Animal	**Secondary Sampling Occasion**										**Animal**	**Secondary Sampling Occasion**									
ID	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**ID**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
1	1	1	0	0	1	1	0	0	0	0	**40**	0	0	0	0	0	1	0	0	1	0
2	1	0	0	0	0	0	0	0	0	0	**41**	0	0	0	0	0	1	0	0	0	0
3	1	0	0	1	0	1	1	0	0	0	**42**	0	0	0	0	0	1	0	0	1	0
4	1	0	0	0	0	0	0	0	0	0	**43**	0	0	0	0	0	1	0	0	0	0
5	1	1	0	0	0	0	0	0	0	0	**44**	0	0	0	0	0	1	0	1	0	1
6	1	0	0	0	0	0	0	0	0	0	**45**	0	0	0	0	0	1	1	0	0	0
7	1	0	0	0	0	0	0	0	0	0	**46**	0	0	0	0	0	1	0	0	0	0
8	1	0	0	0	1	1	1	1	1	0	**47**	0	0	0	0	0	0	1	0	1	1
9	0	1	0	1	0	0	0	0	0	0	**48**	0	0	0	0	0	0	1	0	0	0
10	0	1	0	0	0	0	0	0	0	0	**49**	0	0	0	0	0	0	1	0	0	0
11	0	1	0	0	0	0	0	0	0	0	**50**	0	0	0	0	0	0	1	0	0	0
12	0	1	0	0	0	0	0	0	0	0	**51**	0	0	0	0	0	0	1	0	0	1
13	0	1	0	0	0	0	0	0	0	0	**52**	0	0	0	0	0	0	1	1	1	0
14	0	1	0	0	0	0	0	0	1	0	**53**	0	0	0	0	0	0	1	1	1	0
15	0	1	0	0	0	0	0	0	0	1	**54**	0	0	0	0	0	0	1	0	1	1
16	0	1	0	0	0	0	0	0	0	0	**55**	0	0	0	0	0	0	1	1	0	0
17	0	0	1	0	0	0	0	0	0	1	**56**	0	0	0	0	0	0	1	0	0	0
18	0	0	1	0	1	0	0	0	0	0	**57**	0	0	0	0	0	0	1	0	0	0
19	0	0	1	0	0	1	0	0	0	0	**58**	0	0	0	0	0	0	0	1	0	0
20	0	0	1	0	0	0	0	0	0	0	**59**	0	0	0	0	0	0	0	1	0	0
21	0	0	0	1	0	0	0	0	0	0	**60**	0	0	0	0	0	0	0	1	0	0
22	0	0	0	1	0	0	0	0	0	0	**61**	0	0	0	0	0	0	0	1	0	0
23	0	0	0	1	0	0	0	0	0	0	**62**	0	0	0	0	0	0	0	1	1	0
24	0	0	0	0	1	0	0	0	0	0	**63**	0	0	0	0	0	0	0	1	0	0
25	0	0	0	0	1	1	0	0	0	1	**64**	0	0	0	0	0	0	0	1	0	0
26	0	0	0	0	1	1	0	1	1	1	**65**	0	0	0	0	0	0	0	1	0	0
27	0	0	0	0	1	1	0	0	1	1	**66**	0	0	0	0	0	0	0	1	1	0
28	0	0	0	0	1	0	0	0	0	0	**67**	0	0	0	0	0	0	0	0	1	0
29	0	0	0	0	1	0	0	0	0	0	**68**	0	0	0	0	0	0	0	0	1	0
30	0	0	0	0	1	0	0	0	0	0	**69**	0	0	0	0	0	0	0	0	1	0
31	0	0	0	0	1	1	0	1	0	0	**70**	0	0	0	0	0	0	0	0	1	0
32	0	0	0	0	1	1	1	0	0	0	**71**	0	0	0	0	0	0	0	0	1	0
33	0	0	0	0	1	0	0	0	1	0	**72**	0	0	0	0	0	0	0	0	1	0
34	0	0	0	0	1	0	0	0	0	1	**73**	0	0	0	0	0	0	0	0	0	1
35	0	0	0	0	1	0	0	0	0	0	**74**	0	0	0	0	0	0	0	0	0	1
36	0	0	0	0	0	1	0	1	1	1	**75**	0	0	0	0	0	0	0	0	0	1
37	0	0	0	0	0	1	0	0	0	0	**76**	0	0	0	0	0	0	0	0	0	1
38	0	0	0	0	0	1	0	0	0	0	**77**	0	0	0	0	0	0	0	0	0	1
39	0	0	0	0	0	1	1	0	1	0	**78**	0	0	0	0	0	0	0	0	0	1

batteries should last for several days, even with the shutter cover kept open. To conserve the battery power, most modern electronic cameras go to 'sleep' if the shutter cover is left open. With such 'idiot-proof' cameras, the tripping device should be able to periodically 'wake the cameras up' electronically. Otherwise, the picture of the first animal that 'wakes up' a sleeping camera will be missed.

Several different types of tripping devices can be used in camera traps. In some units two metallic strips held apart inside a pressure pad come into contact with each other when an animal steps on the pressure pad, completing an electrical circuit that fires the camera. More commonly, electronic tripping devices that employ an infrared beam are used. There are two kinds of such infrared tripping devices. The 'active' type units have an infrared transmitter that emits a beam that is received by an infrared receiver placed opposite to it. When an animal walks between the two units, the beam is interrupted for a few seconds, completing a circuit that fires the camera. On the other hand, 'passive' infrared type devices 'sense' the heat emanating from the body of the animals passing in front of it and complete a circuit that fires the camera.

Appendix 4 lists some of the more commonly-used brands as well as review sites and other resources to help you select appropriate equipment. Additionally, the following factors must be considered when choosing equipment for a camera trap based capture–recapture survey aiming to estimate elephant density:

1. Currently, film cameras are often used in preference to digital cameras, due to faster shutter response, low power consumption, long life, lower prices, and better picture quality. However, recent advances in digital camera technology with faster shutters coupled with other obvious advantages such as higher memory and ability to directly down load images into a data storage unit make them increasingly attractive [Karanth et al. 2011, O'Connell et al. 2011].
2. Because two cameras are positioned opposite to each other in camera traps, a built in electronic time delay (of 30–40 milliseconds), which prevents opposite flashes from flaring the pictures is useful for photographs taken at night.
3. Generally, passive infrared units tend to be less prone to false tripping from, e.g., moisture, insects and vibration, compared to active type infrared units. In areas where rain is a major problem, this is a very important advantage of passive infrared units. It is generally easier to compose the picture and fix the elephant's anticipated position with active infrared units, because of the relatively narrow electronic beams that they generate.

4. The minimum time delay that can be set between two successive pictures tends to be much shorter (a few seconds) in active units compared to passive units. The practical implication of this difference is that, if two or more elephants are traveling together, only the first animal is likely to be photographed by the passive unit, whereas the active unit is likely to get others also. If cameras that go into 'sleep' mode are used as mentioned earlier, the tripping device must include the circuitry that 'wakes up' the cameras periodically to prevent potential loss of pictures of elephants. Some manufacturers enclose their units in moisture-resistant containers. In rainy or humid areas this is an important advantage. Sometimes, animals such as bears and elephants damage the units or humans may steal or vandalise them. In such situations, a rugged protective shell may need to be deployed to protect the cameras.

10.4 Fecal DNA based Capture–recapture Surveys

10.4.1 Design and implementation of fecal DNA based capture–recapture surveys: a step-by-step guide

10.4.1.1 Reconnaissance survey: evaluation of a site's potential suitability for fecal DNA based capture–recapture survey methods using fecal concentration surveys

As we have already noted, successful use of capture–recapture based population survey methods relies on a survey design that maximises capture probabilities (Chapter 5). In addition, successful identification of individual elephants from fecal DNA samples is aided by collecting fresh dung samples. This means that surveyors should search for elephant dung piles in places where they are likely to be found, not in randomly selected areas such as grid cells. The aim of the reconnaissance survey of fieldwork should therefore be to assess whether 'fresh' and 'reasonably fresh' dung piles can be found in adequate numbers if survey teams search in likely elephant 'hotspots'. Following Hedges and Lawson (2006), this approach is here called a fecal concentration survey. Once these data have been collected, it may be possible to immediately begin implementing a capture–recapture survey (Section 10.4.1.2). One should aim to conduct both the reconnaissance fecal concentration survey and the subsequent capture–recapture survey during the same time of year (or at least during the same season) in order to minimise the problems caused by seasonal elephant movements.

A 'fresh' dung pile is defined as one that is less than 48 hours old (Box 10.2). Fresh dung piles are identified by their appearance. They will be moist throughout, making them dense (heavy). They will usually feel slimy to the touch. Flies will often be present, and the dung pile should smell of elephant dung, not fungus or earth. Secondary evidence of fresh dung may also be provided by the presence of obvious recent elephant footprints and possibly damage to vegetation (e.g., plants pushed-over or trampled/eaten).) [For surveyors familiar with dung density based methods and the dung decay classification systems used for such surveys (see Chapter 9), it is important to emphasise that 'fresh' dung piles may not be intact; they can be in stage S2 (or even stage S3) when found.] A 'reasonably fresh' dung pile is defined as one that consists mostly of intact boli that are not obviously degraded (mouldy, infested with termites, etc.; Box 10.2). Ideally these 'reasonably fresh' dung piles should be no older than two weeks [Eggert et al. 2003], but we realise that assigning actual ages to dung piles will not be possible.

When planning a fecal concentration survey, local hunters, researchers and others familiar with the survey area should be asked to provide information about likely elephant 'hotspots' including salt licks, water holes, major elephant trails, roads, resting places and areas of frequent human–elephant conflict (e.g., crop depredations by elephants and points of entry into the village from forest). One should identify more potential hotspots than are actually required, as this will provide flexibility in the

Box 10.2 *Classification of elephant dung piles into three age classes*

FRESH
- Moist throughout, so they will be *heavy*.
- They will usually feel *slimy* to the touch.
- They will often be *shiny*.
- Flies will often be present.
- Smell of elephant dung, not fungus, or earth.
- Very fresh dung piles are usually a lighter brown colour than older ones.

REASONABLY FRESH
- Most (> 50%) of the boli (dung balls) are *intact, not broken*.
- Boli are *not obviously degraded*, which means:
  - No obvious fungus;
  - No ant or termite nests inside the boli.

OLD
- Dung piles that are not 'fresh' or 'reasonably fresh'.

subsequent survey-design phase. This data-gathering exercise should be followed by careful searches of the hotspots and any other likely places. During these searches, the number of 'fresh' and 'reasonably fresh' dung piles found at the hotspots should be recorded and their locations recorded using GPS equipment (for an example datasheet, see Appendix 1). Any other 'fresh' and 'reasonably fresh' dung piles found while traveling through the survey area (e.g., walking between hotspots) should also be recorded. The location of all fresh and reasonably fresh dung piles found during the fecal concentration survey should be entered into a GIS database to facilitate analysis and survey planning. A list of the equipment needed for a fecal concentration survey is provided in Appendix 2.

Once the fecal concentration survey is complete, the next step is to map the hotspots identified during the survey of the site. Then, once the hotspot map for the site is available, one should begin to devise a sampling design (by experimenting on the map) using different choices of hotspots (which will become sample collection locations) to optimise the spacing between the sample collection locations (this is why it is a good idea to identify more hotspots than are actually required). The configuration of the sample collection locations will vary from site to site depending on local conditions, but the basic principle is to cover the site so that there are no 'holes' in the network of sample collection locations where an elephant or elephants could move with a zero or near zero probability of their dung ever being collected. To put it another way, every elephant's home range should contain at least 2–3 sample collection locations (Chapter 5).

The distance between sample collection locations must be small if home ranges are expected to be small; if home ranges are large, the distance between sample collection locations can also be relatively large. As already mentioned above, home ranges of elephants can be as small as 14 km^2 or over 10,000 km^2, and there can be very large variation within a population (for a review, see Sukumar 2003).

As with the photographic capture–recapture methods described above, if home ranges are in the order of 10–15 km^2, DNA sample collection locations can be about 2–3 km apart, whereas for areas where home ranges are a few hundred square kilometres or larger, the sample collection locations can be 5–10 km apart.

If, after experimenting with the hotspot map, there are obvious 'holes' in the distribution of sampling effort over the site, one must take steps to remedy this problem. One approach is to divide those parts of the site without an adequate number of sample collection locations into 'blocks' that can be sampled in a single day. Generally, such blocks will be of equal size, but if the terrain, for example, is much more difficult in some places, then this should be taken into account when determining the size of the block. Note that it should be possible to begin implementation of the formal capture–recapture survey more or less immediately after the fecal concentration survey.

To return to the Thailand example mentioned above, a reconnaissance survey throughout Kaeng Krachan National Park (KKNP) identified 25 elephant hotspots (Figure 10.2a). Subsequently, the field teams conducted a formal fecal DNA-based capture–recapture survey. With four survey teams working simultaneously, each round of sampling took about two weeks to complete, with a two-week interval between each sampling round as recommended by Hedges and Lawson (2006). A total of 646 fecal DNA samples were collected over the entire survey period (297, 177 and 172, respectively, in the three rounds of sampling; Figure 10.2b).

10.4.1.2. Conducting the capture–recapture survey

Once the sampling locations have been selected, sample collection can begin. The field teams should return to all the hotspots that have been selected as sample collection locations and collect samples from as many fresh and reasonably fresh dung piles as possible, recording the GPS locations for all sampled dung piles (see Section 10.3.2 for the sample collection protocol; also see Box 10.2). If no fresh and reasonably fresh dung piles are found at the hotspots, the teams should search along animal trails, especially fresh elephant trails, in the area surrounding the hotspot (a broad guideline is to search as much of the area within a 2 km radius of the hotspot as possible). If time permits, it is advisable to search the area around the hotspot and collect additional samples even if fresh dung piles are sampled at the hotspot. A list of the equipment needed for a DNA based capture–recapture survey is provided in Appendix 2.

For those parts of the site without hotspots, the blocks discussed above (Section 10.4.1.1) will need to be searched. The field teams should spend one day in each block, following elephant trails and collecting as much fresh dung as possible. Trails can be used to help cover as much area as possible, but it is important to attempt to follow all fresh elephant trails encountered, and to collect samples from as many fresh dung piles as possible. It is also important to attempt to cover as much of the area in the block as possible, regardless of the density of the trails.

All dung piles from which samples are collected must be destroyed to avoid the possibility of re-sampling from the same dung piles when sample collection locations are revisited in the subsequent rounds of the survey.

One should aim to complete this ‘first round’ of sample collection (the first secondary sampling period within the primary sampling occasion) as quickly as possible. As a guide, ‘as quickly as possible’ means completing the sample collection in no more than 14 days. Once the first round of

Figure 10.2a *Location of elephant hotspots in KKNP (Thailand). Red dots indicate elephant sign recorded during reconnaissance surveys in the dry season and yellow dots represent elephant signs recorded during a similar survey in the wet season. The larger the dot, the higher the density of elephant sign encountered. The open circles represent the fecal sample collection spots selected to represent a network of both dung hotspots and the other (non-hotspot) sites necessary to ensure there were no 'holes' in the network where elephants had zero or near zero chance of being sampled (Manopawitr et al. 2008; see Section 10.3.1).*

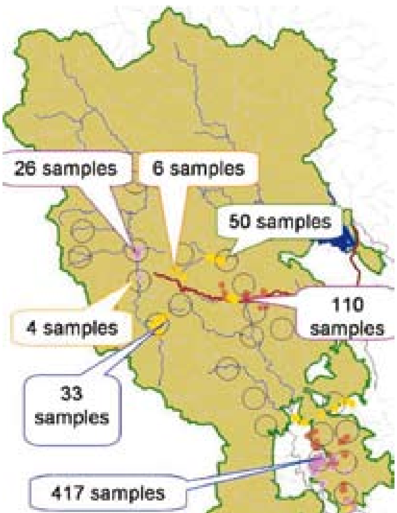

Figure 10.2b *Results of fecal DNA collection totaling 646 total samples from three secondary sampling periods. The different colours represent the four different survey teams' sample collection areas. The methods used followed those recommended in the Hedges and Lawson (2006) CITES/ MIKE* Dung Survey Standards *and described in Section 10.4 (Manopawitr et al. 2008).*

sample collection is complete, the subsequent rounds must be planned. At least two rounds of sample collection are recommended, but >2 rounds (ideally 5–10) are better as discussed above.

On revisiting the sample collection locations (and the blocks without hotspots), it is very important to collect from fresh dung piles that are believed with some confidence to have not been present during the first round of sampling. It is therefore best to wait for about two weeks before revisiting a sample collection location (hotspot) or block. (It is important to avoid waiting too long as the elephants may move out of the survey area.) Thus, three rounds of sample collection (three secondary sampling periods) would require about 2.5 months to complete. Given the long generation times of elephants, it is not unreasonable to assume demographic closure over such a period. However, all sample collection rounds should be completed within the same season to minimise the risk of large scale movements of elephants confounding the results.

10.4.2 Collecting and preserving fecal DNA samples

It is essential that the sample collection protocols and laboratory protocols provide DNA samples of high quality so that misidentification of genotypes is reduced. The collected fecal samples will need to be preserved in the field using a buffer solution; they may also need to be boiled to reduce the chance of pathogens being exported from the survey area with the samples (e.g., boiling samples is a requirement for importing fecal samples into some countries). Based on our field experience, we recommend the following protocol:

- Only collect samples from 'fresh' or 'reasonably fresh' dung piles (see Section 10.4.1.1 and Box 10.2).
- Record on the standard datasheet (Appendix 1) whether the dung pile was 'fresh' or 'reasonably fresh'.
- Wear latex gloves when collecting the samples (Plate 10.5). Do not allow your skin to touch the dung pile or the outside of your gloves when putting them on.
- Only collect from one bolus per dung pile (choosing the freshest one); this is to prevent errors caused by mistakenly thinking boli from two or more dung piles are from one pile, and thus possibly collecting fecal material from more than one elephant per sample.
- It is best to collect samples from the outside of the bolus if it is fresh (Plate 10.5), but from the underside if the sample is not very fresh. Use a plastic fork or a wooden stick to collect approximately 1/5 tube of dung (approximately 10g, usually one or two small 'forkfuls'). Place the dung in the tube, but do not pack it down.

- Do not use the same fork/stick for collecting other samples. Throw it away! (In an environmentally acceptable manner.)
- Mark the outside of the tube and the cap with the sample number, using a permanent marker.
- Each sample you collect should be given a unique code number.
- After collecting the fecal sample, measure the maximum circumference of three intact boli in the dung pile using a plastic measuring tape (Section 9.5 and Plates 9.11 and 9.12), and enter these data on a standard datasheet (Appendix 1). These measurements allow the age of the elephant producing the dung to be estimated (see Chapter 4). If there are more than three intact boli present, then the largest of the three should be measured. If only one or two intact boli are present in a dung pile it (they) should (both) be measured. Boli may need to be inspected carefully to make sure the correct axis is measured.
- All dung piles from which you collect samples must be destroyed to avoid the possibility of re-sampling from the same dung piles when you revisit sample collection locations in the subsequent rounds of the survey.
- For each sample, enter the sample number, the GPS location and the bolus circumference(s) on the datasheet along with any useful comments such as estimated group size and composition, presence of seeds, etc. Place the tube in a plastic Ziplock bag and write the sample number on the bag.
- Upon return to camp in the evening, boil the fecal samples (in their tubes) by placing the tubes in a pan of water for at least 15 minutes (Plate 10.6). (Loosen the tubes' lids but keep them on the tubes to prevent splash-contamination.) Then add approx. 10 ml. of Queen's College Buffer (just enough to cover the sample completely) and shake to make sure it is completely saturated. When pouring the buffer solution into the sample tube, make sure the bottle of buffer and the tube do not come into contact (Plate 10.7). Do not fill the tube completely—the sample will expand as it absorbs the liquid. Return the tube to the correct Ziplock bag.

  Note: Since it may be necessary to ask a Wildlife Department official or another official to sign a certificate stating that this boiling step was done (to allow import into some countries for laboratory analysis), it would be a good idea to show the relevant official how it is done so that they understand what they are signing.
- Protect the samples from sunlight as ultraviolet light may damage the DNA. This means storing the tubes in a dark-coloured plastic box.
- The samples can be kept at room temperature, but if refrigeration is available, it may extend the life of the sample. (Note that we have

used, successfully, buffered samples stored for more than two years in unrefrigerated conditions but we recommend refrigeration if possible and the commencement of laboratory analysis as soon as possible after the survey. We note too that sample storage time could usefully be included as a covariate in an error analysis.)

- Before shipping the samples to the laboratory (or before long-term storage), top-up the buffer if necessary, close the cap tightly and wrap the cap and top of the tube in Parafilm.

10.4.3 Data management for the fecal DNA collection teams

- Record the fecal sample data on the appropriate datasheet (Appendix 1).
- Datasheets should be placed in Ziplock bags to protect them from water.
- When the team returns to the base camp/office after each collection trip, the team leader must enter the data into the computer, make a back up copy of the data file and photocopy the datasheets.
- The original datasheets should be filed in the appropriate project base camp or office. The photocopies should be sent to your project officer. New datasheets should be used for every survey trip. Computer data files should be sent by email to the project leader if possible (this acts as an additional, off-site back up).

10.5 ANALYSIS OF CAPTURE–RECAPTURE DATA

Capture–recapture data obtained from carefully designed field surveys can yield reliable estimates of abundance (population size), density, vital rates and other demographic parameters in wild elephant populations. Details of the types of analyses possible, including model selection and fit and parameter estimation are presented in Chapter 5. In view of the relative novelty of the powerful spatially explicit capture–recapture models, we have presented a detailed example of such an analysis using a Bayesian approach implemented in program SPACECAP (Singh et al. 2010) in Annex 10.1. Additional information on analyses of capture–recapture data can be found in the standard works of reference by Williams et al. (2002), Amstrup et al. (2005), Thompson (2004), Royle and Dorazio (2008), O'Connell et al. (2011), and other citations in this manual.

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ANNEX 10.1

Using Program SPACECAP for Spatially Explicit Analysis of Elephant Capture History Data

Introduction

Program SPACECAP is a user-friendly software package for estimating animal densities using closed capture–recapture models. It is specifically developed from photographic capture data to implement a new generation of spatially explicit capture–recapture models developed. The statistical basis of the program is a recent paper titled 'Bayesian Inference in Camera Trapping Studies for a Class of Spatially Explicit Capture Models' by J. A Royle, K. U. Karanth, A. M. Gopalaswamy and N. S. Kumar, published in *Ecology,* 90(11), 2009, pp. 3233–3244.

Conventional capture–recapture models first estimate animal abundance from capture-histories and thereafter attempt to convert this abundance into animal density using a wide range of *ad hoc* methods (see reviews in Williams et al. 2002; Parmenter et al. 2003). Spatially explicit capture–recapture models implemented in SPACECAP directly estimate animal density by explicitly using additional information on spatial locations of captures under a unified Bayesian modeling framework. This approach offers several advantages: dealing individual heterogeneity in capture probabilities arising from relative locations of home ranges and trap sites; non-asymptotic inferences more appropriate for small samples and direct, easily interpreted probabilistic credible interval estimates of animal densities (Link and Barker 2010). Further technical details about the models and analyses are in Royle et al. (2009). We also point out that a different spatially explicit modeling approach based on conventional likelihood based inference program DENSITY (Efford 2004) based on statistical approaches outlined by Borchers and Efford (2008).

Installing program SPACECAP version 1.0

STEP 1: Download latest version of program R (R Development Core Team)
SPACECAP works within the R programming environment, so the very first step will be to connect to the internet, go to the website http://www.r-project.org, download and install the latest version of R (R 2.9.2 or higher) from the nearest CRAN mirror.

STEP 2: Download the package 'SPACECAP' to the computer.
After launching R, go to Packages→Install package(s), once again select the nearest CRAN mirror and select package SPACECAP for installation.

STEP 3: OPEN the package SPACECAP
In the R environment, go to Packages→Load package, select SPACECAP and load the package.

STEP 4: Launching SPACECAP
In the R environment, at the prompt '>', type the command SPACECAP().

This will launch the Graphic-User Interface of SPACECAP and the stage is set to begin the Bayesian Spatially-Explicit Capture–Recapture (SECR) analysis of capture–recapture survey data.

SECR analysis using SPACECAP

Running an SECR Analysis in SPACECAP essentially involves four simple steps:

1. Setting up the input files
2. Selecting the appropriate model combination
3. Selecting the Markov chain Monte Carlo (MCMC) settings
4. Hitting the 'RUN' button.

Step 1: Setting up the Input Files for Analysis

SPACECAP requires three input files. Store these files at a suitable location on the computer. These files are:

1. Elephant Capture Details File (Elephant ID no., Trap (Observation, DNA or camera trap based), Location no., Sampling Occasion no.)
2. Trap Deployment Details File (Trap Spatial Location, Deployment Activity, Sampling Occasion no.)

3. State-space Details File (describing the Potential Animal Home Range Center Details in terms of their spatial location and habitat suitability indicator for these home range centers)

These three raw data files can most easily be created using spreadsheet applications like Microsoft EXCEL, OpenOffice or other software that one is comfortable with. However, eventually, all files must be saved in an ASCII comma separated format (.csv), because SPACECAP can only read these types of input files.

Upon launching SPACECAP, notice on the input data panel, three separate buttons to load the three input data files. Pressing of these buttons will help locate the corresponding input files using the desktop browser (like Windows Explorer or Finder).

CREATING INPUT FILES

INPUT FILE 1: Elephant Capture Details

The input file containing individual elephant capture histories and locations consists of a 3-column table, each column representing the Location Number, the Elephant Identity Number and the Sampling Occasion number, strictly in that order. Please note that these are all 'number' fields. Do not enter labels containing alpha-numeric characters such as 'ELP-123', 'PLACE-100' or 'January 2011' etc. for these fields, because SPACECAP will not recognise them. Use simple integer numbers. Each unique individual captured during sampling should be given a unique identification number, ranging from 1 to *n*, where *n* is the total number of unique individuals caught during the capture–recapture survey.

Duration of the overall survey is determined by species biology in order to meet the assumption of demographic and geographic closures. For elephants, duration of 45 to 70 days is acceptable to meet the closure assumptions but care should be taken to complete the survey within the same season to minimise risk of large scale movements of elephants confounding the results. The duration of sampling occasions (or periods) will, in turn, be based on how many such occasions are there in the survey duration.

Because the underlying probability structure of the model is different, SPACECAP does not require capture history data presented as samples that cover the entire area of interest in each sweep. This greatly enhances flexibility in survey and analyses. Each sampling occasion must have a unique identity number, ranging from 1 to T, where T is the total number of sampling occasions. Note that each sampling occasion need not cover the entire survey area.

Each trap location must be given a unique identification number, ranging from 1 to J, where J is the total number of trap locations used in the survey.

Assuming a capture–recapture survey was conducted from 10 Jan 2011 to 30 Jan 2011 and treating each day as a sampling occasion, the survey consists of 20 sampling occasions. Assume there were 16 trap locations used in this survey and for logistical reasons the study area was partitioned into 4 blocks and each block contained 4 such trapping locations. Assume further that trapping was carried out for 5 successive days in each block covering each of the four blocks successively in 20 days. Assuming only 6 elephants were captured and identified; the INPUT FILE 1 for SPACECAP would look like:

LOC_Id	Elephant_Id	SO
1	4	17
5	5	13
7	6	17
8	6	16
9	3	20
11	1	14
15	2	12

The first data row tells us that elephant ID no 4 was captured at Location ID 1 on the 17^{th} sampling occasion. In spreadsheet application, one can build up the elephant capture details data as per the above format (please make sure to INCLUDE the header row with the exact titles on column headings as shown above). The file must eventually be saved as a comma delimited ASCII file with an appropriate file name and .csv extension (for example, 'captures.csv') and saved in the working directory.

INPUT FILE 2: Trap Deployment Details

In many capture – recapture surveys of elephants (as in the above example), all trap stations in the study area may not be operational simultaneously for logistical reasons (for example: limited number of cameras or manpower). Therefore, the trap deployment details input file provides SPACECAP with the information on the dates when each trap location was active and operational during the survey. Some trap-nights or trap-days of capture data may be 'lost' as a result of trap failure, theft, vandalism or animal-damage. This type of trap activity/passivity information can also be effectively fed into and used in SPACECAP. The trap deployment details file records both these types of information, thus accurately accounting for trapping effort.

The trap deployment data are stored in a two dimensional matrix of trap locations and sampling occasions in a binary, 0/1 format, where 0 indicates that a particular trap station was NOT operational on a particular sampling occasion, and 1 indicates that it was operational. The trap location is denoted in 3 columns in the table representing the Trap Location ID no., the spatial location expressed in X and Y-coordinates (in Universal Transverse Mercator UTM projection system in GIS). It is important that these coordinates are represented in the UTM projection system, because it is used for all distance measurements and computations in SPACECAP.

The trap deployment data entry is illustrated using a table for the same example of **elephant capture details** described above. Please recall that the trapping survey was conducted with a trap array of 16 trap locations, deployed in 4 blocks with 4 trap sites in each block. The trapping survey was conducted over 20 sampling occasions, during which each trapping block was sampled over 5 sampling occasions. The resulting TRAP DEPLOYMENT DATA file would look like the one shown on the next page:

The table shows that the camera trap sites (Loc 1–4) were operational during sampling occasions 1–5 but were not operational on the remaining sampling occasions 6–20. Additionally, an odd ‘0’ corresponding to Loc 3 and sampling occasion 3 indicates that a trap night/day was ‘lost’ here.

The trap deployment details file should be constructed exactly as above in the spreadsheet application. It must then be converted to a comma separated ASCII file (.csv file) with an appropriate filename (e.g. traps .csv) in the working directory accessed by SPACECAP.

INPUT FILE 3: Potential Home-Range Centers

In SPACECAP analyses, the surveyed area containing the trap array combined with an extended area surrounding it, known as the ‘state-space’ of the underlying point process, say S, which is represented by a large number of equally spaced points in the form of a very fine mesh. These points are visualised as representing all possible potential activity centers (or home range centers) of all the elephants in the population being surveyed. This fine grid or mesh of points can be easily generated using a GIS software (like ArcView, MAPINFO, etc.) as briefly described below. View this as an approximation to an underlying continuous state-space which, in practice, would normally be difficult to characterize for computational purposes except in very basic situations where regular polygons might be reasonable. While estimates of population size, N, will be sensitive to the size and extent of the state-space, the estimated density $D=N/||S||$ is invariant as the extent of the state-space increases. Thus, S should be chosen to be sufficiently large so as to ensure stability of the density estimate. Conceptually, this occurs (under the models fitted in SPACECAP) by choosing S to buffer the trap array by 2 or 3 times the encounter probability scale parameter.

LOC_ID	X_Coord	Y_Coord	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
1	619303	1325966	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	624151	1325013	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3	624722	1323864	1	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
4	621806	1322453	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
5	622451	1320137	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0
6	622599	1317937	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0
7	623179	1315941	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0
8	625156	1315587	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0	0	0	0	0	0
9	626022	1314224	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0
10	627568	1315494	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0
11	619604	1324739	0	0	0	0	0	0	0	0	0	0	1	1	0	1	1	0	0	0	0	0
12	621478	1324515	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0
13	623317	1323989	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1
14	624406	1321603	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1
15	624482	1320577	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1
16	629229	1319793	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	1	1

First a rectangle is formed by connecting the outermost camera trap locations using the GIS software of choice. This rectangle is called the 'minimum area rectangle'. A buffer distance (which is sufficiently large to ensure that no individual elephant outside of the buffered region has any probability of being captured by the traps in the array during the survey) is added to the rectangle around encompassing the trap array. Thereafter, using GIS numerous equally spaced points representing home range centers are generated for this extended area.

In practice, some of these potential home range centers in the mesh may end up in habitats known to be entirely unsuitable for the study species (say in the middle of a village, for elephant data). SPACECAP appropriately deals with this problem because 'Grid Cells' input file clearly specifies which of these potential home range/activity centers lie within suitable habitat and which do not.

The potential home-range centers data file essentially consists of a 3 column table. The first two columns are the X and Y coordinates (both in UTM projection system) of all the potential activity centers (the equally spaced points generated from GIS software) and the third column is a habitat suitability indicator column, indicated with 1s if the potential activity centers lies within suitable species habitat or with 0s otherwise. An example of the format of this file is as given below:

Example of potential home-range centers input file format:

X_COORD	Y_COORD	HABITAT
611734.0	1299581	1
612301.6	1299583	1
612869.2	1299585	0
613436.9	1299587	0
614004.5	1299589	0
614572.2	1299591	0
615139.8	1299593	0
615707.5	1299595	0
616275.1	1299597	1
616842.8	1299600	0
617410.4	1299602	1
617978.1	1299604	1
618545.7	1299606	1

Such a table must be created for all the potential activity centers (this will be a very large table) and saved as a comma delimited ASCII file (.csv file) with an appropriate name, for example, centers.csv, and saved in the working directory.

In the input data panel, there is a text box for specifying the area of potential home-range centers. This area may be imagined to be composed of a point at the centre of a square, which we call 'pixel'. Please enter the pixel size area used by the GIS software in this box. The 'fineness of the mesh' determined by the pixel size used for spacing of potential home range centers is dictated by species biology (e.g., a few hundred meters for elephants). Caution should be exercised here to not specify a state-space that is too fine because the MCMC algorithm run time increases linearly with the size of the state-space grid. Regarding this as a discrete approximation to some underlying continuous state-space probably justifies a reasonably coarse state-space grid. Regardless of the desired state-space dimension, carrying out a trial run with a coarse grid to evaluate the performance is recommended.

Load these files using on screen 'buttons' provided in the input data panel of SPACECAP. Specify the pixel size area of a potential home-range center (as created in input file 3 above) in square kilometers and then click on 'OK'. Please check the frame at the bottom for status or error messages. To edit selections, click on the 'Edit' button and start the selection all over again.

Proceed to the Model Selection frame.

Step 2: Selecting the Appropriate Model Combination for Analysis

The Model Selection panel of SPACECAP consists of a set of options to select an appropriate model combination for the Spatial-Capture Recapture Analysis. These are simple radio buttons indicating each model choice. Some of these model options that are 'grayed out' are expected to be made available in future developments of SPACECAP.

The model choices are:

1. Trap response present OR Trap response absent

   Selecting 'Trap response present ' option runs the behavioral response option (equivalent to Model 'M_b'). Select the 'Trap response absent' option if otherwise. This model implements a local or 'trap-specific' behavioral response under which the probability of encounter in a trap increases subsequent to initial capture in that trap. This is in contrast to the conventional 'global' behavioral response which parameterizes a constant increase in encounter probability (on the logit scale, usually) that is not trap specific.

2. Spatial Capture–Recapture OR Non-spatial Capture–Recapture
   Select 'Spatial Capture–Recapture' for running a spatially explicit capture–recapture analysis, or 'Non-spatial Capture–Recapture' for running a conventional capture–recapture analysis (this is equivalent to the Null Model 'Mo' in non-spatial capture–recapture analysis)
3. Half Normal OR Negative Exponential
   Currently SPACECAP analyses SECR models with only the Half-Normal detection function.
4. Bernoulli (binary) OR Poisson encounter process
   Currently the analysis is run with the Bernoulli encounter model in which the probability of success is derived as the probability of a positive response under a Poisson encounter rate model. This motivates use of the complementary log-log link which relates encounter probability to distance and other covariates. After the model selection is complete, please click on 'OK'. Please check the frame at the bottom for status or error messages. To edit selections, please click on the 'Edit' button, change the Model Selection and click on 'OK' again. Proceed to the Model update frame.

Step 3: Setting the Markov-chain Monte Carlo (MCMC) Parameters (for Advanced Users)

SPACECAP uses the Markov-Chain Monte Carlo simulation algorithm written in Program R (R Development Core Team) to estimate the parameters of the Spatially-explicit Capture Recapture models of Royle et al. 2009). The relevant settings can be set in the model update panel of SPACECAP.

No. of iterations This defines the number of MCMC iterations for the analysis (if not sure what this means, please set this to a value of about 50,000).

Burn-in This defines the number of initial values to discard during the MCMC analysis setting this at about 1000 usually works well in our experience. Note that some evaluation of whether this is sufficient should be carried out using conventional methods (a topic we will address in subsequent releases).

Thinning This defines the thinning rate. Only iteration numbers defined by the thinning rate are stored during the analysis (if not sure of what this means, set this at a value of 1 – that is, no thinning).

Data augmentation Since we are uncertain about the total number of elephants, which is likely to be larger than the minimum number caught during the trapping survey, 'augment' this value by a certain amount. A very large number relative to the number caught would be ideal, but setting it up to be very high will cause the analysis to run for a very long time. As a rule of thumb, set this to a value of about 5–10 times the number of animals captured during the survey. Data augmentation (DA) is a computational device that enables a convenient Bayesian analysis of capture–recapture models where N is unknown. In the context of SECR models, N is the population of individuals having their activity centers on the prescribed state-space. The basic idea of data augmentation is to provide an upper bound on N, say M, which is equal to N plus the number of augmented individuals. Technically, M is the upper limit of a uniform (0, M) prior for N, which is a customary 'noninformative' prior for N in this context. As a practical matter, data augmentation creates a list of pseudo-individuals that are always available for the MCMC algorithm to 'use' if necessary. That is, these pseudo-individuals leave and enter the population depending on the current values of the model parameters. See Royle et al. (2007) for some general context and Royle et al. (2009) for details in the context of spatial capture–recapture models.

After the model update values have been specified, please click on 'OK'. Please check the frame at the bottom for status or error messages. To edit these settings, please click on the 'Edit' button, edit these values and click on 'OK' again. This sets the stage to start the analysis.

Step 4: Running the Analysis

The last step will simply involve activating the RUN option in the top menu bar. This will start performing the analysis and a progress bar indicating the status of the analysis can be seen. Currently, an analysis involving 50,000 iterations takes about 14 hours on a fast computer.

Results

The posterior density estimates along with standard errors appear as a table in the output panel upon the completion of the analysis. This table also reports estimates of parameters lam0, sigma, psi and beta. If the analysis was run with trap response present, the estimates of 'p1' and 'p2' are also reported.

Additionally, all the results are written into a comma separated file (called param_val_<timestamp>.csv) and is saved into the current working directory. All the summary statistics are written into a file called summary_stats_<timestamp>.csv, which is also saved to the current directory. And the posterior density graphs of all parameters are all stored as jpeg files (.jpg) in the current working directory.

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CHAPTER 11

Estimating Distribution and Abundances of Elephant Populations from Sign Surveys at the Landscape Scale Using Occupancy Modelling: Field Methods

K. Ullas Karanth, N. Samba Kumar and M. S. Nishant

11.1 Introduction

Conservation planning requires basic information on species' spatial distribution at regional and at large landscape levels. This is particularly true for wide-ranging species like elephants with home range sizes that can vary widely from 14 km^2 to >10,000 km^2 (see Sukumar 2003 for a review). Moreover, despite their size, elephants can be difficult animals to see in dense vegetation and encounters are too infrequent when they occur at low densities: these considerations typically preclude the use of sighting based survey methods to assess distribution (see Chapter 2). Consequently, occupancy-based sign surveys are often preferred for assessing distribution and habitat use. Fortunately, occupancy methods and sign surveys can be used to identify potential habitat corridors, landscape connectivity, dispersal routes and threats faced by elephants (see Chapters 6 and 12). Furthermore, sign survey based occupancy modelling has the potential to link occupancy data with more intensively measured abundance estimates at local scales

[Royle and Nichols 2003; Conroy et al. 2008; Chapter 6]. In other words, the combination of occupancy methods and sign surveys can now also be used to answer questions such as: What are the distributional ranges of different individual elephant populations over a large region? How are ranges of individual elephant populations expanding, contracting or fragmenting? What are the ecological and management factors influencing habitat occupancy patterns of elephant populations? Thus, if properly conducted, sign-based occupancy surveys can provide much useful information for management of elephant populations, even in the face of severe logistical and resource constraints that impede or prevent more intensive studies using other methods (Chapter 2).

In this chapter, we consider field practices and protocols that can be employed to implement the occupancy modelling concepts we discussed in Chapter 6. Nevertheless, we recognise that these field protocols might require some appropriate modifications to suit local conditions.

11.2 Identification and Recording of Elephant Signs

11.2.1 Recognising elephant signs

Elephants leave conspicuous signs such as dung piles, footprints, rub marks and feeding signs, which are easily detectable by trained observers. Given the herd-based social organisation of elephants and their high defecation rates [Sukumar 2003; Section 4.5], elephant dung in particular is an easy sign to detect and identify because of its size and abundance and its characteristic odour and shape. On the other hand, although occurring at higher frequency, elephant footprints can be surprisingly difficult to detect on hard substrates or on leaf litter. In addition, elephants' dung piles can be aged and categorized as fresh, reasonably fresh or old with relative ease, compared to other elephant signs. It is also possible to assess the age-class of individual elephants from the size of their dung boli (Chapters 4 and 9).

11.2.2 Recording of elephant signs

In occupancy surveys of the type discussed in this chapter, dung piles of elephants are recorded (and sometimes footprints too, but see cautionary remarks above). It is important to note, however, that in order to meet demographic closure assumptions of occupancy models (see Chapter 6), only data on 'fresh' or 'reasonably fresh' dung piles should be collected, excluding older dung piles that may have remained intact for many months [for definitions of 'fresh' and 'reasonable fresh' see Hedges and Lawson (2006) and Chapter 10, especially Box 10.2].

Recording the signs of elephant calves encountered during field surveys will help identify habitat patches in the landscape that harbour breeding individuals. When surveys are repeated over time, increased encounters with signs of calves in different cells could be used to help assess the population dynamics of elephants across the entire landscape. The new multi-state occupancy models (e.g., Nichols et al. 2007) provide an appropriate approach to address such questions. Therefore, if possible survey teams should specifically note the presence of dung piles containing boli with circumferences ≤ 30 cm (see Chapter 9 for further detail).

In addition, recording covariates such as signs of human impacts (e.g., hunting and wood cutting) and habitat variables during occupancy-based sign surveys is also helpful as discussed in Chapter 6 and below.

11.3 Resource and Logistic Considerations for Conducting Sign-based Occupancy Surveys

As for all survey and monitoring methods, an adequate number of personnel should be available to survey the area being assessed (i.e., a 'landscape'). Remember, however, that the aim is not to count every individual elephant or even every elephant sign present in the surveyed area. Rather, the goal is to estimate the proportion of habitat patches or sites occupied by elephants. Therefore, the surveys need not be conducted in all patches or sites simultaneously, as long as overall closure assumptions are met as discussed in Chapter 6. This logistical flexibility helps investigators deploy survey resources optimally.

Skilled survey personnel who can find and identify elephant signs reliably and accurately, and who can also record, map and geo-reference the data collected are essential to a surveys success. Typical elephant sign-based occupancy survey teams may have 3–4 observers, with at least one 'expert elephant tracker', and others who possess the necessary skills to use maps, GPS and to record data.

The surveys can be conducted over shorter periods by employing a large number of government staff, local naturalists or civil society volunteers, if they are available. If sufficient people are not available, smaller teams can sequentially cover the cells (habitat patches, sites) over a longer period, as long as due regard is paid to the constraints of the demographic closure assumption (Chapter 6).

Finally, to the extent possible, sign-based occupancy surveys should be conducted during seasons when sign detection conditions are similar among sites to minimise the variations in detection probabilities.

11.4 SURVEY DESIGN CONSIDERATIONS

As already mentioned in Chapter 6, the size of the occupancy survey's grid cells (habitat patches, sites) should be based on prior knowledge of elephant ecology. For example, in some parts of Asia it might be reasonable to assume the average maximum home range size for an adult male elephant to be < 200 km^2 or < 400 km^2 [see Sukumar (2003) and Fernando et al. (2008) for reviews]. Also, data on movement patterns of Asian Elephants indicate typical daily movement rates of about 1–40 km [Sukumar 2003]. Thus, in the examples above, it would be appropriate to select cell sizes of 200 km^2 and 400 km^2, respectively. This approach to survey design aims at assessing 'habitat occupancy' rather than 'intensity of habitat use' [see Chapter 6 and MacKenzie and Royle (2005)]. In practice, cell sizes can be modified marginally so that cell boundaries can be made to coincide with geographic coordinates such as latitude and longitude or UTM markings on the maps being used by the field teams. This makes it easier for field teams to follow the survey design provided faithfully and record data without errors.

Once the appropriate cell size has been selected, the leaders of the survey should delimit the survey area on a base map (ideally in a GIS) showing altitudinal contours, vegetation types, human settlements, land use and other relevant details. This base map should then be used to create a map of grid cells overlaid on the entire area of interest (the 'sampling frame'). Each grid cell (say of size 200 km^2) is then further divided into 16 smaller sub-cells of equal size (Figure 11.1). The primary purpose of these sub-cells is to ensure that one of them is randomly picked ahead of time so that the field survey team's route can be directed to pass through that sub-cell, while leaving a lot of logistical flexibility on the best routes and manner of sampling the larger cells based on ecological considerations and local expert knowledge. Figure 11.1 gives an example of an occupancy survey with nine grid cells (all with 100% elephant habitat), each with its own randomly selected sub-cell and a hypothesised sequence of field survey effort, with three days being spent in each grid cell to search for elephant sign, and with the sequential order of grid cells to be surveyed being based on logistics and convenience.

If resources and logistic considerations permit, we recommend a 100% spatial sampling of all cells (sites) within the pre-defined sampling frame, to increase sample sizes for reliable occupancy modelling (Chapter 6). Such an approach enables mapping elephant occupancy even if suitable covariate data that permit extrapolation of occupancy estimates to un-surveyed areas are not available. Furthermore, this approach offers some advantages for monitoring changes in occupancy across the years more reliably. However, such 100% sampling may not always be feasible. In such cases, a proper

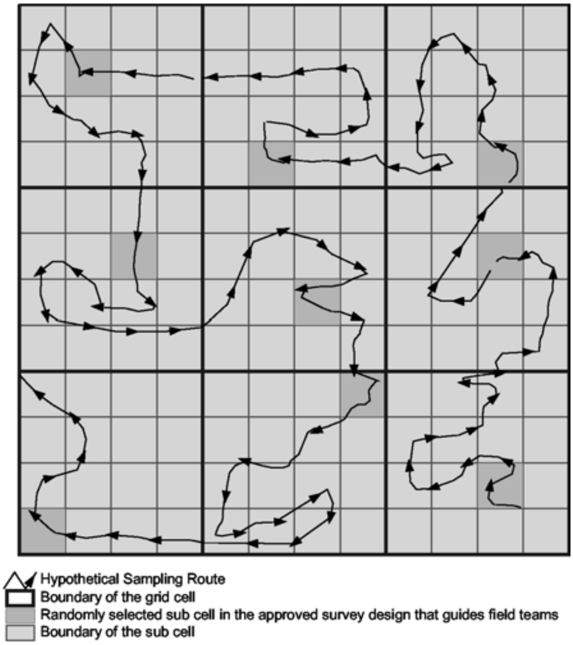

Figure 11.1 *A hypothetical survey frame showing 9 cells each with 16 sub-cells, one of which is selected randomly, and which must be visited by the field team.*

spatial sampling scheme has to be chosen to select cells to be surveyed [MacKenzie et al. 2006, Chapter 6].

Based on our experience with Asian Elephants, all grid cells containing more than 10% elephant habitat (kinds of vegetation where elephants can live) should be included within the survey sampling frame, if cell sizes are 200–300 km^2 in size. The logic here is that elephants cannot 'live' in patches smaller than these (20–30 km^2), although they may occasionally pass through or take refuge in them for a short period. However, this decision, which defines the sampling frame, should be based on local knowledge of elephant ecology as well as the cell sizes used in the specific survey.

Once the sampling frame has been finalized it is necessary to decide on the number of spatial replicates per cell. From experience with sign-based occupancy surveys of elephants in India (Karanth et al. unpublished data), we have found that a 'spatial replicate' (Kendall and White 2009; Karanth et al. 2011; Chapter 6) can be a trail segment of about 1–2 km length. The issue of potential lack of independence among sign detection events on such replicates has been adequately addressed by the development of a new occupancy model [Hines et al. 2010], which has also been successfully demonstrated in the field in a Tiger survey (Karanth et al. 2011).

We suggest that the length of the spatial replicate for an elephant survey be set constant at 1 km, with the number of replicates in a cell being proportional to the effort (total distance to be walked), which in turn is proportional to the extent of potential elephant habitat available in the cell. In the above cited example [Karanth et al. 2011], 40 km of walk/search effort was invested in a cell with 100% tiger habitat and the effort for cells with smaller amounts of tiger habitat was in scaled in proportion to the extent of habitat available in each cell. Thus, the minimum number of 1-km long replicates per grid cell was 4 for a 200 km^2 cell with 10% habitat cover and 40 for a cell with 100% habitat cover. The field survey teams should record detections of elephant sign (or non-detections) for every 100 meter segment along each 1-km replicate, although during the analysis these data may get aggregated as detection histories for each 1-km long replicate (see Karanth et al. 2011). This approach will provide additional flexibility in exercising analytical options based on site-specific considerations subsequently (e.g., in covariate modelling).

11.5 FIELD SURVEY (DATA COLLECTION) PROTOCOLS

Field survey teams should make pre-planned visits to cells and search for elephant sign following a carefully worked out survey design [see above and Chapter 6; also see Hines et al. (2010) and Karanth et al. (2011)]. As described above, the field survey teams should be directed to pass through the randomly located sub-cell in each cell. The field teams should then search potential elephant habitat within each larger cell, guided by local knowledge. The survey of a cell may take 1–7 days depending on the topography, terrain, vegetation density, presence of natural barriers and logistical constraints such as availability of manpower. Each cell should be surveyed in a pre-determined, convenient sequence so that all the cells are covered within the entire occupancy survey period.

When searching for elephant sign, the teams should search along elephant travel routes and visit places likely to be visited by elephants such as *bais* (forest clearings with water outlets), saltlicks, waterholes and the like, in order to increase their chance of encountering elephant signs. The teams should not follow random compass bearings or any such other 'seat of the pants statistical design'. Each trail segment should be surveyed only once, and 'return journeys' made along it should be excluded for data collection purposes.

We note that, sometimes, after completing the search of a cell, substantial distances may have to be walked within that same cell to reach the randomly chosen sub-cell in the next cell to be surveyed. Similarly, long distances may have to be walked to reach overnight camping points. We recommend that all such walking effort expended within 'defined elephant habitat inside each grid cell' should not be considered 'wasted effort' or 'down time', as far as possible. With careful planning, even these distances walked can be combined to meet the required survey effort for each cell, under the agreed survey design. Thus, all elephant sign data should be collected and recorded meticulously, including the survey effort, within all elephant habitat traversed. The only time elephant sign should not be recorded is when the team ends up completely out of elephant habitat, for example while moving to the camp or to the next cell. Only such 'real wasted effort' should be excluded from the survey sampling effort.

11.6 Protocol for Recording Covariate Data

As already mentioned, tremendous value can be added to a survey if appropriate covariate data are collected during a sign-based occupancy survey. Thus, in addition to data on elephant sign, data on pre-defined covariates should be recorded for every 100 m segment (see the datasheet in Appendix 1). As an example, we provide, in Box 11.1, typical covariate data that were gathered during the occupancy surveys for Tigers which we conducted recently [Karanth et al. 2011]. It is important to note here that covariate data need to be collected for every search path segment in every cell irrespective of whether elephant signs were encountered in the segments.

11.7 Collection of Fecal DNA Samples

In some sign-based occupancy surveys a survey goal might also be to collect dung samples in order to extract fecal DNA. Fecal DNA based genetic analysis facilitates the identification of individual elephants, which permits application of advanced modelling to derive elephant abundance estimates and other population dynamic parameters under appropriate survey designs (see Chapters 5 and 10). However, to ensure the dung samples are useful, it is necessary to follow carefully standardised protocols (for example to avoid cross-contamination of DNA, which may render the analyses futile). To this end we direct the reader to the fecal DNA collection and preservation protocol described in Chapter 10 (see Section 10.4.2).

11.8 Conduct of Additional Questionnaire Surveys

If very large regions are to be surveyed in a short period without investing tremendous amounts of effort in the field, occupancy surveys based on 'expert opinion' can be considered. In such a case, each reliable informant or expert consulted becomes a 'replicate sample'. See Karanth et al. (2009, BOX 11.1 *Example of covariate data that can be collected during a sign-based occupancy survey (from Karanth et al. 2011)*

Segment Type ROD = Road, TRL = Trail

Segment type covariate is to indicate the width of the 100 m sample segment being surveyed. Road means a forest road which is wide enough for a four wheel vehicle to pass. Road may have either two visible or no tracks in it. Trail means a narrow forest or animal trail used either by humans or other animals.

Substrate Condition SOF=Soft soil, HAR=Hard soil, LLT=Leaf litter, GCR = Grass cover

Substrate condition covariate is to indicate the prominent substrate condition for every 100 m sample segment. For example, if you are walking on a road, the substrate condition for the tracks on the road should be recorded and not for the centre of the road which is usually covered with grass. The dominant substrate condition of the sample segment should also be recorded. For example, if the substrate condition for the tracks is soft for more than 50% of the 100 m segment and the rest is hard, then it should be recorded as soft (SOF). Similarly if the tracks on the road or trail are covered with leaf litter for more than 50% of the 100 m segment then they should be recorded as LLT.

Habitat Type MDF = Mixed Deciduous Forest, EVG = Evergreen Forest, GRS = Grassland, OTH = Other

Habitat type covariate is to indicate the prominent habitat found in the 100 m segment surveyed. MDF indicates mixed deciduous forest (both moist and dry deciduous forest types are included in this category), EVG indicates evergreen forest, GRS indicates grassland and OTH indicates any other type of habitat that might be found (usually plantations). The prominent habitat type for the 100 m segment walked should be recorded. For example, if more than 50% of the 100 m segment was deciduous and less than 50% was evergreen, then the habitat type should be recorded as MDF for this 100 m segment. The habitat type categories have to be decided *a priori* based on information available on the forest vegetation types prevalent in the overall study area.

2010) for good examples of occupancy modelling of large mammals, including elephants, based on such expert opinion surveys. However, these shortcut methods are not substitutes for conducting field surveys at finer scales as described earlier in this chapter and elsewhere in this manual.

Questionnaire surveys gain importance if large regions cannot be surveyed using field teams. Opinions of reliable local hunters, naturalists, researchers and wildlife personnel can be useful, but soliciting unreliable informants—via random sampling or the like—will entirely ruin the survey. We note that quite often such knowledgeable informants may not be literate or educated, and may be even suspicious about the surveyors' motives, and so special approaches may be required to elicit reliable information from them. Questionnaire surveys should be conducted only by trained personnel capable of assessing the quality of information. If the informants are ignorant or untruthful, the data should be discarded ruthlessly. If neither informants nor survey personnel of adequate calibre are available, there is no point in carrying out questionnaire surveys.

Each individual informant must be treated as a distinct sample unit (replicate) and his or her information must be recorded on a fresh questionnaire survey form. As in the case of field surveys of sign, these forms must be systematically numbered and cross-linked to geo-referenced maps. A specimen questionnaire datasheet is given in Appendix 1. Please note that not all the information needs shown on the specimen datasheet are related to the spatial mapping of elephants and so the form may be modified to suit local needs.

11.9 Organising Field Data for Mapping and Analysis

The data from field surveys or questionnaires will be in the form of several maps and 'elephant detection' data on field forms linked to these maps. The investigator should examine these forms to correct errors, remove ambiguities, discard questionable data, and fill in missing information by interviewing the field survey team members. This must be done immediately after the surveys. It is often a good idea for the investigators to physically check a certain proportion of the data through random field visits as this will enhance data quality. Thereafter, the investigator must ensure data forms and maps are intelligible to persons entering the data into computers, preparing spatial distribution maps or performing other analyses.

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CHAPTER 12

Assessing Threats and Monitoring Law Enforcement

Emma J Stokes

12.1 Introduction

Elephants continue to be killed illegally in Africa and Asia to satisfy a demand for ivory. While the debate on the linkages between CITES decisions on elephant listings and poaching levels remains highly polarised [Stiles 2004; Wasser et al. 2010], there is a broad consensus on two important issues: firstly, that elephant poaching is typically associated with poor governance and weak law enforcement capacity within elephant range states [Stiles 2004; CITES 2010a]; and secondly, that better data on poaching of elephants is needed in order to reliably inform elephant conservation and policy decisions at the site, national and international level [Blake and Hedges 2004; CITES 2010a].

The importance of law enforcement and monitoring in the conservation of elephants both inside and outside protected areas has been clearly documented [Leader-Williams 1990; Jachmann 1997; Hilborn et al. 2006; Blake et al. 2007; Stokes et al. 2010]. Effective law enforcement requires information on where, how, and by whom illegal activities related to elephant killing are undertaken, and the ability to apply this knowledge strategically to reduce poaching—often in the context of limited human and financial means. To achieve this, a site-based mechanism is required that can capture current field data and make the information available in such a way as to be easily understood by protected area staff, wildlife managers and other stakeholders as appropriate. Furthermore, effective law enforcement and conservation policy requires a transparent monitoring system with which to evaluate the progress and performance of law enforcement agencies and personnel in reducing threats to elephants and other wildlife. This requires appropriate threat indicators to be selected and standardised protocols for data collection and analysis to be adopted. This chapter presents guidelines for establishing site-based law enforcement monitoring (LEM) programmes that address both of these requirements.

These guidelines are aimed at site-managers and practitioners implementing patrol-based law enforcement monitoring in protected areas and other areas of elephant habitat with a dedicated law enforcement presence (e.g., wildlife protection units in timber or mineral concessions). An overview of key concepts, data collection standards and recommended management tools and resources is provided. The practical and technical challenges involved in the design and implementation of LEM programmes is highlighted with recommendations for avoiding common pitfalls. In addition, guidance on collecting and interpreting data on threats through third-party informant networks is also given. The limitations of LEM in quantitative assessment of poaching and other threats to elephants are discussed and avenues for improving the interpretation of LEM data are proposed. In preparing this chapter, I draw on experiences gained from the design and implementation of LEM in protected areas with elephants under the *Tigers Forever*[1] programme in Southeast Asia [Stokes 2010] and acknowledge parallel efforts to develop LEM in 'priority elephant sites' through the CITES Monitoring the Illegal Killing of Elephants (MIKE) programme [e.g., CITES 2010a]. However, these guidelines can also be applied to a broad range of species across different ecological and conservation contexts. This chapter is not intended to function as a set of exhaustive step-by-step instructions but, rather, to act as a set of guiding principles. By following the principles outlined here, site managers can go a long way towards improving their understanding of how law enforcement strategies influence both the assessment and mitigation of threats and improve management effectiveness accordingly. Finally, it should be stressed that law enforcement monitoring is a tool for improving law enforcement effectiveness; it can provide managers with the information they need to make strategic decisions but it requires that the appropriate institutional,

[1] A *Panthera* project, in collaboration with the *Wildlife Conservation Society* and local government and non-governmental partners, to recover wild tiger populations at source sites across their geographical range.

legal and judicial support structures and resources are in place, enforcement staff are active and other law enforcement interventions are taking place on the ground. The concepts and methods described in this chapter should therefore be considered as part of an overall investment in and commitment to improving law enforcement effectiveness on the part of elephant range States.

12.2 Patrol-Based Law Enforcement Monitoring

Patrol-based law enforcement monitoring (hereafter patrol-based LEM) is the opportunistic collection of data on illegal or suspected illegal activities (and other data types including observations of flagship or threatened species) by rangers on wildlife protection patrols [e.g., Gray and Kalpers 2005]. Patrol-based data collection has the benefit of being relatively inexpensive (i.e., there are few additional costs), relying as it does on existing patrol efforts and personnel and requiring fewer specialised skills and less equipment than dedicated survey teams. If patrols are conducted regularly over the entire protected area or site, LEM is a cost-effective strategy for providing rapid, yet standardised information on the intensity and distribution of threats over time that can help site managers make informed decisions regarding patrol deployment and allocation of resources as part of an adaptive management approach. Furthermore, given regular feedback to enforcement managers and team leaders, LEM has the potential to motivate rangers and build capacity in their day-to-day patrolling activities. Given that many protected areas in elephant range States are understaffed, underfunded or simply not patrolled at all [Leader-Williams and Albon 1988; Blake and Hedges 2004; CITES 2010a], LEM, as part of an overall investment in and strengthening of law enforcement capacity, can thus greatly improve both enforcement efficiency and management effectiveness in combating poaching.

Patrol-based LEM can be used as a tool for measuring trends in illegal activities. Ideally, patrols are done systematically over the entire protected area or site but in practice resource and budget limitations necessitate a focus on only those parts of the protected area where the most serious threats are known to be occurring, or which are considered the highest priority for conservation. This introduces considerable bias. In addition, patrols often habitually follow existing human or animal trails which can either increase or decrease detectability of illegal activity and wildlife signs [Walsh and White 1999]. In contrast, formal survey methods employ design-unbiased methods, such as line transects [e.g., Thomas et al. 2009; Chapter 3]. Patrol data quality may also vary considerably and be of generally lower quality than data collected for example by specially trained survey teams. Data quality can vary both as a result of individual ranger capacity and education as well as the simple fact that data collection is often compromised during enforcement activities, for example during the active pursuit of poachers. As a result, care needs to be exercised in interpreting the results from patrol-based LEM, and there exist a number of caveats to the use of patrol-based LEM for quantitative analysis of trends, which this chapter aims to address. LEM should not therefore be considered as an alternative to periodic, independent and systematic assessments of threats (or elephant populations) in a site (see Section 12.5.2).

12.2.1 Law enforcement effort

An important feature of patrol-based law enforcement monitoring is the relationship between law enforcement effort and the frequency of illegal activities encountered on patrols. The relationship between patrol effort and observations of illegal activity is typically expressed as catch per unit effort (CPUE), analogous to measures used in commercial fisheries to assess the status of fish stocks [e.g., Maunder and Punt 2004]. CPUE is used as an index of relative abundance for a particular illegal activity or threat indicator, for example, number of carcasses per unit of distance patrolled, or number of poachers arrested per unit patrol time [e.g., Leader-Williams 1990].

Quantification of law-enforcement effort is thus required to adjust for variable effort in measuring incidence of illegal activity over time or comparing frequency of illegal activities between sites. Depending on the type of patrolling[2], the mode of transport, terrain and the permeability of habitats to patrols, law enforcement effort can be expressed in a variety of ways, from extremely simple measures to those corrected for unit time, unit area, size of patrol group and other relevant variables. In general, measures should be kept as simple as possible, with the distance patrolled (e.g., for reconnaissance patrols), the time patrolled (e.g., number of hours or days) and patrol coverage (or area contained within the radius of the patrol) being three of the most critical measures of effort. Measurements of law enforcement effort are also used to assess staff performance, employing measures such as operational budget and patrol staff density [Hilborn et al. 2006; Jachmann 2008b].

[2] Patrol type includes reconnaissance patrols where new information on illegal activity is being sought over a wide area; intelligence or surveillance patrols where specific existing information on illegal activity is used to focus the patrol effort across a set of target locations; and roadblocks or other stationary patrols (e.g., at entrance gates or guard posts).

12.2.2 Selecting indicators for illegal activities and other threats (the 'catch')

Absolute levels of illegal activity, such as elephant poaching, can rarely be measured due largely to their illicit nature which means they are difficult to detect *in situ*. Proxy indicators of illegal activities therefore need to be selected and these should be measurable and sensitive to changes in the level of threat. For example, hunting camps might be a suitable indicator for poaching if hunters typically travel long distances and spend several days on hunting trips, but would fail to account for short or day-long hunting trips where camps were not constructed, and thus underestimate level of hunting pressure; also, hunting camps may not represent a measure of threat to elephants if other species such as deer or antelope are the targets.

As a general rule of thumb, indicators for LEM should be defined according to site management objectives. In reality, these will range from quite general information on human activities, to specific indicators of poaching rates of elephants and other key species. Furthermore, the suite of indicators selected will vary considerably between sites. To permit a level of standardisation of threat monitoring across different sites (for example, either within a national network of protected areas, or range-wide for a particular species) while recognising flexibility in local conditions at the site level, a two-tiered approach using both broad-scale indicators of human impacts and site-specific indicators for particular threats is recommended.

12.2.2.1 Broad-scale human impacts

Development of standardised indicators on human activities that impact elephants and other wildlife can be guided in part by existing classification systems, such as the unified classification of direct threats developed by IUCN and the Conservation Measures Partnership[3] [Salafsky et al. 2008]. This classification, which represents the integration of several independent global classification schemes, is a hierarchical listing of terms and associated definitions. The classifications are comprehensive and exclusive at the upper levels of the hierarchy, expandable at the lower levels, and simple, consistent, and scalable at all levels. Under this scheme, threats to the protected area (or site) as a whole are addressed, including, but not limited to, threats that affect a particular species. This is important in encouraging buy-in to LEM from site managers and enabling a cost-effective and harmonised approach that satisfies multiple management goals, rather than imposing multiple additive species-specific approaches. Furthermore, use of a single classification enables standardised threat measures within and between sites, and is applicable across a wide range of different ecosystems

[3] http://www.conservationmeasures.org/

(e.g., forest and savannah), and types of monitoring (for example, recording observation of human activities by independent biological monitoring teams), beyond just LEM. The system is also scalable and highly adaptable to different local contexts [see also Table 12.2 for examples of the unified classification scheme as applied to protected areas with elephants].

12.2.2.2 Site-specific indicators for particular threats to elephants and other key species

A series of site- and species-specific indicators can also be developed, for example for the poaching of elephants, that are considered to best reflect the levels of poaching pressure [Stokes 2010]. For elephants, carcasses of elephants illegally killed for their ivory (often defined by the presence or absence of tusks; Douglas-Hamilton and Burrill 1991), and elephant poachers encountered on patrol [Hilborn et al. 2006], have typically been used as an indicator of *in situ* poaching pressure or rates of poaching. Carcasses within forest habitats however are often difficult to detect on foot compared to savanna systems where carcasses can be more easily detected during standard aerial surveys [MIKE 2004]. More recently, the proportion of illegally killed elephants (PIKE), expressed as a ratio of all elephant carcasses encountered has been used to determine trends in poaching pressure [CITES 2010a]. This method circumvents the need for corresponding measures of patrol effort (data on which were inconsistently collected) as it is assumed that effort would be included in both the numerator and denominator and thus cancel one another out. An implicit assumption of PIKE is that the detection probability for all carcasses (whether illegally killed or not) is the same. This is unlikely to be the case. For example intelligence-led patrols may preferentially lead law enforcement teams to illegally killed carcasses, which would bias PIKE to overestimating rates of poaching [CITES 2010b]. Other indirect signs of elephant poaching, such as the presence of large hunting camps with meat drying racks and ammunition have also been used as indicators of poaching pressure, particularly in forest environments where carcasses are hard to detect [Blake et al. 2007]. However, the relationship between these indicators and the actual levels of poaching are difficult to quantify.

12.2.3 Assumptions of Catch Per Unit Effort (CPUE)

The interpretation of simple CPUE indices for monitoring levels of illegal activity within a particular site relies on the following assumptions:

1. Patrol records are reliable accounts,
2. Relationship between law enforcement effort and catch is constant, and
3. CPUE is proportional to true abundance of threat.

Guidelines for avoiding some common pitfalls that frequently violate these assumptions are provided below.

12.2.3.1 Patrol records are reliable accounts

Patrol data should be of sufficient quality and consistency, which requires the appropriate skills in sign identification and accurate data recording. Skill level and motivation of rangers can in turn affect overall search effort and willingness to record data in the field. These factors can be standardised to some extent by training, proper management structures, adequate compensation packages and an environment that encourages competition and communication (e.g., regular feedback of information). Data collection protocols also need to be developed in such a way as to remove any ambiguity in the field and to avoid overburdening rangers with extensive data forms and complex procedures, while still ensuring that key information is collected in a standardised manner. Field rangers often experience relatively high staff turnover and encompass highly variable levels of skill and experience and this needs to be recognised in designing LEM protocols. Patrol based LEM protocols also require that observations of illegal activities are recorded *in situ* (i.e., not second-hand or from third-party reports, see Section 12.4) and are recorded only once, thus a system needs to be put in place to ensure that signs of illegal activities (e.g., snares, camps, etc.) are either removed, destroyed, or marked in some way to avoid duplicating records.

12.2.3.2 Relationship between law enforcement effort and catch is constant

To assume that the relationship between law enforcement effort and catch is constant implies that all occurrences of a particular illegal activity indicator (for example, carcasses or poachers) have an equal chance of being detected by patrols. In reality, there are many examples of situations in which this relationship does not hold.

For example, patrol deployment typically varies across space and time, due to variation in accessibility for patrol teams, changes in availability of funds for patrolling, as well as in response to changes in the nature and spatial distribution of threats. Thus, at a minimum, interpretation of CPUE indicators for monitoring trends needs to control for both spatial and temporal variation in patrol effort. Care should be taken in avoiding extrapolation of results over areas without any patrol effort at all, or where patrol effort is low or highly variable—for example, zones that are visited rarely or irregularly such as periodic sweeps to remove land encroachers—as these zones provide very little useful quantitative data for monitoring trends, although may still provide useful anecdotal information for managers on the presence of a particular threat. It is recommended to monitor trends in CPUE indicators in only those areas that are patrolled *regularly* and to divide these areas into *sectors* that have a *relatively constant and even coverage of patrol effort* from month to month. At smaller sites, all sectors might receive regular patrols and be suitable for monitoring CPUE, while at larger sites, only a proportion of the total number of sectors receive regular visits by patrol teams.

Detectability can also vary between different illegal activities. Some illegal activities are more predictable in space and time and therefore easier to detect by patrol teams. For example, loss of elephant habitat resulting from human encroachment on the edge of protected areas is often easier to detect than poaching, which is much less predictable and harder to detect by enforcement teams. Different categories of illegal activity should therefore be treated separately in CPUE-based analyses.

Different types of patrol (e.g., reconnaissance, intelligence-led) and means of transport (e.g., foot, vehicle) can greatly influence the probability of detecting illegal activities if present. For example, foot patrols are much more likely to locate carcasses or snares hidden in the forest, than patrols in a vehicle. Vehicle patrols on the other hand may be more likely to apprehend transporters of illegally logged wood or other trafficked products such as ivory. Also, patrols that are based on specific intelligence are more likely to result in a 'catch' than routine reconnaissance patrols. This is of interest to managers looking to maximise return from their investments and improve the efficiency of enforcement efforts [Jachmann 2008b] yet is rarely documented in a systematic and standardised manner. Data collection procedures should therefore *record patrol type and means of transport*. Furthermore, different patrol types may require different measures of patrol effort that best characterise the time spent actively searching for signs of illegal behavior (e.g., distance patrolled for reconnaissance patrols compared with number of investigative days for intelligence-led patrols). These measures are not easily combined using simple CPUE indicators. It is therefore recommended to distinguish between different patrol types and transport in the analyses, presentation and interpretation of CPUE indicators. We note that it may also be worth experimenting with multivariate composite CPUE measures that incorporate a number of different metrices.

12.2.3.3 CPUE is proportional to true abundance of threat

Perhaps the most important assumption of using CPUE indicators to monitor threat levels is that the CPUE indicator is directly proportional to the actual or true level of threat. In other words, if we measure an increase or decrease in the number of poachers caught, we assume that this represents an actual increase or decrease respectively in the level of poaching pressure. In practice, non-linear relationships between CPUE and the true level of illegal activity are likely to occur in patrol data. In the commercial fisheries literature and in studies of bushmeat offtake in hunting systems [Milner-Gulland and Rowcliffe 2007], non-linear relationships between CPUE and abundance have been grouped into two classes: hyperstability and hyperdepletion (Figure 12.1). Hyperstability describes situations where the CPUE remains high while the size of the sampled population (or true level of threat) declines. Hyperdepletion is the opposite situation, where CPUE drops off more rapidly than the size of the sampled population (or true level of threat). For patrol data, these relationships can, in some instances, be describing true changes in the nature of illegal activity. However, non-linear relationships can also be produced by situations that are independent of true changes in the level and nature of illegal activity. Table 12.1 provides some examples of situations that might result in hyperdepletion or hyperstability, but which are unrelated to changes in the level of illegal activity being measured. This presents challenges for interpreting trends in CPUE of illegal activities from patrol data. It can potentially lead to a situation where data give the impression that a threat is decreasing, when in reality it is stable, or worse, actually increasing.

Some of the potential situations described in Table 12.1 can be addressed through better law enforcement planning and adoption of clear data collection protocols and appropriate analytical methods. In other instances, simple CPUE indicators are insufficient and appropriate modelling techniques need to be applied to adequately describe and characterise the relationship between catch and effort (see Section 12.5.1). Regardless, it is recommended strongly that independent (i.e., non-patrol based) measures of illegal activities are obtained periodically, in order to verify and calibrate the results from patrol-based LEM, and to identify if any of the situations described in Table 12.1 may be operating (see Section 12.5.2).

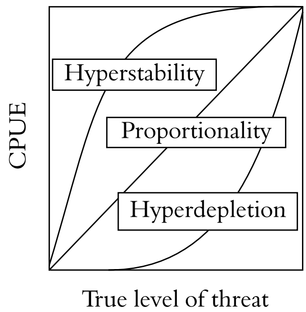


Figure 12.1 *Possible relationships between CPUE and abundance (adapted from Milner-Gulland and Rowcliffe 2007).*

TABLE 12.1 *Summary of independent factors resulting in non-linear relationships between CPUE and level of illegal activity (adapted from Milner-Gulland and Rowcliffe 2007)*

Relationship between CPUE and abundance	*Type of cause*	*Specific example*
Hyperstability (CPUE overestimates the actual level of threat)	Inappropriate analysis	Aggregating data over a wide area (or timescale) to include zones (or time periods) with low enforcement effort, and high levels of threat
Hyperdepletion (CPUE underestimates the actual level of threat)	Law enforcement strategy	Switch in focus of strategy to target a particular activity at the expense of other illegal activities
	Law enforcement strategy	Lack of motivation to detect and/or record signs of illegal activity (may or may not be linked to suspension of bonus/incentives—see below)
	Law enforcement strategy	When incentives/bonuses are provided for enforcement staff, satiation may occur once a target has been reached and teams switch focus to other illegal activities.
	Inappropriate analysis	Aggregating data over a wide area (or timescale) to include zones (or time periods) where law enforcement effort is high (and levels of threat low)
	Poacher (or other violator) strategy	Poachers/violators evade capture by field enforcement personnel, either by avoiding areas that are used predictably by law enforcement teams or changing technique to one that is less detectable

12.3 MIST: An Information Management Tool for Patrol-Based LEM

MIST is an integrated spatial Management Information SysTem (MIST), initiated in 1997 through a collaborative project between GTZ (Deutsche Gesellschaft für Technische Zusammenarbeit) and the Uganda Wildlife Authority (UWA) for implementation across Uganda's protected area network. Ecological Software Solutions LLC (ESS) was contracted by GTZ in 2001 to develop the current MIST software programme. MIST was custom-built to meet the law enforcement monitoring needs of protected area managers by collating standardised data on measures of law enforcement effort, observations of illegal activities, and patrol actions, and converting these into useful information for management planning [Schmitt and Sallee 2002]. Because it was designed using a bottom-up approach, it focuses on the key information and output needs of managers and addresses the technical and practical challenges of data transfer and data management with limited on-site resources and capacity.

MIST is currently maintained and distributed free of charge by ESS for non-commercial use (http://www.ecostats.com/software/mist)*. It is implemented in Delphi with ESS Shape Viewer Objects to obtain GIS functionality and is available as a standalone package (with an optional client/server database application programme) and associated data collection procedures. Both the data collection procedures and the software application were developed in such a way that they can be tailored to reflect differences in issues, objectives, and threats at local level and in different protected areas or even land-use categories throughout a country. One of the greatest strengths of MIST is the capacity to provide a platform on which to apply a standardised approach to the collection, management, evaluation and communication of patrol-based law enforcement monitoring data, through a user-friendly interface that bypasses the need for complex database skills and GIS software packages.

MIST is currently employed by protected area and wildlife agencies in sites across Africa and Asia in a variety of conservation and ecological contexts [e.g., Makombo and Schmitt 2003; Stokes 2010]. MIST is also now replacing the database and reporting protocols of CITES-MIKE for monitoring elephant carcasses [CITES 2010a]. As well as improving management effectiveness, the MIST approach has succeeded in fostering multi-agency collaboration in law enforcement efforts and in harnessing a general interest by government and other agencies in adopting a

* This chapter refers to MIST Version 2.3.4.5 which was the latest version at the time of going to press.

standardised and transparent approach to the monitoring and evaluation of law enforcement efforts. In this section, I provide practical guidance on setting up and implementing MIST, but these guidelines are broadly applicable to any local-level information management tool designed to help improve law enforcement effectiveness.

Before starting MIST implementation at a site there are a number of preparatory and planning steps that should be considered, and which are outlined in Box 1. While MIST itself is designed to improve the efficiency and effectiveness of patrolling, it does require at a minimum that rangers are both available and able to carry out patrols at the site. MIST should therefore only be attempted at sites once there is existing infrastructure and staff on the ground and where law enforcement is supported by the relevant government agency or partner NGOs.

An initial investment of time and effort into setting up the system at a site will help to ensure the LEM programme is sustainable, efficient and fully supported by the relevant stakeholders. For managers wishing to implement MIST in multiple protected areas or sites, it is recommended to first select one or two pilot sites in which to test the system and process, before refining the process and replicating it on a larger scale.

BOX 1 *Summary of steps for setting up MIST-based LEM at a site*

1. Conduct risk assessment
2. Define resources, personnel and training needs
3. Define LEM objectives and monitoring indicators
4. Create MIST data structure for patrol observations
5. Define patrol staff, stations, type of law enforcement activities and other patrol effort attributes
6. Develop MIST data sheets and data collection protocols for patrol teams
7. Determine reporting needs and map outputs
8. Spatially delineate management sectors for LEM reporting and compile other key GIS data layers
9. Create pilot MIST database
10. Field test data forms and observation structure
11. Modify MIST database and data forms accordingly

12.3.1 Personnel and other resource needs

Defining the personnel and capacity needed at the site-level for MIST implementation is a key step that should be accomplished early in the process. One of the benefits of patrol-based LEM is that it is typically not necessary to recruit additional personnel for data collection as existing law enforcement rangers will fulfill this role. In reality, many protected areas across the elephant's range in Africa and Asia do not have this basic fundamental requirement in place [CITES 2010a], and LEM should therefore be seen as an additional incentive for increased investment in, and strengthening of law enforcement capacity in such sites. Additional training of patrol teams in MIST data collection procedures however will be required, and it is recommended to include this as a specific module in standard law enforcement training and refresher courses, which is now being done for example in several MIKE sites in Southeast Asia [Lynam and Lawson 2004].

There are two additional roles that need to be filled for the implementation of MIST at a site. These are commonly referred to as the MIST User and, in certain situations, the MIST Database Manager. These are not necessarily full-time roles but do have differing requirements in terms of location, skills and responsibilities. In general, it is recommended to train at least two staff members in MIST procedures to ensure sustainability. Basic terms of reference for these two positions are as follows:

MIST User The MIST User is the person responsible for MIST data entry at the site and monthly reporting. This is an important role and appropriate training in MIST data entry and reporting is required. The MIST User needs to be able to use a GPS and computer, speak and write basic English[4], and ideally be based at the site; they do not need to have any specialised database or GIS training, but they must understand basic computer functions (Microsoft Windows or other computer operating system). The MIST User also needs to regularly interact with the patrol teams to ensure that data collection forms are filled in correctly, and is required to submit monthly MIST reports (or information as requested) to the site manager.

MIST Database Manager and/or Coordinator The Database Manager is responsible for managing and maintaining the MIST database. This role is particularly important if MIST is to be rolled out in more than one protected area and a central coordination database is to be established. This individual would have a more advanced level of training in computer technology, and would typically be based in the national or provincial capital where electricity supply is more reliable. This individual would also be responsible for the advanced features of MIST such as customising the reporting templates and editing the data structure using MIST Administrator tools. The Database Manager would communicate regularly with the site-based

4 English is currently the default language of the MIST software, although a partial French translation is now available (http://www.pamis.org/trac/mist/wiki/ChangeLocale). While it is possible to customise data collection forms and reports and translate then into local languages, the current programming platform will support only languages based on the Latin alphabet (e.g., Bahasa Indonesia, French and Portuguese).

MIST Users, conduct regular quality-control checks on the MIST data, and assist in training/refresher courses for MIST Users and for rangers in data-collection protocols. The MIST Database Manager would also be responsible for troubleshooting and reporting any software glitches to the MIST developer (Ecological Software Solutions LLC). A moderated MIST community email list-server has now been launched by ESS LCC (mist@lists.pamis.org) which provides an additional support structure for technical ideas and questions about MIST.

Equipment and other resources MIST is a spatial management information system and requires spatial data to be regularly collected by rangers. GPS units, or some form of GPS data logger that can record both the GPS location, time and date therefore need to be available to each patrol team. For patrol distances to be measured with a reasonable degree of accuracy in MIST it is recommended, as a minimum, for rangers to take position waypoints (not tracklogs) every 30 minutes when on patrol. GPS battery requirements should therefore be factored into budgets accordingly.

The MIST User needs to have access to a single computer on-site, on which is installed the MIST database and which is used for data entry, GPS download, and preparation of MIST reports. No other software packages are required. A constant power supply for the computer is not necessary at the site. Data can be block-entered once a month or entered on a continual basis depending upon the set-up.

Internet connection at the site is not required to operate MIST. A site should have some mechanism of sharing MIST reports with the site manager and senior patrol staff on a regular basis. This can be done either by printing hard copies of MIST reports or through projecting MIST reports and maps on a screen during monthly patrol meetings. A site should also have some mechanism for backing up MIST data and, in the absence of e-mail, for providing (and receiving) MIST updates from the MIST Database Manager. Periodic updates for the software will also need to be obtained, and these are typically posted via email by the developer on the MIST list-server. These can be managed and downloaded through occasional connection to internet (e.g., in the local town) or through file transfer via USB flash drive from the MIST Database Manager.

12.3.2 Data collection procedures

Data requirements for patrol-based LEM need to be focussed on providing the necessary information for management without overwhelming rangers with complicated data collection protocols at the expense of the task at hand–law enforcement. MIST works on the following principles for data collection:

- *Standardised* Data needs to be recorded in a consistent and unambiguous way
- *Simple* The data collection system must be easy to use with a minimal amount of formal education and fully localised into regional languages and cultures
- *Fast* Time spent recording data by rangers must be kept to a minimum, and data collection should not compromise enforcement activities
- *Flexible* It must be possible to adapt data collection methods to meet the needs of different users and in different contexts of natural resource management, reflecting differences in objectives and threats in different protected areas or wildlife management zones
- *Specific* Only data which can be processed into information useful for management decision-making should be collected by rangers

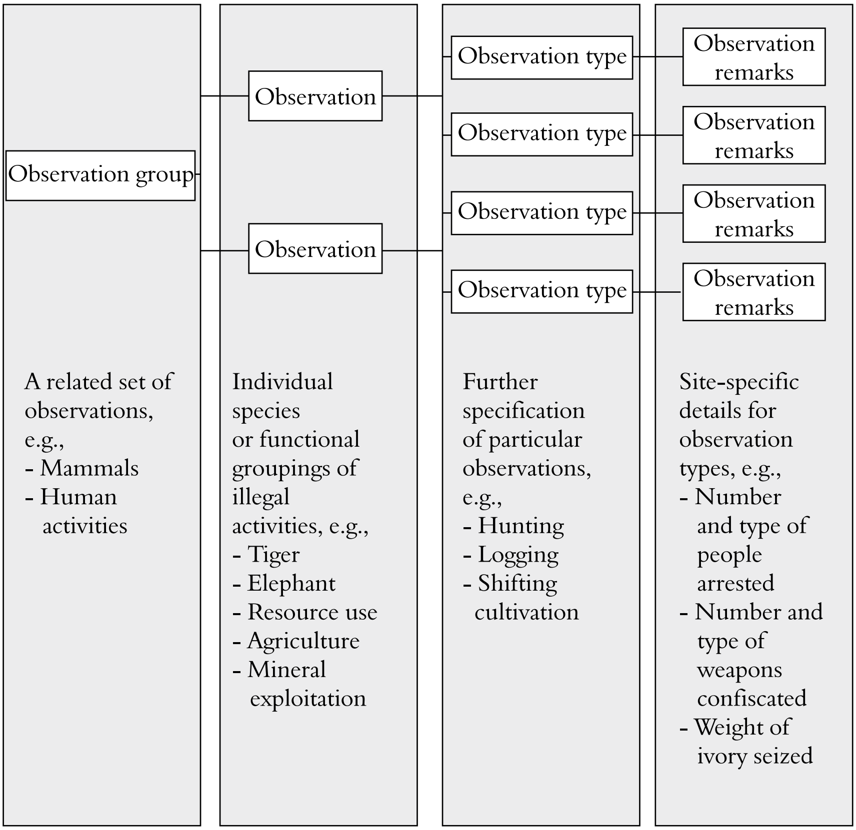
Standard data inputs for MIST include the following:
- GPS waypoints (coordinates, dates and time of observations, and patrol routes)
- Information about the patrol (e.g., patrol dates, names and numbers of rangers, type of patrol and means of transport)
- Patrol observations (e.g., number and type of illegal activities)

MIST uses a standardised nomenclature for patrol observations, which are arranged at four hierarchical levels, as illustrated in Figure 12.2. Observations are pre-defined by the user and in the MIST database these observations appear as look-up lists to facilitate data entry.

The hierarchical data structure in MIST enables the user to customise observations to a particular site or context, while still ensuring a level of standardisation. For example, in Figure 12.2 observation categories are standardised at the Observation Group, Observation and Observation Type level (corresponding to IUCN-CMP threat definitions) but are site-specific at the Observation Remarks level. Note also that the level of detail at the Observation Remark level will depend upon the needs of a particular site. An example of a MIST data structure standardised across multiple sites (taken from MIKE site in Southeast Asia), and an example of a site-specific data structure expanded for a particular observation type (poaching) is provided in Table 12.2 and Figure 12.3 respectively.

Determining the MIST data structure is a key first step in the implementation of MIST at the site-level (see Box 1) as it will determine the key indicators used for reporting and evaluation. This should be a participatory process involving site managers and technical advisors as appropriate, particularly those familiar with the technical requirements of MIST.

Figure 12.2 *Hierarchical structure of observations in MIST*



Data collection forms MIST observation data is currently collected on paper-based forms by rangers in the field. The data collection forms need to accurately reflect the MIST data structure for a site, in order to standardise observations and facilitate data entry into the MIST database.

Example templates for MIST data forms are provided in Appendix 1, together with summary data collection protocols for rangers. These forms have been designed to minimise written text—i.e., they use check-boxes wherever possible to facilitate standardised data recording, reduce errors and save time. There is some flexibility in the design of data forms: for example, these templates may be modified according to the site-specific observation structure and further adapted to the existing ranger capacity or context at a particular site. However, any modification to the forms should still adhere to the general guidelines listed below and in Appendix 1. It is strongly recommended to field test the data collection forms in order to solicit feedback and input from rangers before finalising both the forms

TABLE 12.2 *MIST data structure, illustrating the four hierarchical data levels in MIST (taken from Stokes 2010).*

Observation Group	*Observation*[5]	*Observation Type*	*Observation Remarks*[6]
Human Activities	Biological resource use	Hunting	People, weapons/gears, patrol action, transportation, species/parts(#), camps, gunshots
		Fishing	People, weapons/gears, patrol action, transportation
		NTFP collection	People, weapons/gears, patrol action, transportation NTFP species (#)
		Logging	People, weapons/gears, patrol action, transportation, wood species (#)
	Mining	Gold panning	People, weapons/gears, patrol action, transportation, gold
	Agriculture	Shifting cultivation	People, weapons/gears, patrol action, Transportation, crops, land status, area, camps
		Plantations	People, weapons/gears, patrol action, transportation, crops, land status, area, camps
		Livestock grazing	People, weapons/gears, patrol action, transportation, livestock(#), camps
	Habitat alteration	Uncontrolled fire	Habitat type, area, age of burning
	Trade[7]	Wildlife	People, weapons/gears, patrol action, transportation, species/parts (#)
		Wood	People, weapons/gears, patrol action, transportation, wood species (#)
		NTFP	People, weapons/gears, patrol action, transportation NTFP species(#)
	Human disturbance	Military exercises	People, patrol action, transportation
		Trespassing	People, patrol action, transportation
		Recreational Use	People, patrol action, transportation
Mammals	Key species (e.g., elephant)	Sighting	Age/Sex(#)
		Kill (for carnivores)	Species
		Track	Measurements
		Carcass[8]	Age of carcass, cause of death, seizures
		Scat/Dung	ID/collection

Features	Salt lick	Wildlife use/ not used	-
	New settlement	-	-
	Human trail	Used/ abandoned	-
	Seasonal village	Used/ abandoned	-

[5] Observations (and observation types) are based upon the standardised system of threat classification found in Salafsky et al. (2008). Not all observations/types will be relevant to all sites.
[6] Only categories are included for Observation Remarks—items under each category (e.g., type of weapons, people, etc.) will be site-specific (see Figure 12.3).
[7] Trade is used specifically for checkpoints/roadblocks or market controls, where illegal activities are detected away from their source.
[8] See Appendix 1 for data collection form

and the MIST data structure; this will help ensure that forms are filled out correctly and fully in the field.

There are three essential data collection forms for MIST:

1. *Patrol authorisation form* Information for monitoring ranger performance and strategy
2. *Patrol movement form* GPS information for all patrol movements and patrol observations
3. *Patrol observation form* Details of all illegal activities

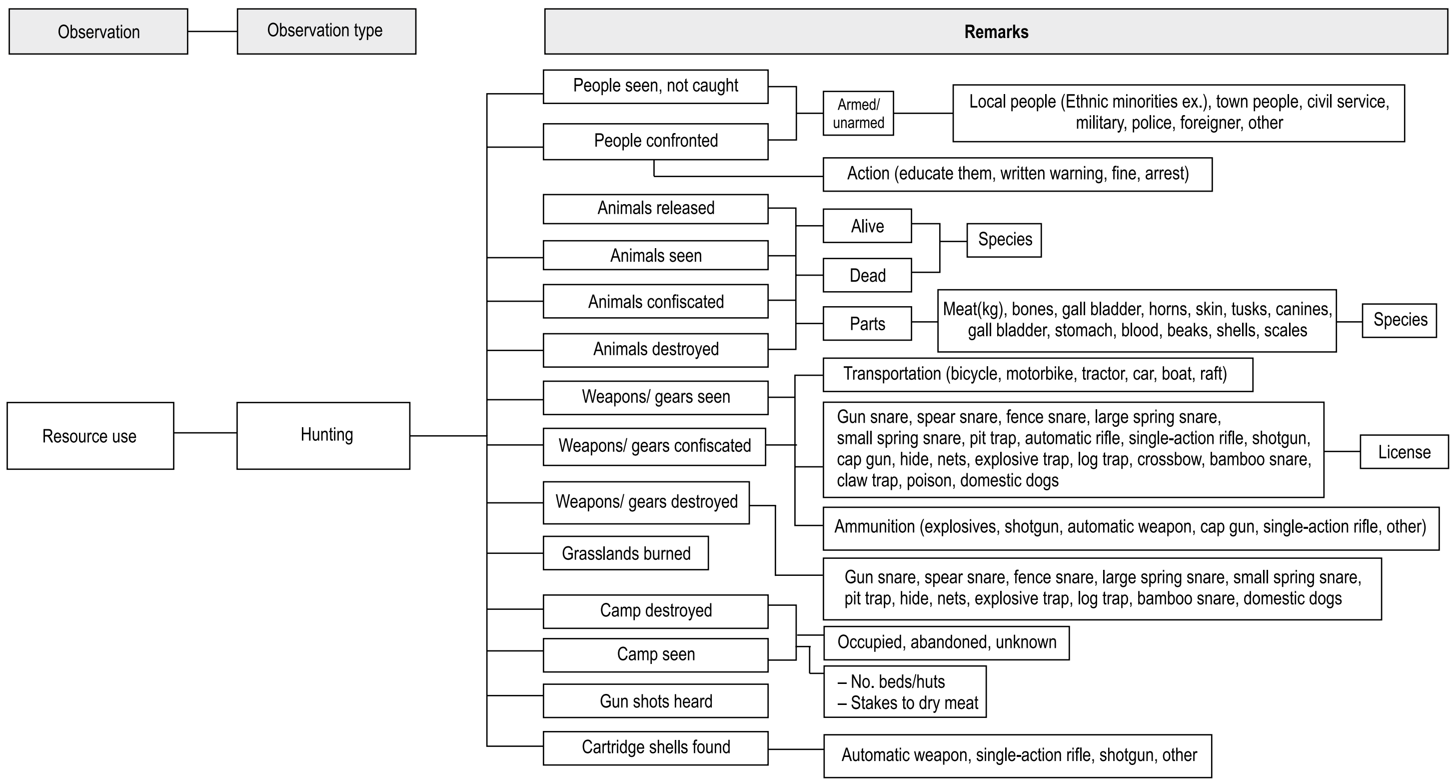
A generic carcass form has also been adapted for MIST (see Appendix 1), which is currently being used in several MIKE sites in Southeast Asia to monitor the illegal killing of elephants.

12.3.3 Defining spatial data inputs

MIST has a built-in spatial component (MIST-GIS) for mapping patrol coverage and the presence of illegal signs. In order to enhance the usability of the patrol data and results, MIST-GIS can also integrate other ancillary spatial information in the form of ESRI shape files. The following GIS data is often added to the MIST database by the MIST User and should be made available:

- Protected area boundary
- Management sector boundaries
- Ranger stations and checkpoints
- Settlements
- Roads and rivers

Figure 12.3 *Example of a site-specific MIST data structure for monitoring poaching (taken from Stokes 2010).*



Because ranger patrols do not systematically cover the whole protected area, it is frequently necessary to subdivide the protected area (or site) into sectors—that are patrolled roughly uniformly. In MIST these are called *management sectors* and they need to be added and spatially defined as a GIS polygon layer in the MIST database. A management sector typically represents the area which is covered by a particular ranger station or guard post within a protected area (also called 'ranges' or 'beats'). Patrol data can be reported either by management sector or for the whole protected area.

12.3.4 MIST outputs

MIST outputs take the form of reports, maps, tables and charts. Standard output formats come pre-installed with the software application. In addition, data-specific outputs using a standard template can be created on-demand and fully-customised output templates can also be created and added by the user using MIST Administrator tools.

MIST summary reports, patrol effort statistics and maps are the primary means of direct and regular feedback of performance (i.e., patrol effort) and threat indicators to site managers. As such, they should be designed in such a way as to be user-friendly, easily understood and tailored to specific local needs, cultures and ranger-operating procedures. Examples of typical outputs include:

- Indicators of illegal activities (expressed as CPUE)
- Distribution maps of illegal activities for monitoring and planning
- Patrol and ranger performance indicators (including number of patrol days and distance patrolled)
- Patrol coverage maps
- Standardised reports to meet institutional requirements

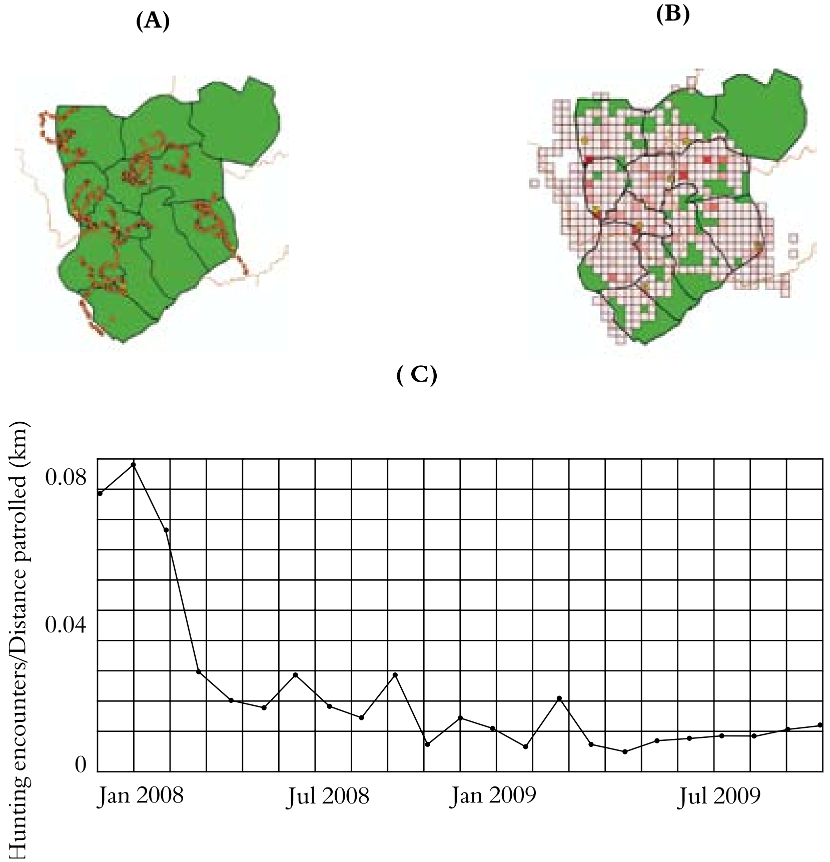
Figure 12.4 shows some examples of standard summary outputs produced and reported directly in MIST. All spatial and attribute data can be further queried by date, patrol, management sector and patrol type and exported as shape files and text files for further analysis.

12.3.5 MIST information flow

In order to provide site managers with prompt up-to-date information it is vital that MIST is fully integrated into management work plans and that regular and direct feedback in the form of MIST reports and outputs are provided as part of the management planning cycle. The MIST information flow at the site level, including roles and responsibilities at each step of the cycle, is illustrated in Figure 12.5.

In order to provide a coordinated flow of information from the site-level to the national-level, where management for planning and resource

Figure 12.4 *Examples of standard MIST outputs for a protected area: A) Protected area with management sectors (black outline), roads (brown line) and guard posts (yellow squares) showing individual routes for reconnaissance patrols on foot for a particular month (orange points); B) Patrol coverage (2.5 × 2.5 km grid squares) for a particular year, showing distribution and intensity of hunting encounters (darker red = higher rate of encounters), C) CPUE of hunting encounters per distance patrolled; monthly summaries presented for all reconnaissance patrols in patrolled sectors only; indicating a decline in hunting rates over the specified time period*



allocation is typically conducted, sharing of MIST data is facilitated and performed by a process of data replication, whereby a central database, housed for example in the relevant wildlife agency's headquarters, receives regular update files (via a USB flash drive or by email) from protected area or site-based databases. Site-users have access only to their individual site database, which is operated on a stand-alone computer. Conversely, users at the wildlife agency headquarters can access information on multiple protected areas from a single central database and conduct further analyses as appropriate. User privileges at all levels can be controlled accordingly (see Figure 12.6).

Figure 12.5 *MIST data flow and adaptive management at the site level*

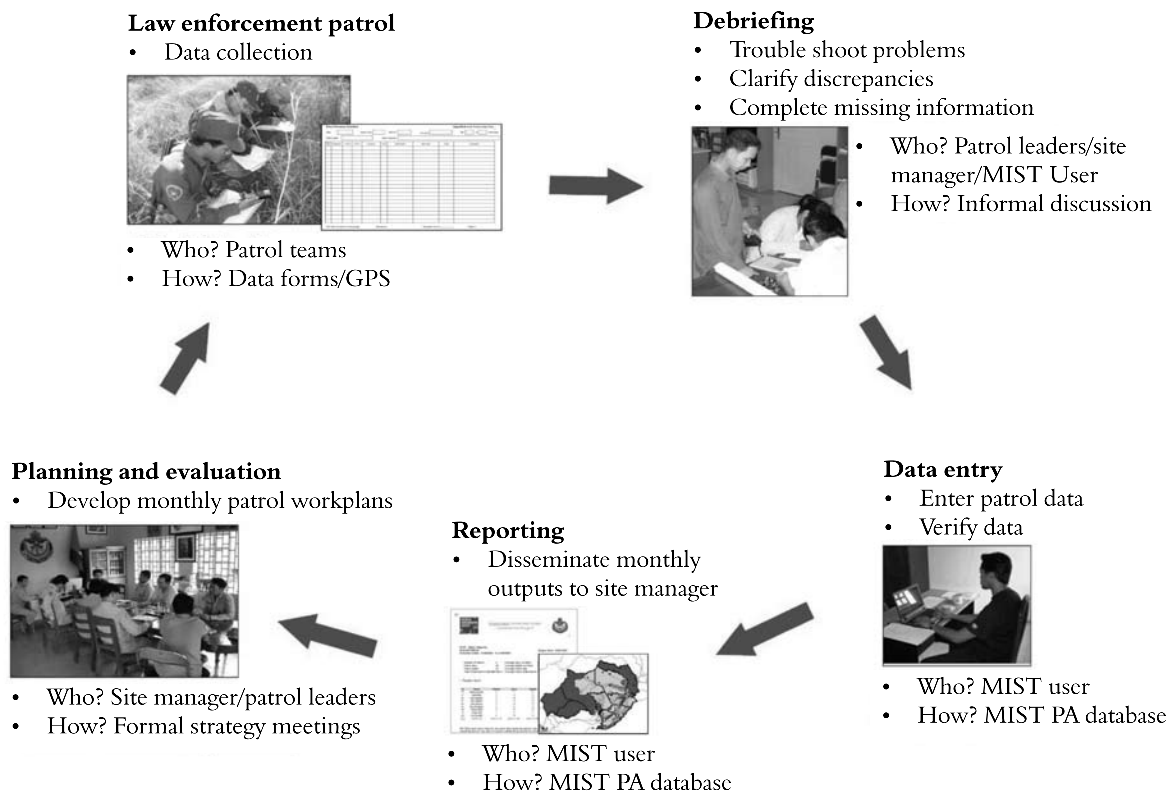
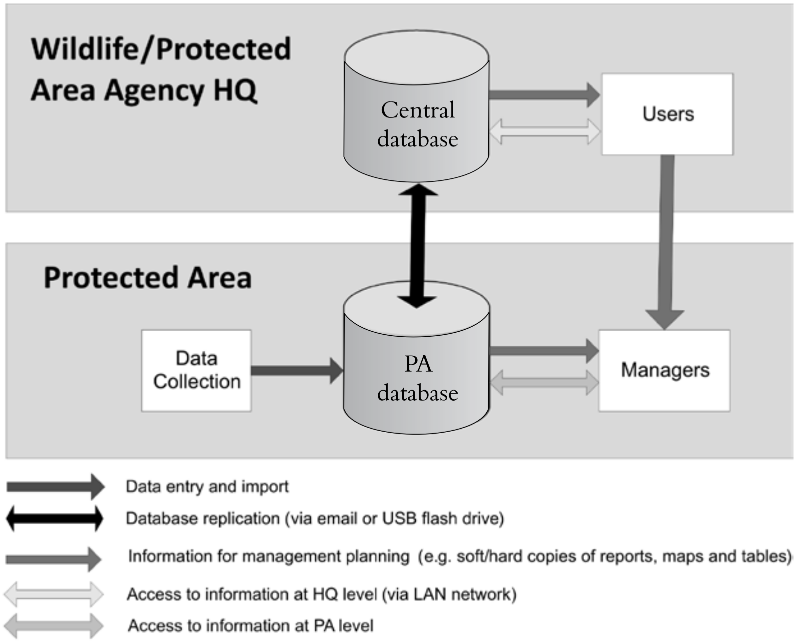


Figure 12.6 *MIST: Data and information flow and user access at site and national level*



In order to maintain a standardised approach to data collection and reporting across all protected areas or sites, database management and editing is typically performed at the central or national level (by the MIST Database Manager). Edits to the database structure (such as changes to the observation structure or reporting templates) are then disseminated to individual sites via the replication process. The result is a flow of *data* from the sites to the central database and a flow of *information* (in the form of consolidated reports) *database and software updates* from the central database to the sites.

12.3.6 Quality control of MIST data

Checks for data quality need to be implemented at all points along the information flow. While technical errors during data entry can be largely avoided by ensuring adequate training of MIST Users, errors due to deliberate falsification or negligent omission of data recording are more challenging to address, yet have been identified [e.g., Jachmann 2008b] as being a major obstacle to effective implementation of LEM programmes. At the level of the patrol teams, this ultimately requires a motivated and dedicated site manager, or someone with authority, to step in and rectify the problem. Formal debriefing sessions after every patrol (see Fig 12.5), with the patrol leader, site manager and MIST User present, can be very useful in correcting errors, clarifying information gaps, empowering patrol leaders and preventing data manipulation. The MIST User can also play an important third-party role in verification of information during the data entry process. A number of formal checks are currently being employed by MIST Users prior to data entry in elephant conservation project sites in Southeast Asia and include the following:

- A Patrol Authorisation and Movement form is submitted by each patrol leader, together with Observation Forms for each human activity observation and any other forms as required.
- All Waypoint coordinates, Time, Observation and Type fields are filled in on the Movement Form
- Intervals of more than 30 minutes between recordings on the Movement form are explained
- Observation Remarks on the Observation Form are clear and complete
- All confiscations recorded on the Observation Form are supported by evidence (physical or photographic) and all recorded arrests/other Patrol Actions are duly reported to the Site Manager with the necessary supporting administrative forms

A number of random checks by the MIST User can also be conducted to safeguard against falsification of waypoints. For example, saved GPS tracklogs on patrol GPS units can be periodically compared with waypoints recorded on the Movement form to ensure integrity of reported patrol routes. Finally, electronic data entry by the MIST User should also be independently checked by the MIST Database Manager (or other third party), through random checks and comparisons with original hard copies of patrol forms. To enable this, all hard copies of all original patrol forms should be safely and appropriately archived on-site.

When data manipulation enters at a higher level of the information chain (e.g., at the level of senior managers) then additional checks are required at the central level. These checks can involve the use of independent (i.e., non-patrol based) data to verify reported levels of poaching by patrol teams (see Section 12.5.2).

12.4 LAW ENFORCEMENT MONITORING THROUGH INFORMANT NETWORKS

For wildlife crimes that are rare, unpredictable or highly covert operations, law enforcement patrols need to be bolstered by strong informant networks and good intelligence, in order to improve detection- and ultimately deterrence-rates. This is particularly the case for wildlife crime that is both organised and lucrative and where opportunity costs are low: i.e., where the potential reward for poachers outweighs the perceived risks of getting caught; for example poaching of elephants for their ivory. Monitoring illegal activities through informant networks has the potential to quickly identify emerging trends and, if implemented effectively, can both complement and strategically enhance patrol-based law enforcement approaches. Compared to patrol-based law enforcement monitoring, even less attention has been given to the methodological and analytical framework of informant networks as a mechanism for monitoring threats to elephants and other wildlife and of evaluating the impact of enforcement interventions. Much can be learnt from the broader criminal literature, such as drug trafficking, which faces many similar methodological problems in understanding trends and evaluating enforcement effectiveness (see Griffiths and Mounteney 2010 for a good overview). A brief introduction of the process and the technical and statistical issues for interpretation of the data is provided here, but this is a field that is rich for further inter-disciplinary research.

Intelligence-based law enforcement relies primarily on establishing effective informant networks, linking enforcement staff with individuals on-the-ground who can provide specific information on illegal activity. Informants can take a variety of forms from enforcement or other protected area staff to community-members and the broader civil society, and can vary in the type of arrangement by which information is provided, from salaried staff, rewards or compensation for *pro-bono* information, to anonymous reports via crime hotlines. Information can be actively sought or passively received. In most instances informant networks are built informally by developing relationships based upon mutual trust, and in most cases are developed at the discretion of one or a handful of key individuals on-site. Effective informant networks take time to develop both in their quantity and quality and their efficacy will likely depend on a number of intrinsic factors relating to incentives, and for community-based informants, land-use and ownership.

A mechanism is needed for converting informant reports into verified information that can, if necessary, be acted upon by the relevant enforcement agencies in a timely manner. Verification is an important part of the process and often achieved by evaluating multiple leads and information sources or through direct physical inspection of an alleged crime scene. For the purposes of enforcement action, the system should be able to integrate intelligence from different informants in order to build up a more complete and accurate picture of the crime, as well as identify key reliable informants and gaps in the intelligence-network. The system also needs to be able to evaluate the efficiency of law enforcement agencies in responding to information, and pursuing crime reports through to a successful conclusion, for example arrest and prosecution. Finally, any system needs to be able to identify trends in the type and nature of illegal activities.

For the purposes of evaluating trends in illegal activities, data collected through informant networks present a number of challenges, and there exist few examples of standardised site-level systems in operation. One exception is the use of a participatory monitoring approach in reporting illegally killed elephants in Laikipia and Samburu, Kenya as part of the CITES-MIKE programme [Kahindi et al. 2009]. Another useful example is the CITES ETIS (Elephant Trade Information System), which uses official enforcement seizure reports to monitor trends in the illicit ivory trade [Milliken et al. 2009]. Although CITES-ETIS is applied at a much larger scale than individual sites, it faces similar methodological issues for monitoring.

What is required from informant networks is knowledge of the *actual level of a covert activity* (for example elephant poaching), whereas the data available are *informant or third-party reports* of this activity. If reports are few, is this because law enforcement is effective and there is no poaching, or because the informant network, intelligence and level of reporting is poor? Similarly, if the number of reports is high, is this because enforcement is poor or because the informant network and level of reporting is well-developed?

There are at least two important variables that we need to understand: reporting effort and law enforcement effectiveness. This is a similar concept to the CPUE indicators for patrol-based LEM. The difference with LEM through informant networks is that reporting effort and law enforcement effectiveness are not directly measurable (they are latent variables[9]); therefore appropriate proxy indicators need to be developed, that are measurable, and which reflect these two processes (Figure 12.7).

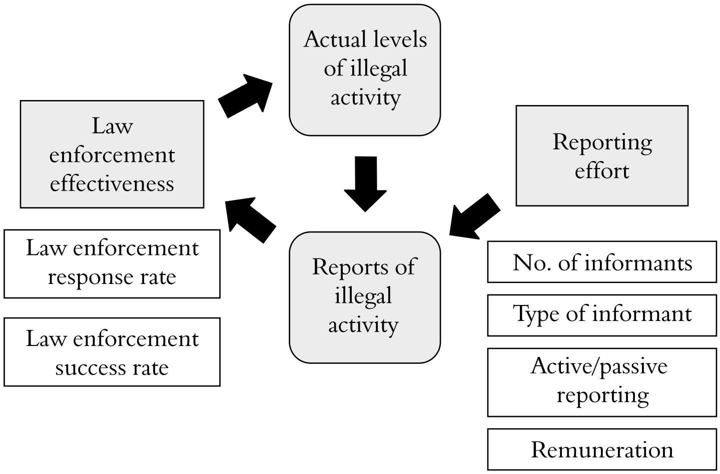
12.4.1 Identifying Proxy Variables

12.4.1.1 Reporting effort

Reporting effort reflects the efficiency of the informant network in reporting illegal activities. This is often a complex system involving reports from multiple sources using multiple methods of data recording obtained through multiple means. Measures for quantifying the reporting effort will depend upon, for example, how many enforcement staff are involved in collecting

[9] Latent variables are variables that are not directly observed but are rather inferred (through an appropriate mathematical model) from other variables that are observed or can be directly measured.

Figure 12.7 *Conceptual basis of law enforcement monitoring through informant networks, showing latent variables (law enforcement effectiveness and reporting effort) and examples of proxy indicators for monitoring listed beneath them*



information and their experience at managing informant networks, the size and coverage of the informant network, who the informants are, if the information was actively or passively received, and whether or not the information was rewarded or compensated in some way. Kahindi et al. (2009) employed both the number of participatory meetings and participants as well as distance travelled by the field researcher to collect informant reports as measures of 'effort' in evaluating the number of illegally killed elephants reported by local communities. CITES-ETIS employs the CITES reporting rate of a particular range State as a measure of reporting effort for illegal ivory seizures (Milliken et al. 2009). At a minimum, information about the nature of the information source, the size of the informant network and a measure of any effort expended in actively obtaining information from informants should be recorded and monitored. Ideally, the relationship between these parameters and the access to information about illegal activities should be evaluated (see Section 12.5.1). There will likely be additional site-specific parameters depending on the context of a particular site.

12.4.1.2 Law enforcement effectiveness

The effectiveness of law enforcement operations in reducing illegal activities depends both on the efficiency with which law enforcement agencies can respond to informant reports (*the ratio between total reports received and total reports that are acted upon*) and the rate at which a law enforcement response results in a successful outcome (*the ratio between total law enforcement responses and the total successful responses)*. The definition of 'success' will depend upon the site and the particular illegal activity, but for an activity such as elephant poaching, a successful law enforcement response might be an arrest, charging of the suspect by police and sentencing in a court of law commensurate with the seriousness of the crime. These are three of several possible proxy indicators for law enforcement effectiveness that are likely to influence the true level of illegal activity and are therefore important variables to monitor. There will likely be other additional site-specific parameters depending on the local context. One example is the role of media exposure as a deterrent, particularly in successful law enforcement outcomes in response to wildlife crime.

12.4.2 Information management tools for intelligence-based LEM

A number of commercially available and valuable tools exist in order to assist law enforcement authorities in conducting intelligence-led investigative approaches to deterring and solving crime. These range from sophisticated applications for developing intelligence networks to performance-monitoring tools for managing and tracking criminal cases (see http://iaca.net/resources.asp?Cat=Software for a relatively thorough review of currently available software for crime analysis). Although none of these were developed with wildlife law enforcement as their primary focus, many of these tools are of considerable use and interest to wildlife law enforcement agents—particularly in tackling organised criminal networks involved in cross-border wildlife trade, and some are currently being used at a variety of different scales by wildlife and environmental agencies to varying degrees. In addition, a number of tools exist with application at the regional or global scale with the purpose of exchanging information on reported wildlife crimes between countries and in managing information on seizures and other wildlife crimes, including the CITES-ETIS database for managing data on the illicit elephant ivory trade.

There is no single commercially available management tool (such as MIST) that addresses all of the issues raised here in managing, monitoring, and evaluating informant led law enforcement approaches at the site-level (although see http://www.pamis.org/trac/cdb/wiki/WildlifeCrimeAbout for a new system under development). Custom-developed in-house systems exist within particular agencies and organisations, but there is little coordination or adherence to any standardised framework for monitoring crime and law enforcement effectiveness. Moreover there exist no management tools of this type which are scalable to a broad range of local contexts and cultures. There is therefore a need for a standardised approach to site-level informant based law enforcement monitoring, and a suitable tool with which to address this need.

There is not likely to be a single management tool that can address all site-based law enforcement monitoring needs (informant networks and patrol-based law enforcement approaches) and different sites will have differing requirements and resources available to them in deciding which tool to use. Nevertheless, the adoption of standardised and complementary monitoring approaches, aimed at improving our understanding of the prevalence of illegal activities, and improving the effectiveness of law enforcement strategies in addressing them is an important first step in this process.

12.5 Improving the Analysis and Interpretation of Law Enforcement Monitoring Data

12.5.1 Statistical approaches for the quantitative analysis of LEM data

There are currently no standard 'off-the-shelf' statistical approaches or models for the quantitative analysis of illegal activities through LEM data. However, a number of modelling approaches exist that can be applied to law enforcement data in order to incorporate sampling error and uncertainty and improve inference of patterns and trends, beyond that of simple CPUE indicators. Modelling approaches should not be seen as a panacea to the inherent technical challenges of law enforcement monitoring and are only as good as the data on which they are based. Their utility will nevertheless be greatly enhanced by data that is collected according to the guidelines presented in this chapter.

Linear regression models [e.g., McCullagh and Nelder 1989] can be used to determine the relationship between indicators of illegal activities and key predictors such as measures of law enforcement effort, including type of patrol, distance patrolled, and number of patrol staff [Jachmann 2008a] in order to extrapolate levels of illegal activity over time. For example earlier analyses of the CITES-MIKE patrol data (CITES 2007) attempted to account for variable patrol effort in encounters of elephant carcasses across MIKE sites by entering measures of effort into the model as a covariate in a linear Poisson parameter [e.g., Borchers et al. 2002]. Generalized Additive Models [GAMS e.g., Wood 2006] can also be appropriate for modelling patrol catch data, particularly where relationships with important predictor variables (i.e., effort) are non-linear. GAMs have been used to model trends in the illegal ivory trade from seizure reports as part of CITES-ETIS, using a number of key covariates including proxy variables for law enforcement effort [Milliken et al. 2009]. Both GAM and Generalized Linear Modeling

(GLM) approaches are also used in constructing standardised indices in commercial fisheries, which are subject to similar sources of bias as patrol-based data (see Maunder and Punt 2004 for a review of recent approaches).

Occupancy-based models [Mackenzie et al. 2002] also have potential for investigating the spatial patterns and processes of illegal activity indicators over time, and for examining the relationship between occupancy, detection probability, and key covariates. Occupancy-based models have the additional advantage of estimating detection probabilities for different illegal activities, and examining the relationship between detectability and covariates such as patrol type (e.g., reconnaissance, intelligence-led) and other key patrol attributes. These models are particularly well-suited to patrol-based data, given their relative robustness to missing values and unequal sampling effort over space and time, but do nevertheless assume reasonable patrol coverage of the area of interest.

LEM data can also be combined with independent data on wildlife abundance and distribution (collected using systematic and rigorous methods as outlined earlier in this manual). Such population-based models have been used to test management assumptions and examine the efficiency and effectiveness of different law enforcement interventions in reducing illegal offtake and impacts on target species [Leader-Williams and Albon 1988; Hilborn et al. 2006; Byers and Noonburg 2007]. Models that examine the factors driving illegal behavior and compliance have been less extensively applied to the conservation of natural resources (although see Milner-Gulland and Leader-Williams 1992) and present a potentially valuable field of interdisciplinary research to better inform the design and implementation of enforcement strategies [Keane et al. 2008].

Finally, simulation models show particular promise for examining trends in illegal activities from enforcement data as they can explicitly address limited data, uncertainty in available data and incorporate additional information from a wide range of different sources [Burton 1999; Pitcher et al. 2002; Magnusson and Hilborn 2010].

12.5.2 Independent assessments of illegal activities

One of the main challenges for the interpretation of trends in illegal activities from law enforcement monitoring data is that the law enforcement teams are both collecting data on illegal activities and deterring illegal behavior. Given this interaction between ranger and poacher behavior, combined with the many other inherent assumptions of patrol-based LEM, it is strongly recommended that periodic and independent (i.e., non-ranger based) assessments of illegal activities are undertaken in order to verify and calibrate LEM results. Examples include wildlife survey teams recording indirect signs of illegal activities, such as carcasses, on systematic transects, reconnaissance lines or plots [Blake et al. 2007], or community-based questionnaires focusing on direct reporting of illegal behavior (see Gavin et al. 2009) for a recent review of the costs and benefits of different methods for recording illegal behavior). Few assessments of the reliability of law enforcement data in monitoring trends in illegal activities compared to independent measures such as self-reporting and direct questioning have been conducted (but see Knapp et al. 2010). Given the added value of law enforcement monitoring in improving overall efficiency and accountability of enforcement efforts, there is considerable value in evaluating further under which conditions LEM is likely to be a more robust quantitative monitoring tool.

Combining different methods for assessing illegal activities also has the potential for testing assumptions governing local driving factors of particular threats and the expected outcomes of law enforcement, and other, interventions. For example, de Merode et al. (2007) compared law enforcement data on poaching violations in the Garamba National Park, Democratic Republic of Congo, with independent market surveys of bushmeat offtake to examine the efficacy of anti-poaching operations on bushmeat hunting during periods of armed conflict, and found that although anti-poaching operations had a demonstrable impact on reducing hunting pressure, social-political factors (i.e., institutions controlling the bushmeat trade) were likely far more influential on hunting levels during periods of civil unrest and should be incorporated into conservation strategies accordingly.

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CHAPTER 13

Using New Methods to Add Value to Old Survey Datasets: Estimating Abundance from Dung Density or Dung Encounter Rates

Simon Hedges

13.1 Introduction

While reliable peer-reviewed methods for estimating elephant population size in relatively small areas of forest and other concealing habitat types have existed for some years and are described in the *Dung Survey Standards for the MIKE Program* [Hedges and Lawson 2006] and in greater detail with suggested improvements in this manual, estimating elephant population size across very large landscapes (especially forested landscapes) has long been—and indeed remains—a significant challenge. The two methods for directly estimating elephant population size in forested areas recommended in the *Dung Survey Standards* and this manual, i.e., dung counts using line transects and fecal DNA based (or direct sighting based) capture–recapture surveys, are primarily suitable for areas smaller than c. 5000 km^2 and areas with fewer than a few thousand elephants, respectively (as discussed in several places in this manual).

For dung counts based methods, the main limiting factor is the need to use so-called retrospective methods for estimating dung decay rates [*sensu* Laing et al. 2003]. In other words, spatially representative samples of fresh dung piles have to be located across the survey area multiple times for a significant (often many months-long) period before every survey (Laing et al. 2003; Hedges and Lawson 2006; Chapters 4 and 9). Work by the Wildlife Conservation Society (WCS) in Asia has shown that such retrospective methods are feasible for survey areas of up to c. 5000 km^2 [Hedges et al. 2005; Hedges et al. 2007a; Hedges et al. 2008; Gumal et al. 2009; also see Hedges et al. in review]. For larger areas the logistic challenges and cost become prohibitive.

Fortunately, fecal DNA based capture–recapture methods have been shown to be capable of producing reliable estimates of elephant population in both Africa and Asia [e.g., Eggert et al. 2003; Hedges et al. 2007a; Hedges et al. 2007b; Hedges et al. in review]. Moreover, recently developed spatial capture–recapture modelling approaches provide improved methods for estimating population density from capture–recapture derived abundance estimates [Efford 2004; Royle et al. 2009a; Royle et al. 2009b]. Fecal DNA based capture–recapture methods are now capable of producing more precise estimates of population size, more quickly and more economically than dung count based methods, while at the same time producing more information about the populations surveyed [Ahlering et al. 2011; Hedges et al. in review]. Thus fecal DNA based methods are now the method of choice even for relatively small areas where retrospective dung decay estimation methods can be used. Nevertheless, there are also practical limitations on the use of fecal DNA capture–recapture methods that currently preclude their use for elephant surveys over very large areas such as those occupied by elephants in Central Africa. For example, Lukacs and Burnham (2005: 3914) suggest that fecal DNA based capture–recapture methods are only likely to be appropriate when elephant population size is likely to be smaller than 'a few thousand individuals'—because otherwise a very large number of samples would have to be collected and analysed making the cost prohibitive.

How, then, can we estimate and monitor elephant population size reliably in large forested landscapes containing more than a few thousand elephants? In Chapters 6 and 11 of this manual, we described recent advances in occupancy survey methods and the integration of population estimation at two geographic scales—occupancy estimation across large landscapes and abundance estimation in selected areas within those landscapes [e.g., the Conroy et al. (2008) approach and the extensions to the Royle and Nichols (2003) approach] and we argued that these methods present what are probably the most reliable currently-available methods for estimating elephant abundance in large landscapes. Nevertheless, because of the challenges (logistic, financial, theoretical) of estimating elephant abundance in very large landscapes, there a number of surveys (primarily in Central Africa) from the last c. 20 years for which the surveyors were only able to estimate dung pile density or—if they attempted to estimate elephant density—for which they were forced to borrow dung decay rate data from other sites/periods [e.g., Blake 2005; Blake et al. 2007; Stokes et al. 2010; Yackulic et al. 2011]. Therefore, it is appropriate to ask whether it would be possible to revisit these old survey datasets and estimate elephant abundance using either newly developed methods or extensions to old methods. In this final chapter, then, we address this question and, specifically, describe (1) a possible extension to the two-phase sampling and modelling in a Bayesian framework approach proposed by Conroy et al. (2008) and (2) reconsider the rainfall/dung density models developed by Barnes [Barnes et al. 1997] and Barnes and Dunn [Barnes and Dunn 2002].

13.2 Two-Phase Occupancy and Capture–Recapture Surveys to Estimate Elephant Abundance from Old Landscape-Level Survey Datasets

13.2.1 Summary of the basic approach

Recall from Chapter 6 that adopting the two-phase sampling and modelling in a Bayesian framework approach that Conroy et al. (2008) propose to estimate the current abundance of elephants across a large landscape would involve, in a first phase, estimating occupancy from surveys to detect elephant sign (dung piles) in all selected sites in the landscape, where selection would be of all sites available (if possible), or a random sample of sites (if not). In a second phase, if a detection threshold had been achieved in the first phase, fecal DNA based capture–recapture sampling would be conducted to estimate elephant abundance. Or following the recommendations of Conroy et al., in landscapes where it is likely that the elephant population is highly over-dispersed, instead of using a threshold value, capture–recapture sampling would be conducted in a randomly selected set of sites (from which detection samples are also taken). In addition, population-size constraints might need to be applied to the definition of sites to keep population size per site below a few thousand elephants. [This constraint follows from the observation of Lukacs and Burnham (2005) that fecal DNA based methods are only likely to be appropriate when elephant population size is likely to be smaller than 'a few thousand individuals' (Chapter 5).] Detection and capture–recapture data would then be used in a joint likelihood approach to model probability of detection in the occupancy sample via an abundance–

detection model. Capture–recapture modelling would then be used to estimate abundance for the abundance–detection relationship, which will be used to predict abundance at the remaining sites, where only detection data were collected.

13.2.2 Extension of these methods to add value to old datasets

Excitingly, a relatively simple development of the Conroy et al. (2008) approach appears to provide a tool that can be used to estimate past abundance reliably and across large landscapes from old sign-encounter-rate or dung density based datasets. As we noted above, given that there are a number of old datasets [e.g., from 2006 for the 28,000 km^2 Ndoki-Likouala landscape (Stokes et al. 2010)] from which it should be possible to estimate occupancy at the time of the survey, it should be possible to use the abundance–detection relationships from new two-phase surveys (in the same landscapes as the old surveys) to estimate elephant abundance for those earlier survey periods for which we can estimate occupancy from the old datasets. In other words, it should be possible to build an empirical abundance–detection relationship from new surveys using the Conroy et al. approach, and then use the relationship so developed to predict abundance for the old survey period(s) having used the original dung encounter rate data sets to estimate occupancy.

A critical assumption is that the abundance–detection relationship remains the same, which seems reasonable assuming that habitat or other conditions affecting detection have not changed over the intervening period. It should also be possible to test this assumption.

We are also assuming here that some of the old data sets lend themselves to estimating occupancy (i.e., that they were collected under a design that incorporated replication in order to account for incomplete detection) which we think is reasonable. Even though the previous elephant dung count based data sets were not collected with the aim of estimating occupancy the fact that they were collected using line transects and recces means that they can, we think, in most cases be used to estimate occupancy. Whether a given survey data set can be used to estimate occupancy will depend on the transect and recce survey effort per unit area (per occupancy survey cell). Estimating occupancy from the early transect/recce datasets would have to be based on spatial replication, and until recently most recommendations were for spatial replicates to be selected randomly and with replacement—which was not how the transects and recces were implemented. Nevertheless, recent work by Hines et al. (2010) suggests that new Markovian occupancy models for data expected to show spatial dependence (such as records of dung piles along transects and recces) work well (Chapter 6), thus allowing for valid inference about occupancy from the old transect- and recce-based elephant dung surveys.

To conclude, we believe that it would be worthwhile to conduct a number of simulations to test the Conroy et al. method under different patchiness/density scenarios in terms of elephant distribution and then use the insights gained to examine the extension to old datasets proposed here.

13.3 Use of Rainfall Data to Calculate Elephant Density from Old Dung Density Datasets

13.3.1 Rainfall-based models: basis and limitations

Recall from Chapter 4 that the rationale behind rainfall/dung density models is to use rainfall to predict expected dung pile density, and use this relationship to calculate elephant density from the estimate of actual dung pile density in the survey area, as suggested by Barnes et al. (1994; 1997). However, Walsh and White (2005) argue that even assuming that one can derive a rainfall model that explained a high proportion of variance in decay rate, the dung piles detectable at the time of the survey to estimate their density will include a variety of cohorts, each of which will have experienced a unique time series of environmental conditions different from that observed during the index decay study. Walsh and White's simulations suggest that, for long-lived sign such as elephant dung piles and ape nests, a single number such as the rainfall in the month before survey, is not likely to adequately capture the unique effects of each cohort's history.

Indeed, as Barnes and Dunn (2002) note, existing rainfall models assume that the dung pile density estimate was derived from transects that were surveyed simultaneously. This is clearly an unrealistic assumption, which complicates the use of rainfall models. If this approach were to be used more widely, further work would be required to calibrate dung pile estimates from sets of transects surveyed in months with different rainfall totals [Barnes and Dunn 2002].

Moreover, the work required to establish the site-specific relationships between expected dung pile density and rainfall is considerable and suggests that the retrospective methods of estimating appropriate dung decay rates of Hiby and Lovell (1991), Marques et al. (2001), Buckland et al. (2001: 183-186), and Laing et al. (2003) are likely to be more efficient, at least for relatively small sites (Chapters 4 and 9). Unfortunately, the retrospective method is impracticable for most large sites or for landscapes because it is too difficult, logistically, to find and monitor multiple cohorts of fresh dung piles in a spatially representative manner (Chapter 4). Note too that while the use of rainfall/dung density/elephant density relationships was intended to avoid the need for dung decay monitoring, establishing the relationship in the first place is difficult and time-consuming for large sites and for landscapes [the three sites where Barnes et al. (1997) derived their rainfall/dung density/elephant density relationships where all less than 370 km^2 in area].

As noted earlier in this manual, as DNA based capture–recapture methods have been refined they have become the method of choice for sub-landscape level (i.e. sites smaller than c. 5000 km^2) elephant surveys, at least those in forests, because they are more precise, more informative, quicker and cheaper than dung count based methods [Hedges et al. in review; Chapters 5 and 10]. For landscape level surveys, where a simple capture–recapture survey is not practicable, the use of combined two-phase occupancy and capture–recapture surveys to estimate elephant abundance (as described above) appears to provide an efficient and informative option (Chapters 6 and 11). In addition, and as described above, the combination of occupancy surveys and DNA based capture–recapture methods provides a potential way of adding value to old dung encounter rate or dung density only data sets (or data sets with low-quality dung decay data). Nevertheless, given the constraints on the use of occupancy based methods (also see below) it is necessary to revisit rainfall/dung density models to re-evaluate their potential utility for landscape-level surveys.

13.3.2 Revisiting rainfall models in order to add value to old dung density datasets

One further potential use of rainfall methods, which may be worth exploring, is as an additional tool for adding value to old dung-density-only datasets: such data sets as those from surveys in Central Africa which were based on line transect (or line transect plus recce) derived estimates of dung density for which there are no appropriate estimates of dung decay rate (e.g. where the surveyors were forced to rely on spatially and/or temporally unrepresentative estimates of dung decay rate).

The two-phase occupancy and capture–recapture surveys described above provide one potential tool for estimating abundance from old datasets but in some cases the old datasets will not be suitable for estimating occupancy because (1) the survey areas were too small relative to elephant home range size to allow estimation of true occupancy (Chapters 2 and 6) and/or (2) there were too few sign survey replicates per unit area. In these cases, it may pay to establish rainfall/dung/elephant density relationships during new surveys in those sites and then use those relationships to estimate elephant abundance at the time of the old surveys from the old dung density datasets.

Note that the two-phase occupancy and capture–recapture surveys described above will probably only work for very large areas (> c. 25,000 km^2) but will work for both dung encounter rate and dung density data sets; while the rainfall methods are only likely to work for relatively small areas (> c. 5000 km^2 because of the difficulty of monitoring the decay of dung piles in a spatially representative manner across very large landscapes) and only work for dung density data sets, not dung encounter rate sets.

Despite the possibilities presented by the use of rainfall models to add value to old data sets, the following points need to be kept in mind:

- A large effort is likely to be needed to derive the rainfall/dung/elephant density relationship, especially given fine-grained variation in rainfall and the effects of faunal communities (e.g. pigs, birds, and beetles) and terrain on dung persistence rates.
- Secondly, there is the problem of low-quality data on historical rainfall. For example, is it even possible to derive sufficiently fine-grained rainfall data for the old Central African survey blocks?

Despite these cautionary remarks, we believe that it is worth re-evaluating rainfall/dung density models both as a tool for current survey needs (see Chapter 4) and for their potential as a means of estimating elephant abundance from old dung density datasets.

13.4 References

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Appendix 1a: Non-Occupancy Survey Datasheets

Elephant Dung Pile Decay (Disappearance) Rate Monitoring Datasheet

Sheet number: *Dates:* *Locations:* *Observers / technicians responsible:*
Data entered to computer? (Y/N): *Computer file name(s):*

Dung pile ref. number	Date dung pile found [dd/mm/yy]	Number of boli	Dung pile moved? (yes/no)	Location [give UTM or lat./long. coordinates] On trail?	Vegetation type, slope, altitude	Information to be recorded for each dung pile during each trip: enter number of days since dung pile was originally found, the date (dd/mm), whether it was found again during subsequent visits, and the stage it was in for the first and last visits								
						Visit #1	Visit #2	Visit #3	Visit #4	Visit #5	Visit #6	Visit #7	Visit #8	Remarks

Line Transect based Dung Count Datasheet

Date (dd/mm/yyyy):	Line transect number:
General description of location:	
Start point (UTM):	
Finish point (UTM):	
Compass bearing:	Distance at finish (m):
Start time:	Finish time:
Team members' names:	

Distance from start (m)	Dung pile data				Other notes
	Number of boli	Perpendicular distance (cm)	Decay Stage	Circumferences (cm)	

Dung 'Hotspot' Survey Datasheet

Date (dd/mm/yyyy): GPS serial number:

General description of location:

Start point (long./lat.): Start time:
Finish point (long./lat.): Finish time:

Team members' names:

Waypoint name	GPS location		Time	Dung pile age: F, RF or O*	Circumfer. (cm)	Domestic or wild / saltlick / pool or waterhole / other notes
	N	E				

* F = fresh; RF = reasonably fresh; O = old; Circumfer = circumference

Sighting based Line Transect Datasheet

Date (dd/mm/yyyy): Line transect number:

General description of location:

Start point (UTM):

Finish point (UTM):

Compass bearing: Distance at finish (m):

Start time: Finish time:

Team members' names:

Distance from start (m)	Elephant cluster data				Other notes
	Number of elephants in cluster	Sighting distance (m)	Sighting angle (degrees)	Composition of cluster (sex, age classes seen)	

DATASHEET FOR COLLECTING FECAL DNA SAMPLES

Date	Sample #	Veg. type	GPS long.	GPS lat.	Decay stage/ height	Fresh (<48 hrs) or reasonably fresh >48 hrs and <4 days	Circumferences of three largest intact dung boli (to nearest cm)	Comments (recent rainfall and other factors that may be important)

Appendix 1b: Occupancy Survey Datasheets

Elephant Occupancy Survey

Field survey form for assessing occupancy of elephants (and other species)

Form no.:	Date:	Grid no.:		
Area surveyed:	Survey team:			
GPS no:	GPS file name:	Page no.:		
Start time:	End time:	Replicate walk no.:		
Start location: X:	Y:	End location: X:	Y:	Detailed description of survey route:

Record no.	Time	Species	Evidence	Nos seen	GPS reading or locality	Wpt no.	Segment no.	Segment type	Substrate condition	Vegetation type	Photo ID no.	Remarks

Instructions: Record each direct sighting of all study species including livestock, scats of Tiger, Leopard and Dhole, and all carcasses and kills seen. In cases of calls heard, tracks of predators and prey, and pellet/dung piles, record ONLY the first encounter of each such evidence within each 100 metre segment. Use only dung dimensions to identify elephant calves. Signs of Tiger cubs should be marked in the Remarks column. Segment No., Segment Type, Substrate Condition and Habitat Type should be entered for every 100 metre segment.

Species codes: TGR = Tiger, LPD = Leopard, DHL = Dhole, BTG = Banteng, BER = Bear, SBR = Sambar, PIG = Wild Pig, MJK = Muntjak, GAR = Gaur, TPR = Tapir, ELP = Elephant, ELC = Elephant calf, LVS = Livestock

Evidence: PLT = Pellet/Dung pile, SCT = Scat, TRK = Track, SCR = Scrape/claw mark, KIL = Kill, DST = Direct Sighting, CAL = Call, CAR = Carcass

Segment Type: ROD = Road, TRL = Trail

Vegetation type: MDF = Deciduous forest, EVG = Evergreen forest, GRS = Grassland, OTH = Other

Substrate condition: SOF=Soft soil, HAR=Hard soil, LLT=Leaf litter, GCR = Grass cover

ELEPHANT OCCUPANCY SURVEY

Field survey form for assessing human impact and disturbance

Form no.: Date: Grid no.:
Area surveyed: Survey team:
GPS no: GPS file name: Page no.:
Start time: End time: Replicate walk no.:
Start location: X: Y: End location: X: Y:
Detailed description of survey route:

Record no.	Time	Impact category	GPS reading or locality	Wpt no.	Photograph no.	Segment no.	Remarks

Impact Categories:

OBE= Organised biomass extractions; LBE= Local biomass extraction; FIR= Fires; NTP= Collection of NTFP; PCH= Poaching; MIN=Mining; CLR=Clearance

Instructions: Record all disturbances seen ONLY ONCE in each 100 metre segment. Use the remarks columns to record the evidence (refer to the explanatory sheet provided for additional details).

Explanatory notes for recording human impacts and disturbance

Disturbance	Evidence types to be entered in the remarks column
Organised biomass extraction (OBE)	Record extraction of: 1. Bamboo, large scale extraction 2. Timber, carried out on large scales, for e.g., truckloads 3. Evidence of camps or processing of cut timber 4. Removal of dead wood at a large scale or through contracts or tenders 5. Removal of leaf litter, at a large scale or through contracts or tenders
Local biomass extraction (LBE)	Record instances of people making away with one or two boles as local logging, extraction lopping and fuel wood extraction Dead wood and leaf litter removal at low scale. Indicate timber/fuel wood/bamboo/dead wood/leaf litter
Fire (FIR)	Record signs of both current and old fires
NTFP (NTP)	Extraction of: 1. Honey 2. Lichen 3. Fruits 4. Roots 5. Bark 6. Leaves 7. Charcoal
	Detection of: 1. Traps (steel jaw traps, etc.) 2. Snares (made of telephone wire, steel cable, rope, vine, etc.) 3. Bones, quills, feathers (sometimes around a fire) 4. Pesticide packets (tell-tale sign: skull and crossbones) 5. Animals smoked out of burrows 6. Crushed *Randia* fruits 7. Traps on trees to poach arboreal animals and birds
Mining (MIN)	Actual mining or quarrying site. Dumping of mining or quarrying waste. Sand, latterite and mud quarrying. Indicate what mineral is being extracted.
Clearance (CLR)	Patches cleared for ganja (marijuana) cultivation Encroachments, indicate new or old. This will have to be verified with toposheets. In case it is not possible to verify any encroachment, indicate human settlement.

General instructions: If you have a local field assistant or people who are familiar with the area, enquire what methods are used to hunt / collect NTFP and look for those signs in addition to the types of evidences included above.

Appendix 1c: Protocol for Rangers for Completing MIST Forms

1. Patrol Authorisation Form

1. One patrol authorisation form is completed for each patrol.
2. The Patrol Identification Number (PIN) is a unique number assigned to each patrol and should be used on all patrol forms used for a given patrol.

2. Patrol Movement Form

1. A new form is to be completed for each patrol day. More than one Movement Form can be filled out during one patrol or during one patrol day.
2. The Run point column on the Movement Form starts with 1 on each form.
3. The Waypoint column must record exactly the same waypoint number as read from the GPS (do not automatically start from 1 on each form).
4. The Location column should indicate the management sector.
5. Observation and type: Use only the observations and type categories listed at the bottom of the form (these categories will be modified according to the MIST observation structure for a particular site). A coding system can be used if desired to avoid writing out options in full.
6. The Totals column is for direct sighting of wildlife only. Leave blank for all other observations.
7. Only observations by the patrol team are recorded, not observations reported by secondary sources. For instance; if the team receives information about a poached Tiger, then this is only recorded on the form if the patrol team goes to the reported location and finds the carcass.
8. If more than one observation type is observed in the same location (e.g. a camp with signs of hunting and NTFP collection), this should be recorded as two separate observation types (Hunting and NTFP collection) on two separate lines.

3. Patrol Observation Form

1. One form is filled out for EACH observation of human activity recorded on the Patrol Movement Form.
2. The Observation Form is linked to the relevant observation on the Movement Form by the Date, Run No., Waypoint No. and Time. These data must be correctly recorded on each form

4. Carcass Form

1. One form is filled out for EACH carcass encountered.
2. The Carcass Form is linked to the relevant observation on the Movement Form by the Date, Waypoint No. and Time. These data must be correctly recorded on each form.

PATROL IDENTIFICATION NUMBER(PIN): (site code/start date (ddmmyy)/leaders initials)

PATROL AUTHORISATION FORM

1. Team name:
2. Reporting officer:
3. Name and rank of patrol leader:
4. Patrol transport:

   Foot ☐ Motorcycle ☐ Vehicle ☐

   Motor boat ☐ Non-motorised boat ☐ Ultra-lite ☐

   Fixed point ☐ Other ☐________________
5. Patrol Type:

   Surveillance patrol ☐ Investigation patrol ☐

   Follow-up patrol ☐ Other ☐

   Road Block point ☐ Market control point ☐

   Bio-inventory ☐
6. Specific patrol objectives

7. Start date of patrol: ______________

   Start coordinate(LAT/LON) ______________
8. End date of patrol: ______________

   End coordinate (LAT/LON) ______________
9. Number of persons in patrol

Name	Organisation	Armed
Total		

10. Name and signature of authorising officer ______________
11. Name and signature of patrol leader ________________

MIST Patrol ID (to be filled in during data entry): Date received:
Date entered into MIST: Entered by:

PATROL IDENTIFICATION NUMBER(PIN): (site code/start date (ddmmyy)/leaders initials)

Patrol Movement Form

Date ______ Patrol Leader ______ GPS User ______ GPS No. ______

Run No.	Waypoint	LAT	LON	Location	Time	Observation	Obs. Type	Total	Adults		Young	Comments
									M	F		

OBSERVATIONS:
Mammals: Tiger, Rhino, Elephant, Sambar, Wild Pig, Sun bear, Muntjac, Serow, Clouded Leopard, Cat spp. Siamang, Orang-utan, Gibbon, Thomas leaf monkey
Features: Saltlick,Waterfall, Rare plants, Cave
Human activity: Human
Position: Position

OBS TYPE:
Mammals: Sign, track, sighting, dung, call, scratch, carcass
Features: Active, Inactive
Human activity: Hunting, fishing, logging, NTFP, gold panning, gold mining, tin mining, oil drilling, gas drilling, shifting cultivation, plantations, livestock grazing, crop farming, forest fire, settlement, road, dam, bridge, other (specify)
Position: Start, rest, re-start, position (every 30 mins), end

MIST Patrol ID (to be filled in during data entry): Date received: Date entered into MIST: Entered by:

PATROL IDENTIFICATION NUMBER(PIN): (site code/start date (ddmmyy)/leaders initials)

PATROL OBSERVATION FORM

PIN: Page of

Entry field for each Observation encounter
Time:
Run no:
Waypoint no :
NOTES:

Wildlife	Seen	Confiscated	Destroyed	Released
Species…..				
Alive (no.)				
Dead (no.)				
Parts:				
Meat (kg)				
Bones				
Gall Bladder				
Horns				
Skin				
Tusks				
Canines				
Stomach				
Blood				
Beaks				
Shells				
Scales				
Other……….				

Agriculture
Crops seen…………………
Clearing: ☐ Burnt: ☐
Planted: ☐ Size (ha)……

Livestock
Cow (no)…………………………
Pig (no.)…………………………
Buffalo (no.)………………………
Goat (no.)………………
Size(ha) of grazing…………….

People	Armed	Unarmed	Unknown
Local people			
Gayo			
Aceh			
Batak			
Nias			
Padang			
Java			
Kluet			
Singkil			
Alas			
Town people			
Civil service			
Military			
Police			
Other……….			
Seen ☐ Confronted ☐			

Action:
Educate ☐
Written warning ☐
Arrest ☐
Case No……………………

Commercial mining activities
Licence? ☐

Weapons	Observed	Confiscated	Licence
Home-made gun			
Military gun			
Sport-hunting gun			
Other……….			
Cartridges Found: ☐			

Ammunition	Observed	Confiscated
Home-made gun		
Military gun		
Sport-hunting gun		
Other……….		

Transportation		
Bicycle		
Motorbike		
Cart		
Tractor		
Car		
Boat		
Raft		
Buffalo		
Horse		

Fire
Forest ☐ Grassland ☐
Burning ☐ Smoking ☐
Blackened ☐ Regrowth ☐
Size(ha)…………………

Gears	Seen	Confiscated	Destroyed
Parang			
Knife			
Spear gun			
Pit trap			
Live pit trap			
Box trap			
Domestic dogs			
Nets			
Leg-snare			
Body-snare			
Spear-snare			
Poison			
Jaw-trap			
Glue			
Fire			
Axe			
Hook			
Handsaw			
Chainsaw			
Bulldozer			
Fishing net			
Fish trap			
Fishing rod			
Battery/electric rod			
Dynamite			
Spear			
Headlamp/mask			
Hoe			
Spade			
Pan			
Mercury			
Other……….			

Camps/Huts	Observed	Destroyed
Occupied		
Abandoned		
Unknown		
# People/beds		

Logging	Observed	Confiscated	No. of tree	No. logs	M 3
Species 1: …………					
Species 2: ……………					
Species 3: ……………					
Species 4: ……………					

NTFP	Observed	Confiscated	Kg
Species 1: ……………			
Species 2: ……………			
Species 3: ……………			

Gold	Observed	Confiscated
mg		

MIST Patrol ID (to be filled in during data entry): Date received: Date entered into MIST: Entered by:

PATROL IDENTIFICATION NUMBER(PIN): (site code/start date (ddmmyy)/leaders initials)

Carcass Form

Location ☐ UTM E ☐
UTM N ☐ Waypoint No. ☐
Date ☐ Time ☐
Team ID. ☐ Reporter ☐
Animal species ☐ Photo ID: ☐
Sample ID: ☐

Age of carcass: ☐ Fresh (still intact) ☐ Recent (decomposing)
☐ Old (only bones)

Cause of death:

☐ Unknown	☐ Natural or management	☐ Illegally killed	
Description:	☐ Disease	☐ Spear gun	☐ Shotgun
	☐ Forest Flood	☐ Blow pipe	☐ Handmade gun
	☐ Predation	☐ Miltary weapon	☐ Nets
	☐ Roadkill	☐ Traditional snare	☐ Spear
	☐ Other_____	☐ Large cable snare	☐ Trap
		☐ Small cable snare	☐ Poison
		☐ Pit trap	☐ Other_______
		☐ Hunting Rifle	
Please provide details on what led you to this conclusion			
Please indicate the motivation for the illegal killing	☐ Meat ☐ Horns/antlers ☐ Stomach ☐ Gall bladder ☐ Blood	☐ Bones ☐ Tusks ☐ Canines ☐ Other_________ ☐ Unknown ☐ Conflict	☐ Genitalia ☐ Claws ☐ Skull ☐ Skin ☐ Whiskers ☐ Bezoar stones

☐ Confiscations ☐ Bury ☐ Burn
☐ Tusk ☐ Antler/horn ☐ Canine tooth/fang ☐ Skull
☐ Paw ☐ Body ☐ Meat ☐ Skin
☐ Bones ☐ Other, specify ___________

Sex of animal: ☐ Male ☐ Female ☐ Unknown
Age of animal: ☐ Adult ☐ Young ☐ Unknown

MIST Patrol ID (to be filled in during data entry): Date received:
Date entered into MIST: Entered by:

Appendix 2: Equipment Needs for Dung Surveys

Requirements of each line transect dung survey team

Navigation and data collection

- Maps (and if available satellite images and/or aerial photographs)
- Two sighting compasses
- Two GPS units, plus copious batteries
- Two topofils (e.g., HipChains) and adequate thread
- Three metal measuring tapes (5 metres)
- Two flexible plastic measuring tapes to record the circumferences of dung boli
- Approved datasheets and folders and/or waterproof notebooks
- Plastic Ziplock bags to protect notebooks and datasheets from water; water-tight bags
- Pencils and indelible pens
- Dung classification field reference sheet (illustrated with photos and diagrams)
- Line transect field methods reference sheet (e.g., the relevant sections from this manual)
- Standard vegetation-type classification reference material (appropriate for site/region)
- Simple cheap digital camera for recording carcasses and other things of interest (optional)

Cutting transects

- Cutlasses, machetes, parang (or equivalent)
- Secateurs
- Pole or stake that is at eye level when pushed into the ground and which is of large enough diameter to support a sighting compass

Requirements of each dung decay monitoring team

Dung pile monitoring equipment

- Metal or plastic (PVC) stakes for marking dung piles

- Hammer
- Brightly-coloured flagging tape (for indicating dung pile locations)
- Red or 'day-glo' pink and paint brushes
- Permanent marker pens (lots!)
- Two GPS units, and copious batteries
- Two sighting compasses
- Approved datasheets and folders and/or waterproof notebooks
- Water-tight bags for datasheets and folders
- Pencils and indelible pens
- Dung classification field reference sheet (illustrated with photographs and diagrams of classified dung piles)
- Vegetation classification field reference material (appropriate for site/region)
- Clinometer

Requirements of each fecal DNA survey team

Navigation and data collection

- Maps (and if available satellite images and/or aerial photographs)
- Two sighting compasses
- Two suitable GPS units, plus copious batteries
- Two topofils (e.g., HipChains) and adequate thread
- Clinometer
- Waterproof notebooks and approved/or datasheets
- Plastic Ziplock bags to protect notebooks and datasheets from water
- Pencils and indelible pens
- Dung classification field reference sheet (illustrated with photographs and diagrams)
- Fecal concentration survey field methods reference sheet (e.g. the relevant sections from this manual)
- Standard vegetation-type classification reference material (appropriate for site/region)
- Simple cheap digital camera for recording carcasses and other things of interest (optional)
- Cutlasses, machetes parang (or equivalent)
- Secateurs

Sample collection equipment

- 30–50 ml polypropylene tubes with polypropylene caps, lots!
- Plastic forks, large numbers

- Queen's College Buffer (20% DMSO, 0.25 M EDTA, 100 mM Tris, pH 7.5, saturated with NaCl)
- Parafilm for sealing tubes
- Permanent marker pens
- Latex gloves, lots!
- Ziplock bags, lots!
- Test-tube rack
- Saucepan that can hold test-tube rack when boiling samples
- Small camping stove
- Non-transparent boxes to store samples out of the light

Appendix 3: Data Management

Adapted from 'Hedges, S. & Lawson, D. 2006. *Dung Survey Standards for the MIKE Programme*. CITES MIKE Programme, Central Coordinating Unit, PO Box 68200, Nairobi, Kenya.'

1 File Management System

- Ease of analysis and reporting and guarding against dataloss requires a logical and simple file management system.
- Every file generated should find a suitable place in your file structure. There will clearly be a need to add sub-directories as files proliferate, but all directories should follow the same principles.
- Filenames should accurately describe the contents of the file. A file named 'Survey results.xls' is meaningless. That file will be lost easily.
- All files should follow a clear and unambiguous naming system. For example, you could use the following pattern:
- Site name _typeofdata_startdate_enddate.xxx
- Dates should start with the year, month and day (i.e., yyyy/mm/dd)
- For example in the case of transect data: Minkebe_transectdata_ 20030725_20030921.xls
- Or for a progress report: Minkebe_progressreport_LEM_20030912.doc

2 Backing Up Data

2.1 Introduction

Survey notebooks, datasheets and electronic data including computer files and GPS data are the product of weeks or months of hard physical toil. If these data are lost the effort and money used to collect them will have been for nothing. This is not only embarrassing; it may have grave consequences for the monitoring program, and for the future funding potential of important field operations. Yet the annals of field biology are replete with tales of biologists who have lost their original data through theft, fires, accident or computer failure. Data are at risk until copies have been made, distributed and stored in at least two different physical locations.

2.2 General methods

- Make at least three photocopies or electronic scans of the original datasheets (digital cameras provide an additional, often quicker, method of copying datasheets). All copies, scans and photographs should be checked for legibility and completeness before distribution. These copies/scans/photographs should be distributed to the relevant people, e.g., one set to the monitoring project officer (or equivalent); the second set should be given to the site management authority (or equivalent); the third set should be kept by the survey team leader. Even after they have been entered into a computer, original data-sheets are still valuable for verification purposes.

- Do not keep the photocopies of the datasheets with the originals: if the original are lost, then the copies will be lost too. Keep the copies in a separate building (to guard against fire or theft).
- It is highly desirable that all notes and records on datasheets and in notebooks be transcribed into digital format (entered into a computer) but this is not always possible. Notes are often invaluable to understanding the background or context of numeric data, and they are frequently overlooked if not entered into the electronic database. Keywords may be used to help searches for important events qualitatively described in the notes section. All notes should be referenced by the date and time they were written.
- All computer files must be backed up. At the end of each day make a copy of each file that you have changed during the day (or better still use a computer program that does this automatically). Use two USB external hard disk drives or flashcards to copy files, and either take the backup home or store it in a separate building. There should always be three copies of any computer file, one on your computer's hard disk drive and two copies on external hard disk drives or other media (flash drives, DVDs, etc.).
- It is also essential, if possible, to email backup copies of all data collected to relevant people 'off site'. See, e.g., the discussion of a possible email backup strategy in Chapter 9.
- Every week all data should be backed up again either onto a CD, DVD or an external hard disk drive.
- Every month a complete copy of all files—data and otherwise—should be burned onto DVDs or CDs and distributed as follows: one copy to the monitoring project officer (or equivalent), one copy to the site management authority (or equivalent), one copy to remain with the person who was responsible for data collection and transcription.
- When the editing work is completed (e.g., all the data from a transect survey have been transcribed into a database file), a complete copy should be burnt onto DVDs or CDs and distributed as follows: one copy to the monitoring project officer (or equivalent), one copy to the site management authority (or equivalent), one copy to remain

with the person who was responsible for data collection and transcription.

3 Data Reporting

3.1 Writing survey reports

Each survey must be fully documented for posterity. Reports should follow standard scientific writing practices. The narrative report must give a full description of the survey and typically should contain the following sections (which are here illustrated by taking a line transect based dung survey as an example):

Background

1. Location, dates, description of the area
2. Summary of previous information (e.g., past surveys)
3. Objectives
4. Survey design, stratification and sampling intensity

Results

5. Tables for each stratum showing observations of dung piles and perpendicular distances for each line transect
6. Print-outs from program DISTANCE
7. Calculations of elephant density
8. Any other important observations made on line transects (e.g., illegal activities, observations of elephants) or during the pilot study

Discussion

9. Compare with previous surveys and comment on any problems that were encountered.

References

10. Sources for pre-existing information about the area should be cited.
11. Sources of methodology/design unique to the survey should be quoted.

Appendices

12. Details of methods
13. List of personnel
14. Dates of each trip into the forest

15. Map of survey zone showing strata and location of each transect
16. Copy of the original datasheets (the data should be copied onto a CD or DVD that should be included with the narrative report following the data management protocol)
17. List of the files and formats for the data on disc, and a brief description of each file

Appendix 4: Websites for Free Analytical Software and Other Resources

Websites for Free Analytical Software

Line transect methods

The program Distance (version 6.0 at time of writing) and the out-of-print introductory book by Buckland et al. (1993) are downloadable from the website of the Research Unit for Wildlife Population Assessment, University of St. Andrews, Scotland, UK. A wealth of additional resources including an extensive list of distance sampling related references and information about the Distance listserv is also available from the website: http://www.ruwpa.st-and.ac.uk/distance/.

Capture–recapture methods

The programs MARK and CAPTURE and other related resources are available from the website of Gary White at Colorado State University, Fort Collins, Colorado, USA: http://warnercnr.colostate.edu/~gwhite/mark/mark.htm. The Program MARK online discussion forum, Analysis of Data from Marked Individuals, can be found at http://www.phidot.org/forum/index.php.

The program CAPTURE is also available from the website of the USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA: http://www.mbr-pwrc.usgs.gov/software.html.

The program SPACECAP is a software package for estimating animal abundance and density using Bayesian spatially-explicit capture–recapture models, and was developed by the Wildlife Conservation Society's India Program in collaboration with Jim Nichols, Andy Royle and Jim Hines at the USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA. SPACECAP is available through the Comprehensive R Archive Network (CRAN): http://cran.r-project.org/ (select SPACECAP from the list of packages). The package runs the spatially-explicit capture–recapture models from the paper by Royle et al. (2009), which was published in *Ecology* (see Chapter 6). Program SPACECAP provides a user-friendly software package that practising wildlife biologists and managers should find simple to use for analysing capture–recapture data. Additionally, SPACECAP reliably calculates density and abundance by making use of the spatial locations of animal captures, and overcomes the problems in using the earlier *ad hoc* techniques discussed in Chapter 6. You will need to install the R language program in your computer before being able to install this package. SPACECAP is available in both Windows and Mac versions and comes with a comprehensive 'help' file.

Program SECR, Spatially explicit capture–recapture in R, version 1.4, by Efford (2010) is available from the Department of Zoology, University of Otago, Dunedin, New Zealand and can be downloaded from: http://www.otago.ac.nz/density/.

The program DENSITY (Efford et al. 2004) also implements methods for estimating the density of animal populations from capture–recapture data (including spatially-explicit models) and is available from the website: http://www.landcareresearch.co.nz/services/software/density/index.asp.

Program CloseTest is a Windows program for testing capture–recapture data for closure, where closure means no individuals were added to or lost from the population of interest over the sampling period. Test statistics are computed using the closure test presented in [Stanley, T. R. and K. P. Burnham. 1999. A closure test for time-specific capture–recapture data. *Environmental and Ecological Statistics* 6: 197–209.] For additional information on how to use the program and its results, see [Stanley, T. R. and J. D. Richards. 2005. Software review: a program for testing capture–recapture data for closure. *Wildlife Society Bulletin* 33: 782–785.] CloseTest is available at: http://www.mesc.usgs.gov/Products/Software/clostest/

Program RELEASE computes survival estimates and goodness-of-fit tests for a large class of survival experiments based on capture–recapture of marked populations. The general model is the Cormack-Jolly-Seber model for each experimental group (survival and capture probabilities different for each group), with a progression of submodels to the null model of the same survival and capture probabilities for all groups. Details of the procedures and a user's manual are provided in [Burnham, K. P., D. R. Anderson, G. C. White, C. Brownie, and K. H. Pollock. 1987. Design and analysis methods for fish survival experiments based on release-recapture. *American Fisheries Society Monograph* 5. 437 pp.] The program and example input files from the AFS Monograph cited above can be downloaded from: http://warnercnr.colostate.edu/~gwhite/software.html. Please note that RELEASE.EXE runs interactively in DOS although another file (REL_32.EXE) is available for handling larger jobs by means of batch processing in Windows 95 and NT. Note that the Windows 95 and NT version do not have the interactive interface provided in the RELEASE.ZIP file. Please also note that all the models computed with RELEASE can now also be computed using Program MARK, and continued use of RELEASE is not recommended, 'except possibly for investigating the goodness-of-fit of a model and some simulations'.

Occupancy based methods

The program PRESENCE estimates patch occupancy rates and related parameters and is available from the USGS Patuxent Wildlife Research Center's website: http://www.mbr-pwrc.usgs.gov/software/doc/presence/presence.html. Worked examples and a user manual are also available on the same site.

The program GENPRES generates patch occupancy data and analysis using program MARK and is also available from the Patuxent Wildlife Research Center's website. A user manual is available on the same site.

Dung disappearance rate analysis

A GENSTAT macro (program add-in) for retrospective analysis of dung decay (disappearance) rate data using the method described in Laing et al. (2003; Chapters 4 and 9), which was written by Bob Burn, is available from the authors of this manual. An R language program that does the same analysis, written by Mike Meredith of the Wildlife Conservation Society (WCS) Malaysia Program, is also available from the website: http://www.wcsmalaysia.org/analysis/Nest_dung_decay.htm. In addition, take a look at the exercise that Mike Meredith has put together on estimating decay rates using the retrospective available at the same website.

The DUNGSURV program discussed in Section 4.6.2.4 is also available on the web (with supporting material): http://www.conservationresearch.co.uk/dungsurv/dungsurv.htm.

Other Software Resources

R is a powerful and flexible language and environment for statistical computing and graphics. It is similar to the S language and environment which was developed at Bell Laboratories (formerly AT&T, now Lucent Technologies) by John Chambers and colleagues. R can be considered as a different implementation of S. R is available as Free Software and it compiles and runs on a wide variety of UNIX platforms and similar systems (including FreeBSD and Linux), Windows and MacOS. To download R, see the website: http://www.r-project.org/.

A number of commercially available and valuable tools exist in order to assist law enforcement authorities in conducting intelligence-led investigative approaches to deterring and solving crime. These range from sophisticated applications for developing intelligence networks to performance-monitoring tools for managing and tracking criminal cases (see http://www.iaca.net/Software.asp for a relatively thorough review of currently available software for crime analysis). Although none of these were developed with wildlife law enforcement as their primary focus, many of these tools are of considerable use and interest to wildlife law enforcement agents—particularly in tackling organised criminal networks involved in cross-border wildlife trade—and some are currently being used to varying degrees.

CAMERA TRAP MANUFACTURERS

For a review and comparison of camera traps see www.chasingame.com and www.jesseshunting.com/site/gamecams.html.

We advise readers to check the relevant literature (published studies that used camera traps, survey reports, etc.), websites, and listservs for up to date news on the availability and specification of camera trap equipment. The following manufacturer's are useful starting points:

www.crowsystems.com/cameras.htm
www.trailmaster.com
www.camtrakker.com
www.trailsenseengineering.com

ADDITIONAL RESOURCES

In addition to the sites referenced above, the following sites are also likely to be of interest:

The Vermont Cooperative Fish and Wildlife Research Unit Spreadsheet Project (http://www.uvm.edu/rsenr/vtcfwru/spreadsheets/) provides many useful exercises in occupancy estimation and modeling, estimating and modeling abundance and estimating demographic parameters.

The IUCN/SSC Primate Specialist Group provides a series of best practice guidelines for great ape conservation, including surveys and monitoring of great ape populations much of which is also of interest for elephant populations. The guidelines can be found at: http://www.primatesg.org/BP.surveys.htm.

Appendix 5: Abbreviations, Acronymns and Glossary of Technical Terms

[The glossary component of this appendix is adapted and expanded from that in: Thompson, W. L., White, G. C. and Gowan, C. 1998. *Monitoring vertebrate populations*. Academic Press, New York, NY, USA.]

Absence Non-occurrence of a species in a sampled unit. This is distinct from non-detection, which describes the situation where the species may or may not be present but is not detected.

Abundance Total number of individuals or items of interest in some defined area and time period; also known as absolute abundance.

Adaptive sampling Adaptive sampling is a sampling design in which sampling regions, defined as 'units', are selected based on values of the variables of interest observed during a sampling survey. For example, in a survey to assess the abundance of a rare or elusive species, neighbouring sites may be added whenever the species is encountered. By contrast, in conventional sampling design, the selection for a sampling unit does not depend on previous observations made during an initial survey; entire sampling units are selected before any sampling occurs. The primary advantage of adaptive sampling methods is the ability to obtain more precise estimates of population density because for a given sample size and cost, more data can be collected than is possible under conventional designs. To use the adaptive sampling technique, however, different estimators must be used to avoid biases.

AfECF African Elephant Conservation Fund of the USFWS.

AsECF Asian Elephant Conservation Fund of the USFWS.

Bias A persistent statistical error associated with parameter estimates whose source is not random chance. More precisely, bias is the difference between the expected value of a parameter estimate and the true value of the parameter. For example, a negatively biased estimator produces estimates that, on average, are smaller than the true quantity being estimated.

Capture–mark–recapture (CMR) Sampling approach and associated models for estimating abundance in which animals are captured or otherwise marked (or identified using natural markings or using genetic markers) and subsequently recaptured or re-identified from their marks.

Capture probability (p) Probability that an animal that is alive at a particular sampling occasion in a CMR survey is captured.

CPUE Catch per unit effort; estimation of abundance based on the relationship between numbers of animals caught and known capture effort. The relationship between patrol effort and observations of illegal activity can also be expressed as catch per unit effort.

Census A complete count of individuals, objects or items within a specified area and time period. A census generally refers to a complete count of all elements within a sampled population and/or target population; however this term also may be applied at the level of the sampling unit to represent a complete count of elements within a sampling unit, such as a 'plot census'. 'Census' is frequently misused as a synonym for 'survey'. True censuses are extremely rare in work with wildlife populations.

CI *See* Confidence interval.

CITES The Convention on International Trade in Endangered Species of Wild Fauna and Flora. CITES is an international agreement between governments with the aim of ensuring that international trade in specimens of wild animals and plants does not threaten their survival.

Closed population A fixed group of individuals within a defined area and time period, i.e., there are no births, deaths, immigration and emigration in the area for the period of interest.

CMR *See* Capture–mark–recapture.

Coefficient of variation (CV) Ratio of a standard error of a parameter estimate to the parameter estimate. The coefficient of variation is used in computing sample sizes and as a measure of relative precision when comparing degree of variation among different estimates or sets of data.

Confidence interval (CI) An interval around a parameter estimate that provides a measure of confidence regarding how close a sampled-based estimate is to the true parameter. The usual two-sided symmetrical confidence interval around the parameter estimate is generated by adding and subtracting the quantity computed from the product of the standard error and the t value or z value corresponding to the pre-specified (1-K) % confidence level (K is frequently set at 0.05). For example, a 95% confidence interval will contain on average, the true parameter of interest 95 of 100 times if 100 such intervals were calculated in a like manner. That is, confidence refers to the procedure of obtaining an interval rather than the interval itself. There is not a 95% probability that the true parameter occurs in the interval; either a parameter is in the interval or it is not.

Covariate A variable that may be related to a parameter of interest. Sometimes this relationship is of direct interest. In other cases, the covariate relationship is not of intrinsic interest itself. In an analysis of covariance, the relationship between the dependent variable and the covariate is first adjusted for before the effects of the other factors are examined.

CV *See* Coefficient of variation.

Density Total number of individuals or objects of interest per unit area (also known as absolute density). Sometimes, the concept is broadened to mean number of animals per unit resource, where resource could be suitable habitat, food abundance, etc.

Detectability Probability of correctly noting the presence of an element within some specified area and time period. Detectability can also be viewed as the expected proportion of elements that is detected.

Distance sampling Count-based sampling approach where the data from the distances to the detected objects are used to model incomplete detection.

DNA Deoxyribonucleic acid is a nucleic acid that contains the genetic instructions used in the development and functioning of all known living organisms (with the exception of some viruses). Scientists can use DNA in faeces, hair, skin, etc. to identify individual animals and thus estimate abundance using, for example, capture–mark–recapture methods.

Estimate A numerical value calculated from sample data collected from a sampled population and used to represent the parameter of interest.

Estimator A mathematical formula used to calculate an estimate. An unbiased estimator will produce unbiased estimates when appropriate assumptions are satisfied.

ETIS Elephant Trade Information System; ETIS is a comprehensive information system to track illegal trade in ivory and other elephant products, managed by TRAFFIC on behalf of the CITES Parties and currently housed at the TRAFFIC East/Southern Africa office in Harare, Zimbabwe.

FFI Fauna & Flora International, an NGO with a mission 'to act to conserve threatened species and ecosystems worldwide, choosing solutions that are sustainable, based on sound science and take into account human needs.'

GIS Geographical information system; a tool for capturing, storing, managing, analysing and presenting geographical data (data that are linked to location(s)). Can be thought of as the merging of cartography, statistical analysis and database tools.

gof Goodness of fit, the measurement of agreement between a statistical model and the data under test.

GPS Global positioning system; a global navigation satellite system that provides reliable location (and time) information anywhere where there is an unobstructed line of sight to sufficient number of GPS satellites.

Index A relative measure used as an indicator of the true state of nature.

Index of relative abundance Count statistic believed to provide a proportional measure of the number of individuals within an area.

Index of relative density Count statistic believed to provide a proportional measure of the number of individuals per unit area.

IUCN International Union for Conservation of Nature, 'helps the world find pragmatic solutions to our most pressing environment and development challenges. It supports scientific research, manages field projects all over the world and brings governments, non-government organisations, United Nations agencies, companies and local communities together to develop and implement policy, laws and best practice.'

LEM Law enforcement monitoring.

MIKE Monitoring the Illegal Killing of Elephants; a CITES program the overall goal of which is to provide information needed by elephant range States to make appropriate management and enforcement decisions, and to build institutional capacity within the range States for the long-term management of their elephant populations. More specific objectives within this goal are: a) to measure levels and trends in the illegal hunting of elephants; b) to determine changes in these trends over time; and c) to determine the factors causing or associated with such changes, and to try and assess in particular to what extent observed trends are a result of any decisions taken by the Conference of the Parties to CITES.

MIST Management Information SysTem, initiated in 1997 through a collaborative project between GTZ (Deutsche Gesellschaft für Technische Zusammenarbeit) and the Uganda Wildlife Authority (UWA) for implementation across Uganda's protected area network. MIST was custom-built to meet the law enforcement monitoring needs of protected area managers by collating standardized data on measures of law enforcement effort, observations of illegal activities, and patrol actions and converting these into useful information for management planning.

Model A conceptual, graphical, algebraic, numerical or other abstraction of the real world.

Monotonic relationship A relationship that is continually increasing (or decreasing).

Occupancy Probability that a site is occupied.

Occupied At least one individual is present at a site (although it may not be detected).

Open population A group of individuals whose number and composition are not fixed within a defined area over a period of interest, i.e., there could be births, immigration, deaths and/or emigration in the area over the period of interest.

Parameter An unknown numerical quantity or constant associated with some measure of a target population.

pdf probability density function; a statistical measure that defines a probability distribution for a random variable and is often denoted as *f(x)*. If a pdf is graphically portrayed, the area under the graph will indicate the interval under which the variable with fall.

PIKE The Proportion of Illegally Killed Elephants, is expressed as a ratio of illegally killed elephants to all elephant carcasses encountered and has been used to determine trends in poaching pressure.

Plot A sampling unit of some defined area or volume.

Population trend An average change over time in magnitude and direction of some population parameter within a specified area.

Presence Actual occurrence of an individual at a site (although it may not be detected).

Precision The degree of spread in estimates generated from repeated samples. Variance, standard deviation and standard error are all measures of precision.

Probability distribution The probability structure generated from all possible values of some random variable.

Random sample A collection of sampling units chosen based on some known chance of selection. Random selection allows some probability or degree of certainty to be attached to resulting sample estimates in order to assess their usefulness.

Robustness The ability of an estimator to produce estimates with relatively small bias even if the underlying assumptions are not met.

Sample A group of sampling units selected during a survey.

Sampled population All elements associated with sampling units listed or mapped within the sampling frame.

Sampling design Protocol for obtaining parameter estimates for a sampled population. The purpose of a sampling design is to make inferences about the sampled population, usually in conjunction with an observational study. In monitoring, a spatial sampling design specifies a means of selecting spatial sampling units that permits inference about the sampled population. A sampling design may also be used in an experimental approach for obtaining estimates of differences in parameters associated with treatment groups.

Sampling distribution The probability distribution of a sample estimate based on probability of occurrence of estimates generated by all possible samples of a given size.

Sampling unit A unique set usually of one or more elements. In spatial or area sampling a sampling unit (e.g., plot of ground) may not contain any elements.

Sampling variation A measure of the degree of spread whose source is solely from random chance associated with the selection procedure (i.e., among-unit variation) and/or counting protocol (i.e., enumeration variation).

Site occupancy Probability that a site is occupied by a species at a particular time or for a finite area, the proportion of sites occupied by a species at any time.

Spatial distribution A geographical range of locations or areas occupied by a species.

Standard deviation Square root of variance of individual items in a probability distribution. In this case, 'distribution' refers to either the true or population distribution, such as the distribution of all plot abundances, N_i (called the population standard deviation), or the distribution within a single sample, such as the distribution of items within a single plot sample (called the sample standard deviation or just 'standard deviation').

Standard error Square root of variance; the standard deviation of a sampling distribution of sample estimates. The 'population standard error' describes this measure for a sampling distribution of all possible sample estimates. An estimator of this quantity, called the sample standard error (or just 'standard error'), may be obtained from a single sample and, for infinite populations, is equal to the sample standard deviation divided by the square root of sample size. The standard error is especially useful for computing a confidence interval for a parameter estimate.

Survey A count (usually incomplete) of individuals, objects or items within a specific area and time period.

Trend A change in average status of some quantity or attribute over a defined time period.

USFWS U.S. Fish & Wildlife Service.

Variance A measure of precision; average of squared differences between a set of values and the mean of the distribution of those values.

WWF Formerly the 'World Wildlife Fund', WWF is now known by simply by its acronym and is an international NGO with a mission 'to stop the degradation of the planet's natural environment and to build a future in which humans live in harmony with nature'.

WCS The Wildlife Conservation Society. An international NGO, founded in 1895, with a mission 'to save wildlife and wild places across the globe.'