Evaluating the Suitability of Wildlife Monitoring Methodology for Tropical Forest Conservation

A Comparison of Transect Surveys and Camera Trapping

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# Contents

List of Acronyms ........................................................................................................ iv  
Abstract .................................................................................................................... v  
Acknowledgements .................................................................................................... vi  

## 1. Introduction

1.1 Rationale for research ......................................................................................... 1  
   1.1.1 Sustainable Forest Management ................................................................. 2  
   1.1.2 Wildlife monitoring .................................................................................... 3  
   1.1.3 Wildlife Wood Project ............................................................................. 4  
1.2 Aims and Objectives ......................................................................................... 5  
1.3 Thesis structure .................................................................................................. 6  

## 2. Background

2.1 Wildlife monitoring for conservation ............................................................... 7  
   2.1.1 Current issues affecting successful monitoring ........................................... 7  
2.2 Monitoring design ........................................................................................... 9  
   2.2.1 Monitoring methodology .......................................................................... 9  
2.3 Method selection ............................................................................................. 10  
   2.3.1 Method comparison studies .................................................................... 10  
   2.3.2 Comparing cost efficiency ...................................................................... 11  
2.4 The role of monitoring in forest conservation management ......................... 12  
2.5 Monitoring methods suitable for tropical forest conservation ...................... 14  
   2.5.1 Linear transects ....................................................................................... 14  
   2.5.2 Recce transects ....................................................................................... 15  
   2.5.3 Camera trapping ...................................................................................... 16  
   2.5.4 Comparison of methods ......................................................................... 17  

## 3. Methods

3.1 Data collection .................................................................................................... 18  
   3.1.1 Study site ................................................................................................. 18  
   3.1.2 Repeated linear transect surveys ............................................................... 18  
   3.1.3 Paired linear and recce transect survey ..................................................... 19  
   3.1.4 Camera trapping ...................................................................................... 21  
3.2 Data analysis ..................................................................................................... 23  
   3.2.1 Transect data ........................................................................................... 23  
   3.2.2 Camera trapping data ............................................................................... 25  
   3.2.3 Detection function .................................................................................. 27  
3.3 Costing ............................................................................................................... 28  
   3.3.1 Cost of precision ...................................................................................... 29  
   3.3.2 Cost projections ...................................................................................... 29  
3.4 Time and Effort requirements .......................................................................... 29  
3.5 Statistics ........................................................................................................... 30
4. Results

4.1 Correlation between relative abundance and precision

4.2 Optimum survey design

4.2.1 Transects

4.2.2 Camera trapping

4.2.3 Optimum camera number

4.3 Method comparison

4.3.1 Precision

4.3.1.1 Cost to generate specified precision

4.3.1.2 Precision for a specified budget

4.3.1.3 Time and Effort

4.3.1.4 Detection accuracy

4.3.2 Cost projection

4.4 Focal species comparison

4.5 Camera trapping costing

4.5.1 Effect of using rechargeable batteries

4.5.2 Effect of camera cost

5. Discussion

5.1 Correlation between relative abundance and precision

5.2 Optimum survey design

5.2.1 Transects

5.2.2 Camera trapping

5.2.2.1 Camera number

5.3 Method comparison

5.3.1 Cost

5.3.1.1 Reducing camera trapping cost

5.3.1.2 Rechargeable batteries

5.3.2 Time and Effort

5.3.3 Cost projection

5.3.4 Detection accuracy

5.3.5 Focal species

5.4 Method benefits and limitations

5.4.1 Data collection and analysis

5.4.2 Alternative outputs

5.5 Recommendations

5.5.1 Recommendations to the Wildlife Wood Project

5.5.2 Recommendations for wider use

5.6 Study limitations and Further work

Reference list
Appendices
Appendix A ........................................................................................................ 69
Appendix B ........................................................................................................ 71
Appendix C ........................................................................................................ 73
Appendix D ........................................................................................................ 74
Appendix E ........................................................................................................ 77

Tables and Figures

Table 3.1 - Cost per item for equipment per monitoring method .................... 28
Table 4.1 – List of species acronyms ................................................................. 31
Table 4.2 – Spearman’s Rank CC results .......................................................... 31
Table 4.3 - Effect of transect survey design on precision of RA ....................... 32
Table 4.4 – Effect of camera trapping survey design on precision of RA .......... 34

Figure 3.1 - Biomonitoring surveys in SFID and Pallisco ................................. 19
Figure 3.2 - Paired linear and recce transects in Pallisco ................................. 20
Figure 3.3 - Camera trapping placement in SFID and Pallisco ....................... 22
Figure 4.1 – Optimum transect survey design ............................................... 33
Figure 4.2 – Optimum number of cameras per survey ..................................... 35
Figure 4.3 – Cost to obtain RA estimates with a CV of 25% ............................ 37
Figure 4.4 – CV obtainable for $10,000 budget .............................................. 38
Figure 4.5 – Time (days) required for RA estimates with CV of 25% ............... 39
Figure 4.6 – Effort (man-days) required for RA estimates with CV of 25% ...... 40
Figure 4.7 – Effect of detection accuracy on CV estimates ............................. 41
Figure 4.8 - Cost projection over 20 repeat surveys ....................................... 43
Figure 4.9 - Survey length required for different combinations of focal species 44
Figure 4.10 - Effect of rechargeable batteries on camera trapping cost .......... 45
Figure 4.11 – Effect of reducing camera cost on method suitability ............... 46
List of Acronyms

CBD – Convention on Biological Diversity
CV – Coefficient of Variation
FSC – Forest Stewardship Council
RA – Relative abundance
REDD+ – Reducing Emissions from Deforestation and Forest Degradation
RIL – Reduced Impact Logging
SFM – Sustainable Forest Management
WWP – Wildlife Wood Project
WWP-Cam – Wildlife Wood Project, Cameroon

Species acronyms
BAD – Bay duiker
BLD – Blue duiker
C - Chimpanzee
E - Elephant
G - Gorilla
PD – Peter’s duiker
RRH – Red river hog
S - Sitatunga
YBD – Yellow-backed duiker
Abstract

With over a third of the world’s remaining natural forest currently designated as timber concessions it is essential that the management of these areas is appropriate to allow survival of the indigenous wildlife. The Wildlife Wood Project in Cameroon, is a current example of an initiative working to achieve this through the promotion of sustainable management practices. A key component of the project is to initiate cost effective monitoring programmes in order to track the impact of logging on wildlife in two timber concessions. To determine the most appropriate methodology, the technique of resampling with replacement was used to compare the performance (within the two concessions) of three different monitoring methods (linear transects, recce transects and camera trapping) over a variety of different survey designs. Relative abundance estimates, with a coefficient of variation of 25%, were calculated for nine focal species. Transect surveys were shown to be most effective in terms of survey cost, survey length in days, and man-effort required to carry out the necessary data collection. The use of this method and specifically recce transects, was strongly recommended over camera trapping. Alternative outputs that can be generated through camera trapping surveys were considered to be highly advantageous to wildlife monitoring schemes, and where budgets permit, the combination of this method with transects would be an optimal solution. The results presented in this study demonstrate the importance of considering the precision of monitoring outputs, in terms of survey cost and time, to ensure that the data collected (within the constraints of the monitoring programme) has the highest power to detect change in population size over time. There is potential for wider application of the conclusions drawn from this analysis to assist in the design of cost effective monitoring schemes which can be adopted as a part of effective sustainable forest management practices.

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1. Introduction

1.1 Rationale for research
The area of the planet covered with undisturbed natural forest habitat has halved in the past century (Soule & Sanjayan, 1998) despite a growing concern for the loss of biodiversity, and international efforts to prevent further reductions (Bawa & Seidler, 1998). The current rate of deforestation stands at 0.8% per annum, equating to a yearly loss of approximately 15 million hectares of forest (Whitmore, 1999). In recent years the logging industry has become the most intensive land use in many forested areas, with over 30% of remaining forest worldwide (Laporte et al., 2007), and over 40% of forest in Africa (Clark et al., 2009) now designated as logging concessions. In contrast, just 12% of remaining forest is currently included in protected areas (Laporte et al., 2007).

Protected areas of forest are an essential tool for biodiversity conservation (Putz & Romero, 2001) but their limited size, number and fragmented distribution mean they alone are inadequate to preserve natural habitats (Sist et al., 2008) and provide protection to wide ranging species (Dudley et al., 2005). The majority of protected areas in developing countries are threatened by poor governance and corruption (Clark et al., 2009), and can often be nothing more than “paper parks” due to difficulties in enforcing protection laws (Soule & Sanjayan, 1998). Additionally, areas designated for protection are not always the most biodiverse, but instead the least suitable for exploitation (Liberia FDA, 2007) making them less than ideal areas to concentrate conservation action on. Focusing solely on expanding the area covered by protected areas is therefore not a practical solution; alternative conservation action is also necessary to ensure that biodiversity outside protected areas is not neglected (Putz & Romero, 2001). Due to their large size and varied habitat, there is potential for production forests to complement existing protected areas and enlarge the conservation estate (Putz & Romero, 2001, Clark et al., 2009).
1.1.1 Sustainable Forest Management

Sustainable forest management (SFM) ‘aims to ensure that the goods and services derived from the forest meet present-day needs, while at the same time securing their continued availability and contribution for future generations’ (FAO, 2010). If implemented effectively within production forests SFM could be the key conservation strategy needed to protect the millions of hectares of forest lying outside protected areas (Putz & Romero, 2001). In addition, the implementation of reduced impact logging (RIL) practices, whereby the timber harvested is carefully controlled and carried out in ways designed to minimise deleterious impacts will be vital (Putz et al., 2008). Production forests should not be considered as a replacement for protected areas, but the application of good management within these areas will be just as, if not more important than the establishment of new protected areas for the conservation of forest biodiversity (Clark et al., 2009).

Forest certification is a process by which forest management is assessed in accordance with pre-determined standards, thereby providing proof that timber produced from the named forest adheres to required criteria (Rametsteiner & Simula, 2003). It has the potential to become a significant factor in forest protection (Rice et al., 1997), promoting the move towards SFM and RIL practices (Durst et al., 2006) by providing access to environmentally sensitive markets (Gullison, 2003), enabling trade that otherwise would not have been permitted. In an ever increasingly environmentally conscious world, access to such markets will play a large part in timber trade. For example, currently over 80% of timber traded to the UK is from certified forests (UK Timber Federation, 2010).

Desire to gain certification has prompted noticeable changes in forest management in some parts of the tropics, and the total area of certified forest currently grows by several million hectares each year (Putz & Romero, 2001). However, the certification process can be costly, requiring implementation of new management practices, and the lack of certification in countries with high deforestation rates
suggests that in such areas it is yet to become a viable conservation strategy (Gullison, 2003).

1.1.2 Wildlife Monitoring

The recent interest in SFM has prompted the development of a range of initiatives to promote and encourage forest conservation, including forest certification and Reducing Emissions from Deforestation and Degradation (REDD+) schemes (ITTO/IUCN, 2009, Parker et al., 2009). To assist managers in the implementation of new management techniques, guides of ‘best practice’ have been produced (for example (Morgan & Sanz, 2007, Rayden, 2008). The importance of involving monitoring in SFM practices is acknowledged by the majority of these initiatives (Ebeling & Fehse, 2009, van Kuijk et al., 2009), but neither specific requirements for monitoring nor guidelines addressing optimal survey designs are provided.

Forest certification standards, used to define minimum levels of management (Gullison, 2003) will rarely include wildlife conservation (Bennett, 2001b). Wildlife monitoring is addressed in Principle 8.2c of the Forest Stewardship Council (FSC) certification standards, which simply states ‘data collection and research needs to monitor composition and observed changes in flora and fauna’ (FSC, 1996). The lack of specificity in certification criteria concerning monitoring requirements (Bennett, 2001a) together with the lack of guidance on how to select cost-effective monitoring programmes will likely lead to cheap, simple but imprecise methods such as presence/absence counts (Moore & Kendall, 2004) being used to monitor forests merely in order to meet criteria. Guidelines assisting the design of cost-effective efficient monitoring programmes together with recommendations for survey design are essential in order to enable wildlife managers to collect better quality data in order to monitor forest health to achieve the best outcomes possible within a limited budget.
1.1.3 Wildlife Wood Project

The Wildlife Wood Project (WWP), a programme established by the Zoological Society of London (ZSL) in collaboration with UK importer of African hardwood Timbmet, is currently working towards achieving conservation targets in Central and Western Africa. This area is widely acknowledged as being of great importance in global forest conservation (Laporte et al., 2007), and is fast becoming one of the most important timber producing regions worldwide (de Blas & Perez, 2008). The focus of the WWP currently lies with pilot projects taking place in Ghana and Cameroon. The main aim of these projects is to help timber companies secure a sustainable future for their concessions and the wildlife that inhabit them through the implementation of low-impact logging practices and the application of cost-effective forest management (Arnhem et al., 2009). This will be achieved through determining robust indicators for wildlife that can be monitored and used to inform concession management. An important aim of this project is to help timber companies better understand the negative impact logging can have upon forest ecosystems. This should help emphasize the importance of implementing sustainable forest management and encourage companies to work towards achieving forest certification.

The Wildlife Wood Project based in Cameroon (WWP-Cam) commenced in 2007, and is focused on working with two timber companies, SFID and Pallisco, to develop practical and efficient methods for managing and monitoring wildlife populations and to assess the extent of impact that logging practices are having on the forest (Arnhem et al., 2009). As partners, both timber companies made a commitment to sustainable forest management, and to achieving FSC certification for all their concessions. Pallisco were able to achieve this in 2008 (Arnhem et al., 2009). WWP-Cam is working to build in-house capacity for each of the timber companies to be able to independently run bio-monitoring programmes throughout their concessions. An important aspect of this has been training wildlife monitoring teams in the skills required, and running pilot surveys.
Both the timber companies involved in WWP-Cam have carried out wildlife monitoring surveys in timber concessions using three different monitoring methods: linear transects, recce transects and camera trapping. In order to move forward and achieve the WWP aim of establishing a scientifically rigorous, cost effective efficient wildlife monitoring system for timber companies to implement, guidelines are needed to assist the selection of the optimum method over a variety of situations. These recommendations will not only enable progression in the monitoring and management practices used within the two timber concessions currently at the focus of the WWP-Cam, but also allow expansion of the project to other timber concessions, first in Central and Western Africa, and ultimately further afield. There is currently a lack of guidance on the most cost-effective way to monitor wildlife, and so recommendations made are likely to have a wider application for use in other forest conservation incentives, such as monitoring as a component of a REDD+ scheme, as proof of effective SFM and for other extractive industries trying to reduce the impact they have upon the environment.

1.2 Aims and Objectives
The main aim of this thesis is to produce recommendations and guidelines to assist in the selection of the most appropriate method to achieve the objectives of a wildlife monitoring programme. In order to do this, data collected using the three monitoring methods piloted by WWP-Cam will be compared and contrasted to evaluate efficiency and cost effectiveness in relation to a range of different factors affecting survey design. In addition to recommendations on method selection, a further aim is to provide guidelines on the optimum survey design for each monitoring method, in order to maximise cost-effectiveness of each method in achieving the required level of precision of outputs. Precision is not often considered as an influential factor in method selection, but is important in order to ensure that monitoring outputs have the power to detect the required level of change in population (Walsh & White, 1999).
To achieve these aims, the objectives are to:

- Compare methods in terms of precision of relative abundance estimates for variable survey designs and ascertain the optimum survey design for each method
- Compare methods in terms of the cost, time and man-effort (in terms of man-days required) for a specified level of precision
- Compare the efficiency of the three methods in monitoring different combinations of focal species
- Compare the cost per method over multiple repeat surveys
- Take into consideration other, less quantifiable method qualities such as the range of potential outputs, the need for expertise in data collection and any alternative benefits.

1.3 Thesis structure

Chapter Two provides a background to the thesis, describing the need for monitoring, the strengths and weaknesses of different monitoring methods and reviews previous method comparison studies. The chapter concludes by exploring the requirements monitoring programmes to be suitable for use within timber concessions.

Chapter Three describes the techniques implemented during data collection, followed by a detailed description of the methods used to analyze the data.

Chapter Four presents the results of the study in sequential order.

Chapter Five discusses and places results in a wider context, presents guidelines for optimum method selection and survey design and concludes by discussing the limitations of the study and making recommendations for future work and application of the results presented.
2. Background

2.1 Wildlife monitoring for conservation

In order to tackle primary conservation challenges, such as the effective design of wildlife reserves, initiation of sustainable management practices, control of extractive industries and initiation of Reducing Emissions from Deforestation and Degradation (REDD+) mechanisms, it is important to have a prior appreciation of the biodiversity of an area. There is currently a critical shortage in this much-needed data (Gardner et al., 2008), and as a result attempts to overcome these challenges are less effective. There is a growing need for biological assessments (Boddicker et al., 2002) and proven methodological approaches to address the lack of essential baseline data requisite for managing biodiversity (Rodriguez, 2003).

The purpose of wildlife monitoring programmes is to generate such data, to provide support to management decisions (Danielsen et al., 2007). These data will assist strategy planning (Sorensen et al., 2002), enable managers to set limits of acceptable change for both species populations and habitats (Walsh & White, 1999, Plumptre, 2000, Roberts et al., 2007) and identify priority areas or species for conservation action (Brashares & Sam, 2005, Danielsen et al., 2007). Data from long term monitoring programmes can be used to quantify the rate of change to populations over time (Buckland et al., 2005) in order to monitor the impact from anthropological activities (Balmford et al., 2003), such as logging and hunting and assess the effectiveness of current management practices in meeting objectives and mitigating threats (Stokes et al., 2010).

2.1.1 Current issues affecting successful monitoring

Wildlife monitoring can be described as a core conservation activity, and features as a major component in more than 50% of all conservation programmes contributing to reserve management schemes (Dajun et al., 2006). Yet despite the numerous advantages monitoring can bring and the fact that all countries party to the Convention on Biological Diversity (CBD) are obliged under Article 7b to monitor
biodiversity (Danielsen et al., 2003a), there is still a shortage of effective and sustainable long term monitoring programmes. This shortage can be attributed to the disproportionate number of monitoring schemes that are carried out in the developed world in comparison to the number of species present (Green et al., 2005).

The lack of monitoring in the developing world can generally be attributed to a lack of resources, both financial and otherwise (Danielsen et al., 2007) and a lack of capacity in terms of staff education and expertise, to be able to design effective monitoring programmes independently (Gardner, 2010). Shortages in funding create a barrier against the implementation of monitoring programmes (Brashares & Sam, 2005) and are almost always the limiting factor in their design (Gardner, 2010). Together with financial constraints, there is also a lack of knowledge on how to effectively budget the funds available (Gardner, 2010). Large amounts of money are wasted on over investment into ineffective monitoring schemes (Whitten & Balmford, 2006), and simply increasing funds available is likely to exacerbate this, rather than effectively address the issue at hand.

A key element of successful monitoring is having the means to maintain it over time (Brashares & Sam, 2005). Monitoring schemes are not often designed in a sustainable manner and therefore may collapse as soon as funding runs out (Danielsen et al., 2003a). To remedy this it is vital that together with the ecological needs of wildlife, the economic needs of people are taken into consideration (Stokes et al., 2010). Failure to successfully involve local communities will impact on the success of monitoring programmes (Yoccoz et al., 2003). Participatory monitoring schemes designed to involve non-scientists in data collection have proven to be beneficial particularly when used alongside additional professional monitoring (Danielsen et al., 2003b), by both assisting conservation action and in reducing the overall costs of monitoring (Danielsen et al., 2007).
2.2 Monitoring design

The design of an effective monitoring programme must start with the formation of clear precise project aims and objectives (Yoccoz, 2001). Survey specifics, such as study duration, complexity of data collection and the extent of spatial coverage will all be dependent upon the objectives of the study (Pollock et al., 2002), but are constrained by resource limitations. It is essential to determine whether a monitoring programme will be able to effectively deliver and achieve all objectives within resource constraints (Gardner, 2010). When managing a monitoring programme with a low budget, intelligent use of resources and creative survey design will be essential in order to avoid the need to sacrifice important aspects of data collection (Rodriguez, 2003). Failure to do so can result in monitoring that will generate data requiring a complex, and often flawed analysis (Buckland et al., 2005) compromising the integrity of results and adversely effecting the management inferences that can be drawn from them (Brashares & Sam, 2005).

During the design of a monitoring programme tradeoffs between scientific ideals and practicalities of financial and time budgets are likely (Brashares & Sam, 2005), and finding a middle ground on which to compromise will be problematic without clearly defined objectives. It is important to balance priorities for monitoring with their associated costs (Brashares & Sam, 2005, Gardner et al., 2008), and to carry out systematic conservation planning to ensure method selection and survey design are optimum in order to reduce cost as far as possible without compromising precision of outputs (Naidoo et al., 2006).

2.2.1 Monitoring methodology

Population monitoring programmes can be highly variable in nature, ranging from simple surveys, through complex studies focusing on a single species to an entire community, based at a single site, or spread over a continent or even worldwide (Marsh & Trenham, 2008). The techniques used for monitoring are also highly variable, but commonly focus on collecting data on either direct or indirect signs of one or more focal species (Lurz et al., 2008), and using the data to make inferences on the abundance of study species (Bennun et al., 2004). Improvements in
technology have over time increased the complexity of data collected (Silveira et al., 2003), but at the same time have reduced the effort required to do so. For example, camera traps can now be used to carry out capture-recapture analysis on species with distinguishable individuals, removing the need to capture and mark individuals manually (Pollock et al., 2002). It is important to remember the suitability and effectiveness of any method will always be relative to specific aims and resource constraints of an individual survey (Gaidet-Drapier et al., 2006).

2.3 Method Selection

The selection of a method to use for monitoring is a critical decision, one that can have a large influence over the accuracy and comprehensiveness of research outcomes (Garden et al., 2007). Exploring the balance between positive and negative characteristics of all suitable methods in relation to specific survey constraints is vital in order to ascertain the most beneficial technique to use. The budget available for a monitoring programme can have a large influence over method selection (Silveira et al., 2003, Joseph et al., 2006, Garden et al., 2007); the most cost-effective method to fulfil required objectives is likely to change with an increase in financial resources. The importance of this is demonstrated through the study carried out by Joseph et al (2006) comparing the effectiveness of presence/absence and abundance data. The authors concluded that when constrained by a limited budget the most cost-effective data was derived from presence/absence techniques, despite common perception that abundance data would always be better.

2.3.1 Method comparison studies

The decision making process for method selection remains rarely studied (Joseph et al., 2006) and the current lack of an effective theoretical framework to refer to during method selection and planning can present a major barrier to implementing cost-effective monitoring programmes (Joseph et al, 2006, Gardner et al., 2008).

contemporary methods of estimating density for group living species are compared against selection criteria to determine the most efficient. The study concludes that no single method was optimum, but all were different according to the situation, highlighting the need for a guide to assist in the adoption of an appropriate method in a variety of different situations. Marshall et al (2008) also conclude that in situations where populations are low, or information on detectability is unavailable, alternative monitoring methods to line transects are required to optimise data collection.

An additional method comparison, carried out by Franco et al (2007) was also unable to come to one conclusion concerning optimum method choice. The study, which compared the use of line transects and radio-telemetry for monitoring lesser kestrels (*Falco naumanni*), used a sub-sampling technique to determine the relationship between the sampling intensity, and quality of outputs generated (Franco et al., 2007). The use of resampling is a highly effective way of identifying cost effective sampling designs (Gardner, 2010), enabling the accuracy of estimates, their precision and hence their usefulness to be compared for increasingly large datasets from each of the two methods. The study concluded that telemetry was the more expensive and time consuming method to implement, and so in situations of strict cost or time constraints transects were recommended. However, the authors stressed that the advantages presented by telemetry, such as the potential to cover a much wider area, needed to be evaluated in relation to the aims and resource limitations of each monitoring programme, to determine whether the additional cost and time could be justified on a study-specific basis. The results from this comparison are specific to the study in question, but are also able to be extrapolated and applied in different situations to aid method selection (Franco et al., 2007).

### 2.3.2 Comparing cost-efficiency

Despite being the most sensible factor to use for deciding method selection, the lack of information on the costing of different techniques means considering the cost-effectiveness of different methods is difficult and therefore infrequently explored (Gaidet-Drapier et al., 2006, Franco et al., 2007). The benefit of incorporating cost
effectiveness in method selection is shown in a study into the most appropriate methodology to use when detecting occurrence of mammal and reptile species (Garden et al., 2007). The study compares six different techniques in terms of the number of species, and the number of captures it can achieve for a specified cost, revealing the most cost-effective method for each factor (Garden et al., 2007). This study also demonstrated that the most cost-effective method is likely to alter for species of differing body size (Garden et al., 2007), an important factor to consider when determining the most effective method to use for a survey monitoring multiple species.

2.4 The role of monitoring in forest conservation management

The implementation of effective forest conservation management is seriously constrained by the lack of information available on forest species composition – in some areas even the main species for a vegetation type are unknown (Rametsteiner & Simula, 2003). There is great potential for effective monitoring programmes to provide much needed data on the biodiversity present and contribute to forest protection (Gardner, 2010) and new initiatives such as REDD+ mechanisms (Parker et al., 2009), certification schemes (FSC, 1996) and the control of timber industries (ITTO/IUCN, 2009). The ability to successfully minimise costs without affecting precision of results will not only help monitoring schemes to achieve long term sustainability, but may also help to encourage wildlife managers to initiate monitoring as part of sustainable forest management.

The effects of logging are too inconsistent between species to allow overall conclusions to be drawn concerning the impact it has on forest biodiversity (Guo et al., 2008, Bicknell & Peres, 2010). Forest certification schemes can be used to reduce any negative impacts logging may have and maximise the contribution it makes towards the management of production forests (Gullison, 2003). Certification can be used to ensure that forest management is complying with designated standards to work towards sustainability, and to provide a guarantee that timber is being correctly managed (Araujo et al., 2009). However, the cost of certification, both the process, and the indirect costs associated with implementation of new management
practices (such as monitoring schemes) can be significant for small logging companies (Ghazoul, 2001), and notably those in tropical regions (Gullison, 2003). Advantages to the logging companies must be seen to outweigh or at least match these disadvantages in order to ensure compliance.

To implement monitoring in production forests the methods used need to be accessible to reserve managers in terms of price, complexity and efficiency of data collection techniques, but also remain scientifically valid and able to generate data that is precise enough to detect adequate levels of population change. Analysing the type of monitoring is required is also essential in order to ascertain whether achieving the required objectives will be feasible and within budget or not (Milner-Gulland & Rowcliffe, 2007). Effective method selection is important in order to minimise any potential for bias in data collection, while considering efficiency and cost-effectiveness in relation to field conditions, time and resources available (Marshall et al., 2008). Allocating resources to programmes lacking direction will lead to a loss both in interest from wildlife managers, and of opportunity to initiate adequate management before forest use intensifies and causes irrecoverable damage (Gardner, 2010). It is vital to empower logging companies to take responsibility for the health of their forest. Monitoring methods need to be designed to be suitable for use by field teams with limited experience, while being able to collect data with adequate power to detect change. The need to simplify complex methodology in order to make it achievable and affordable can compromise the quality of data and the potential to inform management, rendering the monitoring useless (Milner-Gulland & Rowcliffe, 2007). It is therefore essential to have the capacity to select the correct monitoring methodology specific to the purpose of a study: in this case one that is simple to execute while able to generate precise estimates of species abundance within budget.
2.5 Monitoring methods suitable for tropical forest conservation

Monitoring programmes used to assist forest conservation schemes aiming to control logging, minimise the effects of development or provide protection to forests will commonly focus on terrestrial mammals. Monitoring will focus on surveying select indicator species in order to infer about the overall health of the forest (Schulte-Herbruggen & Davies, 2006). Some species found in tropical forests will be sufficiently abundant to allow direct counting of individuals in order to estimate population size, but others will require population and abundance estimates to be made using signs of presence (Dajun et al., 2006). Survey walks in the form of linear and recce transects, and camera trapping surveys are commonly used to monitor forest mammals, due to the potential for a wide spatial coverage with small survey teams (Bennun et al., 2004).

2.5.1 Linear Transects

Linear transects are a simple and efficient way to collect monitoring data (Plumptre & Cox, 2006) generating a measure of encounter rate for each species studied (Marshall et al., 2008) which can be used to describe populations through estimates of either relative abundance or absolute density (Plumptre & Cox, 2006). The type of data collected using linear transects can be selected to suit the study species; direct observations of individuals can be recorded for abundant or bold species (Silveira et al., 2003), and indirect signs such as tracks, dung and nests, are often more appropriate for elusive, rare and nocturnal species (Boddicker et al., 2002). The difficulties in estimating production and decay rates of indirect signs such as dung or nests make the accuracy of absolute abundance derived from indirect observation estimates questionable (Plumptre, 2000), and the benefit gained from doing so should be carefully considered in relation to project objectives. The ability to adapt methodology to suit focal species makes linear transects one of the most commonly appropriate methods to use for monitoring terrestrial mammal populations (Marshall et al., 2008). Linear transects can however be problematic for monitoring rare species, as poor encounter rates can lead to sample sizes not large enough for data analysis (Bennun et al., 2004). The restriction on following a precise path can make surveying in difficult terrain problematic (Hiby & Krishna, 2001);
clearing a pathway through dense vegetation can become highly labour intensive and in some cases prohibit data collection (Walsh & White, 1999). However, the need to do so does ensure that a true representation of all habitat types is covered in a survey (Bennun et al., 2004). The reliability of transect data can be affected by observer bias (Chen, 1998, Rovero et al., 2006, Fitzpatrick et al., 2009) notably so if data is collected by inexperienced or inadequately trained observers (Azlan & Sharma, 2006). However, in general, linear transects are an effective, rapid and systematic method of monitoring terrestrial mammals (van Lavieren & Wich, 2010).

2.5.2 Recce Transects
The methodology for the implementation of reconnaissance, or recce, transects was developed from linear transects, and the process of data collection ultimately remains the same (Bennun et al., 2004). Recce transect lines do not follow a precise path but take the path of least resistance, avoiding patches of difficult terrain while heading in a specified general direction (Walsh & White, 1999). This modification enables transects to be completed faster (Plumptre & Cox, 2006), and with less manpower, thereby reducing the costs of monitoring (Walsh & White, 1999). The relaxation of path-restrictions also reduces the damage to the forest incurred through the cutting of a direct transect path (Plumptre & Cox, 2006). As with linear transects, data can be collected on both direct observations of species and indirect signs and the reliability of results is likely to be affected by the level of observer bias incurred in data collection (Rovero et al., 2006, Fitzpatrick et al., 2009). However, the avoidance of dense vegetation during transect lines will also generate bias in the habitat surveyed (Plumptre & Cox, 2006). It is important to consider the habitat preference of study species before implementing recce transects as the encounter rate for species favouring dense vegetation, such as gorillas (Sanz et al., 2007), may be negatively affected when using this method. A recent study (Arnhem & Fankem, 2010) has shown that this bias may be less of a concern than first anticipated but it remains essential to consider implications it is likely to have prior to data collection.
2.5.3 Camera Trapping

It is now also possible to monitor wildlife using automatic camera traps (Maffei & Noss, 2008). Motion-sensitive cameras are positioned in the area to be surveyed, and triggered to take photographs every time movement is detected (Bennun et al., 2004). The use of cameras to collect data is able to reduce observer bias (Rovero & Marshall, 2009) and once set up, cameras can be left in the field for weeks at a time, constantly collecting data with negligible environmental disturbance (Dajun et al., 2006). Logistical problems associated with faulty cameras, battery life and detection ranges are a possibility (Srbek-Araujo & Chiarello, 2007) and there is also potential for bias in species trapping rates towards trap-curious species compelled to return to the camera locations more frequently (Wegge et al., 2004). However, constant improvements to camera design and capabilities in the field continue to minimise the size and impact these limitations have upon data collection. Camera traps show a similar efficiency in collecting data on both diurnal and nocturnal species (Silveira et al., 2003), making them a viable way to carry out a thorough species inventory for an area (Treves et al., 2010). Species identification takes place following data collection, opposed to immediately in the field, which improves accuracy (Silveira et al., 2003) and enables the involvement of experts in identifying rare species (Tobler et al., 2008). The ability to collect data on rare or secretive species commonly hard to observe can lead to great improvements in understanding of community composition and species density (Azlan & Lading, 2006). The initial cost of camera trapping can be prohibitive (Srbek-Araujo & Chiarello, 2007), and despite being perceived to be the most efficient method in almost all environmental conditions (Silveira et al., 2003), the use of camera trapping is often restricted to larger budgets (Lyra-Jorge et al., 2008).

Capture-recapture analysis can be used to calculate absolute density estimates through camera trapping for those species with distinguishable individuals, such as tigers (Karanth and Nichols, 1998, Karanth et al., 2003) or jaguars (Silver et al., 2004), and for species for which this is not possible there is wide potential to use camera trap data to generate relative abundance estimates instead (Rovero & Marshall, 2009). Additional benefits such as the generation of a permanent record
of species presence (Lyra-Jorge et al., 2008) as a by-product of data collection make camera trapping a very powerful tool in the quest to improve standards of monitoring (O’Brien, 2008). The recent development of methods to use camera trapping data to generate estimates of absolute density without individual identification (Rowcliffe et al., 2008) will only improve this.

### 2.5.4 Comparison of methods

The recent development of camera trapping techniques to monitor wildlife has prompted a number of studies comparing the benefits of this new method with more established transect methodology. Trolle et al (2007) carried out such a study, comparing the use of both camera trapping and linear transects to study the abundance of Brazilian tapirs (*Tapirus terrestris*) in Brazil. Overall, the authors recommend the use of camera trapping, due to its ability to overcome the limitations currently restricting the potential of linear transect data, namely the capability to efficiently collect data on nocturnal and elusive species (Trolle & Kery, 2003). Two additional studies comparing the effectiveness of camera trapping to alternative monitoring methods (Silveira et al., 2003 and Lyra-Jorge et al., 2008) describe the benefits of this method in relation to accurate species identification and efficiency of detection of rare and nocturnal species, and recommend its use in monitoring programmes. All three studies state the main limitations of camera trapping as the cost and maintenance of all necessary equipment, but the restriction these costs are likely to place on the suitability of this method, in particular for implementation in developing countries where the current need for monitoring programmes is highest, is not discussed.

There is a great need for an increase in the number of method comparison studies, in order to improve understanding of which techniques are most suitable to use in different situations (Franco et al., 2007). More studies will improve the ease with which results can be generalised and applied to different situations, and improve the simplicity of method selection in the future (Franco et al., 2007).
3. Methods

3.1 Data collection

3.1.1 Study site
Data were collected in two different timber concessions, one (covering 14556 ha) assigned to the timber company SFID and the other (covering 769 ha) to the timber company Pallisco. Both timber concessions (from here on referred to by the name of the timber company) are located in the upper Nyong province of the Eastern region in Cameroon. To enable accurate comparisons, for each method data collected in each concession was kept separate during all analysis.

3.1.2 Repeated linear transect surveys
Linear transect surveys of terrestrial mammals were carried out in both SFID and Pallisco between October 2008 and October 2009. In both concessions biomonitoring stations, consisting of four 2km transects arranged in a cross formation, were set up throughout areas of forest experiencing different levels of logging impact: undisturbed forest, forest logged five years ago and forest currently being logged. A total of four biomonitoring stations were set up in SFID, and five in Pallisco, resulting in a total of 16 and 20 transects in the two concessions respectively (Figure 3.1). During the year long survey, repeat missions comprising of a visit to all transects in all biomonitoring stations were made to each concession. On each mission transects were walked at 1km per hour allowing two transects to be completed per day. Throughout the year, 11 missions were carried out in SFID, and nine in Pallisco.
3.1.3 Paired Linear and Recce transect survey

A second transect survey was carried out between April and July 2009. Data were collected in Pallisco to compare linear and reconnaissance (recce) methods of conducting transects. During the study, 30 pairs of transects were used, each one consisting of one recce and one linear transect. A starting point for each pair of transects was selected, and a 2km recce transect was walked in a designated direction corresponding to one of eight compass points. After 2km had been covered, a 2km linear transect would then be cut in a straight line back to the starting point. Due to taking a more direct route through the forest, on occasion this path extended past the starting point of the earlier recce transect (Figure 3.2).
All transects in both the repeated linear transect survey and paired linear/recce survey were walked by a group of five trackers, and data were collected using Cyber tracker® hand held Palm Top PDAs. Records were made for all direct and indirect signs of mammal species. During the repeated linear transect survey for observations of great ape nests, a measurement of the perpendicular distance from the transect line to each observation was also taken. With the exception of ape nests, observations of the same species estimated to be of the same age found within 15m of each other were considered a single record, due to the high likelihood that they were made by the same individual. Upon sighting an ape nest, care was taken to ensure that all nests within the same nest group were recorded. Both chimpanzees and gorillas commonly live in communities, and build sleeping nests with other conspecifics (Sanz et al., 2007). A nest group can be defined as a cluster of nests made by the same species and of the same age, with no more than 20m between each nest (Hashimoto, 1995, Tutin & Fernandez, 1984). The species of all mammal tracks and signs observed while on transects were identified in the field, requiring field staff to be experienced and competent enough to be able to do so without error.
3.1.4 Camera trapping.

Camera trapping was carried out in both SIFD and Pallisco, using 39 and 37 Stealth Cam ® cameras respectively, placed at 1km intervals in a grid formation throughout the study area (Figure 3.3). After 17 days of trapping cameras were revisited, and batteries changed. Cameras were then left to trap for a further 20 days before being collected. To allow comparison between cameras data collected were then truncated at the estimated minimum time to battery death, (a total of 36 days trapping for the SFID survey and 32 days in Pallisco). Following trapping, all photographs were analysed and species identified. When activated, cameras would take up to six photographs in very quick succession, potentially recording multiple photos of the same animal. To remove such replicates data was collated into ‘species events’ – the number of different times a species was trapped by a camera, rather than the actual number of photographs taken.
Figure 3.3 Maps detailing the placement of camera traps in a) Pallisco and b) SFID in relation to the placement of biomonitoring stations used for repeated linear transect surveys. Adapted from Arnhem et al. (2009).
3.2 Data analysis

During both transect and camera trapping surveys data were collected on all species encountered. For the purpose of this analysis, nine focal species were selected to focus on: gorilla (*Gorilla gorilla gorilla*), chimpanzee (*Pan troglodytes*), sitatunga (*Tragelaphus spekii*), elephant (*Loxodonta africana cyclotis*), red river hog (*Potamochoerus porcus*), yellow-backed duiker (*Cephalophus sylvicultor*), blue duiker (*Philantomba monticola*), bay duiker (*Cephalophus dorsalis*) and Peter’s duiker (*Cephalophus callipygus*).

Datasets from the following six surveys were used to compare the three monitoring methods:

- Repeated linear transect survey in SFID
- Repeated linear transect survey in Pallisco
- Linear transect dataset from Paired transect survey in Pallisco
- Recce transect dataset from Paired transect survey in Pallisco
- Camera trapping survey in SFID
- Camera trapping survey in Pallisco.

3.2.1 Transect data

Although data were collected on all indirect signs of species presence, only the most commonly recorded sign was used for each species during analysis. This resulted in records of tracks being used for all species bar the great apes, for which nests were the most frequently recorded indication of presence. For both species of great ape, observations of nests within a nest group were collated, so that one record was made per nest group, rather than per individual nest, to ensure independence of records.

Similarly for other species, many tracks made by a single individual could be seen within a short distance. Signs seen within 15m of another were ignored in the field, however this separation is not sufficient to ensure independence of records.

Estimates of home range from the literature were used to define home range diameter for each species (Appendix A), and this distance was used to define sampling units closer to being independent along transects. Any number of tracks recorded within this distance were refined as a single record. Each data set was then
collated to show the total counts of observations made of each of focal species per visit to each transect.

Data collected from each of the four transect surveys were then used to calculate estimates of relative abundance (RA – number of records per transect kilometre) for each of the nine focal species, using the following formula:

\[ N = \frac{\sum n_i}{\sum e_i} \]

Where \( n_i \) is the number of records on the \( i \)th transect, and \( e_i \) is the length of transect in kilometres, which was constant at 2km for all transects in all four surveys.

To determine the expected precision of RA estimates for varying sampling effort, resampling with replacement was used to simulate alternative datasets, following the process detailed below:

1. Transect were sampled from the list ‘\( t \)’ times with replacement, where \( t \) is equal to the number of transects to be surveyed.
2. For each of the selected transects the number of records per transect was then sampled ‘\( v \)’ times with replacement, where \( v \) is equal to the number of repeat visits to be made to each transect.
3. The simulated dataset was then used to calculate the RA of the focal species.
4. The above steps were repeated 1000 times (Efron & Tibshirani, 1993)
5. The precision of the RA estimates for given sampling effort was then summarised as the coefficient of variation (CV, standard deviation divided by the mean) across the 1000 resampled estimates:

\[ CV = 1000 \frac{\sigma(N_j)}{\sum(N_j)} \]

Where \( N_j \) is the RA estimate calculated for the \( j \)th resampled dataset, and \( \sigma \) denotes standard deviation.

This was repeated for each of the nine focal species for each of the four transect datasets. Appendix B details the code used for resampling data sets and subsequently calculating RA and CV estimates.
To ascertain whether maximising the number of transects, or number of repeat visits to each transect was the most efficient survey design, the expected precision for the following combinations of 100 transects was calculated:

- 100 transects repeated once
- 50 transects repeated twice
- 25 transects repeated four times
- 10 transects repeated ten times
- 5 transects repeated 20 times.

The minimum number of transects required to achieve a given CV value for each of the focal species was calculated by resampling the data as above for single visits to each transect (\(v=1\)), and finding the number of transects required to give the desired CV. Adequate level of precision was set at 25% (Stokes et al., 2010). Using the following formula:

\[
\Delta = 2.77(CV)
\]

achieving this level of precision can be shown to equate to the power to detect a 70% change in population size over time (Walsh & White, 1999).

### 3.2.2 Camera trapping data

A similar method was adopted to examine the effects on precision when using camera trapping. For this method two datasets were compared, one from the camera trapping survey in SFID, the other from Pallisco. No photographs were taken of elephants during either survey, reducing the number of focal species for this method to eight. Each dataset was collated into total counts of species events per species per camera. Estimates of RA (number of species events per camera day) were then calculated for each of the datasets, using the following formula

\[
N = \frac{\sum n_i}{\sum e_i}
\]

Where \(n_i\) is the number of species events taken by the \(i\)th camera, and \(e_i\) is the effort (number of trapping days), which was constant at 36 days for the survey carried out in SFID, and 32 for the survey in Pallisco.
To increase the number of species events recorded without increasing the number of camera traps used, cameras could either be left in the same position for a longer time, or be moved to new positions in the forest after a designated time period. To model the effects of multiple camera placements and varying placement length on the expected precision of relative abundance estimates, alternative datasets were simulated. The total species events per camera per day for \( e \) days, for \( p \) placements were resampled as above for each of the following survey designs of 14400 camera trapping days (using 40 cameras):

- 10 placements of 36 trapping days
- 5 placements of 72 trapping days
- 2 placements of 180 trapping days
- 1 placement of 360 days

The CV for RA estimates was then calculated for each species, and repeated for both sets of camera trapping data.

Together with the length of camera placements, the number of cameras used would affect the number of placements required to achieve a given CV value for RA estimates. The efficiency of varying camera number was compared, and an overall optimum number selected for use in further precision analysis.

The minimum number of camera placements required to achieve a given CV value when for each of the focal species was then calculated by resampling datasets as above for multiple placements (\( e=36 \)) using 20 cameras, and finding the number of placements required to give the desired CV. Again, a benchmark of 25% was used.

Additionally, for each dataset the expected precision was calculated for the actual survey design used in data collection. The RA and CV for focal species in each data set were then compared using Spearman’s Rank Correlation Coefficient, in order to compare the relationship between abundance and precision of estimates for different monitoring methods.
3.2.3 Detection function

Calculating the expected empirical CV does not alone provide a realistic estimate of precision. The inability to detect every sign or sighting for a species will also have an effect on the precision of RA estimates (Buckland et al., 2005). The perpendicular distances recorded for ape group sightings during the repeated line transect surveys in both SFID and Pallisco were used to provide an idea of the level of imprecision in detecting ape nests along the transect. Considering the two timber concessions are directly next to each other, the ability to detect nests in each was taken to be equal. Chimpanzees nest solely in trees, and gorillas will predominately nest on the ground (Sanz et al., 2007). As such, the detection of nests for either species was considered separately.

The perpendicular distances to all nest groups of each species observed from the transect line were compiled and the order resampled without replacement 10 times. DISTANCE software version 6.0 release 2 (Thomas et al., 2010) was used to calculate the CV of detection (CV_d) for increasing sample sizes for each list. Estimates of CV_d were averaged for each sample size, plotted as a function of sample size, and an exponential curve was fitted to the points. The graphs used to calculate CV_d equations can be found in Appendix C.

This equation was then able to be used to model the CV_d for any given sample size. To demonstrate the effect of involving detection in CV estimates, datasets for a survey consisting of 100 transects were resampled as detailed above for both Gorilla and Chimpanzee data collected in both concessions. For each of the 1000 repetitions, the CV_d was calculated using the total number of observations from the 100 resampled transects. The 1000 CV_d estimates were then averaged, to give one estimate and combined with the empirical CV (calculated as described above) using the delta method:

$$CV_e = \sqrt{(\mu(CV_d))^2 + (CV_e)^2}$$

Where $\mu(CV_d)$ = the mean detection CV, and $CV_e$ = empirical CV.
This was then compared to the empirical CV to demonstrate the effect that detection will have upon the precision of RA estimates, and repeated for both species using data collected in both concessions.

### 3.3 Costing

The cost of each monitoring method can be divided into two categories: fixed costs, and running costs. Fixed costs are predominately incurred at the start of implementing a monitoring scheme, are a one-off payment and include the purchase of equipment necessary to collect the data required. Running costs include the purchase of consumables and the payment of staff to carry out data collection.

<table>
<thead>
<tr>
<th>Method</th>
<th>Expenditure</th>
<th>Type</th>
<th>Singular price XAF</th>
<th>US$</th>
</tr>
</thead>
<tbody>
<tr>
<td>All Man-day</td>
<td>Running</td>
<td></td>
<td>3500</td>
<td>6.44</td>
</tr>
<tr>
<td>General equipment</td>
<td>Fixed</td>
<td></td>
<td>725</td>
<td></td>
</tr>
<tr>
<td>Transects</td>
<td>Cybertracker</td>
<td>Fixed</td>
<td>50</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Metal tape</td>
<td>Fixed</td>
<td>50000</td>
<td>0</td>
</tr>
<tr>
<td>Camera Trapping</td>
<td>Cameras</td>
<td>Fixed</td>
<td>185</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Delivery</td>
<td>Fixed</td>
<td>25</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Straps</td>
<td>Fixed</td>
<td>15</td>
<td></td>
</tr>
<tr>
<td></td>
<td>SD cards</td>
<td>Fixed</td>
<td>13</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Padlock</td>
<td>Fixed</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Batteries</td>
<td>Running</td>
<td>325</td>
<td>0.60</td>
</tr>
</tbody>
</table>

The pricing used to calculate the cost of methodology is listed in Table 3.1. Costs originally in local currency (CFA Francs, XAF) were converted to US dollars using the conversion rate of 1 XAF = 0.00184 US$ (Oanda, 2010).

The cost per man-day will include both the daily rate for a tracker, plus a daily food allowance. General equipment costs include all non-method specific equipment, including GPS, compass, machetes, boots, tents and other camping material. Both running and fixed costs will vary with survey design and length.
3.3.1 Cost of precision
Following the estimation of the minimum survey design necessary to achieve relative abundance estimates with a CV of either 25\% for each species, the cost of achieving this level of precision could then be calculated. Running costs for each method were multiplied to cover the level of effort required, and then added to the fixed start up costs which were adjusted accordingly for the survey design. For each focal species costs were calculated for the minimum survey design estimated from each dataset for each of the three monitoring methods.

3.3.2 Cost projections
The costs calculated represent the investment required in the first year of monitoring. However, for monitoring programmes to be successful they need to be a long term procedure. Following payment of the fixed start up costs in year one, the cost of carrying out monitoring in subsequent repeat surveys will decline. Using the estimated minimum survey designs, costs to carry out each method of monitoring were projected over 20 repeat surveys for each of the focal species.

3.4 Time and Effort requirements
Variations in survey design will also vary the total time and effort required to complete data collection. Using the estimated minimum survey design required to obtain RA estimates with a CV of 25\% the length of time (in days) that this would require was then calculated for each species using each data set from all three methods. Estimates of time for camera trapping included both the time taken to set up, move and collect cameras, and the time cameras were left trapping in the field. The effort required to achieve the designated level of precision (in man-days) was then calculated using the time taken to complete data collection and the number of staff required to do so. For camera trapping, effort calculations only involved the time staff are required in the field. Again, this was calculated for each species using all datasets from the three monitoring methods.
The estimates of cost, time and effort required to achieve the specified level of precision were then used to compare the cost-effectiveness of the three monitoring methods.

### 3.5 Statistics

Data was analysed according to the procedures described above unless otherwise stated using R© software version 2.11.1 (R Development Core Team, 2010). Where appropriate, statistical significance was set at $p<0.05$. 
4. Results

Throughout the following analysis, focal species will be referred to using the following acronyms.

**Table 4.1** Acronyms used for focal species in all tables and graphs.

<table>
<thead>
<tr>
<th>Acronym</th>
<th>Species</th>
</tr>
</thead>
<tbody>
<tr>
<td>BAD</td>
<td>Bay duiker</td>
</tr>
<tr>
<td>BLD</td>
<td>Blue duiker</td>
</tr>
<tr>
<td>C</td>
<td>Chimpanzee</td>
</tr>
<tr>
<td>E</td>
<td>Elephant</td>
</tr>
<tr>
<td>G</td>
<td>Gorilla</td>
</tr>
<tr>
<td>PD</td>
<td>Peter’s duiker</td>
</tr>
<tr>
<td>RRH</td>
<td>Red river hog</td>
</tr>
<tr>
<td>S</td>
<td>Sitatunga</td>
</tr>
<tr>
<td>YBD</td>
<td>Yellow-backed duiker</td>
</tr>
</tbody>
</table>

**4.1 Correlation between Relative Abundance and Precision**

Species abundance is known to affect detection rate, and therefore the number of observations per survey (Bennun et al., 2004). A low sample size can be linked to poor precision in RA (relative abundance) estimates. Spearman’s rank correlation coefficient was used to calculate the degree of correlation between the ranked values of RA and coefficient of variation (CV) for each species for each dataset. A strong negative correlation was found to be significant for all methods in all areas, but more highly significant for all transect datasets ($p<0.01$) than for camera trapping data ($p<0.05$). This demonstrates that the more abundant a species is, the lower the CV for RA estimates is likely to be. For all data sets, species ranked highest for RA were found to have the lowest ranked CV.

**Table 4.2** - Spearman’s rank CC between RA and CV of each species for different monitoring data sets (data reported to 4 decimal places)

<table>
<thead>
<tr>
<th>Data collection method</th>
<th>rho</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Linear transects – repeated survey SFID</td>
<td>-0.8667</td>
<td>0.0045</td>
</tr>
<tr>
<td>Linear transects – repeated survey Pallisco</td>
<td>-0.9333</td>
<td>0.0007</td>
</tr>
<tr>
<td>Linear transects – paired survey Pallisco</td>
<td>-0.8667</td>
<td>0.0045</td>
</tr>
<tr>
<td>Recce transect – paired survey Pallisco</td>
<td>-0.9</td>
<td>0.0020</td>
</tr>
<tr>
<td>Camera trapping - SFID</td>
<td>-0.7425</td>
<td>0.035</td>
</tr>
<tr>
<td>Camera trapping - Pallisco</td>
<td>-0.8571</td>
<td>0.0107</td>
</tr>
</tbody>
</table>
Actual RA estimates and accompanying CV for all species from all surveys can be found in Appendix D.

4.2 Optimum survey design

4.2.1 Transects

The optimum survey design for transect data collection (both linear and recce) was shown to be one visit to each of the 100 transects, as opposed to making repeat visits to a reduced number of transects. This was shown to be true for all four transect datasets (Table 4.3 presents an example set of results from the repeated linear transect survey in SFID, data for all other surveys are located in Appendix E). The CV of RA estimates for all species increases as the number of transects decreases and the number of repeat visits to each increases. For all focal species, making 20 repeat visits to five transects will generate RA estimates with a CV at least twice that of the RA estimates calculated from a survey visiting 100 transects once.

Table 4.3 Showing the effect of different combinations of transects and repeat visits on CV of RA estimates per species using data collected with linear transects in SFID.

<table>
<thead>
<tr>
<th>Survey design</th>
<th>Coefficient of Variation (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Transects</td>
<td>BAD</td>
</tr>
<tr>
<td>100 transects</td>
<td>1</td>
</tr>
<tr>
<td>50 transects</td>
<td>2</td>
</tr>
<tr>
<td>25 transects</td>
<td>4</td>
</tr>
<tr>
<td>10 transects</td>
<td>10</td>
</tr>
<tr>
<td>5 transects</td>
<td>20</td>
</tr>
</tbody>
</table>

Comparing the cost, time (in days) and man-effort (in man-days, equal to time in days multiplied by the number of field staff) required to carry out different combinations of transects and repeat visits to cover a total distance of 200km (100 2km transects) demonstrated that visiting each transect once per survey is also the most efficient survey design in order to minimize all three of these factors (Figure 4.1). Repeat visits to transects will require extra travelling time, increasing the time a survey takes to complete, which will have a consequential effect on both the man-effort required and overall survey cost.
For all survey designs, recce transects are seen to be the more cost effective and efficient (in terms of man-effort) method in comparison to linear transects. The difference between recce and linear transects can be attributed to the different number of field staff required for data collection, which effects both man-effort (Figure 4.1c) and survey cost (Figure 4.1a) due to an increase in the cost of payment due to the survey team members. A team of three trackers is required for recce transect surveys, but due to the need for pre-cut paths a team of five is required to carry out linear transects.

**Figure 4.1** Different combinations of surveys consisting of 100 transects, compared in terms of:

- a) cost
- b) time
- c) man-effort required for both linear and recce transect methods.

### 4.2.2 Camera trapping

To determine the most efficient camera trapping survey design to minimize both the CV for RA estimates and survey cost, the effect of increasing both the length of a camera placement (in days) and the number of placements per camera during a survey were compared. Implementing a camera trapping survey with multiple camera placements each lasting for one battery life was shown to be the optimum design in order to reduce the CV of RA estimates. However, the cost per survey will increase with the number of camera placements, due to the extra man-effort.
required to collect and re-distribute cameras to each placement (Table 4.4). Balancing the tradeoff between the number of placements and placement length will depend on the importance of attaining precise RA estimates, and restrictions on time and financial budgets specific to each survey.

Table 4.4 CV per placement/trapping time combination for all focal species using data collected in SFID (white cells) and Pallisco (shaded cells).

\( p = \) Number of placements, \( d = \) Placement length, in days.

<table>
<thead>
<tr>
<th>Survey design</th>
<th>CV per species for survey design (2dp)</th>
<th>COST (US$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>SFID</td>
<td></td>
<td></td>
</tr>
<tr>
<td>( p )</td>
<td>( d )</td>
<td>BAD</td>
</tr>
<tr>
<td>10</td>
<td>36</td>
<td>15.73</td>
</tr>
<tr>
<td>5</td>
<td>72</td>
<td>18.27</td>
</tr>
<tr>
<td>2</td>
<td>180</td>
<td>25.25</td>
</tr>
<tr>
<td>1</td>
<td>360</td>
<td>33.17</td>
</tr>
<tr>
<td>Pallisco</td>
<td></td>
<td></td>
</tr>
<tr>
<td>( p )</td>
<td>( d )</td>
<td>BAD</td>
</tr>
<tr>
<td>10</td>
<td>32</td>
<td>23.25</td>
</tr>
<tr>
<td>5</td>
<td>64</td>
<td>28.76</td>
</tr>
<tr>
<td>2</td>
<td>160</td>
<td>41.38</td>
</tr>
<tr>
<td>1</td>
<td>320</td>
<td>55.1</td>
</tr>
</tbody>
</table>

4.2.3 Optimum camera number

The number of cameras used in a survey will effect on the amount of time each camera needs to stay in the field in order to obtain the same overall total number of trap days (the number of days cameras are active in the field multiplied by the number of cameras used) for a study. To consider this the cost, time (days) and man-effort (man-days) required to achieve an overall total of 14400 trap days is considered for a range of numbers of cameras (Figure 4.2).

The optimum number of cameras to use in a survey will vary in order to minimize the cost, time or man-effort required to reach a specified number of trap-days. Increasing the number of cameras used in a survey will increase survey cost through to the need to purchase more cameras, and decrease survey time, as the number of placements required per camera in order to reach a set number of trap-days will decrease. The man-effort required per survey will be affected by the required number of placements per camera, as this will affect the number of visits made to the study site, and the number of field staff required to set up and collect cameras.
For less than 15 cameras, a team of four field staff are required. For 15 or more cameras, six field staff are required, to enable two groups of trackers to set up cameras simultaneously, thereby minimizing the time taken to position all cameras. The effort required when surveying with five cameras is greater than for any other number of cameras due to the large number of camera placements required (61), and therefore the large number of repeat visits to the field that are required in order to obtain 14400 trap days.

![Figure 4.2 Comparing the impact of different numbers of camera traps used in a survey on the a) cost (US$) b) time (days) c) man-effort (man-days) required to carry out survey over 14400 trap-days. Dark bar demonstrates the optimum number of cameras to use for each comparison factor.](image)

Essentially, the decision to balance the trade off between survey cost, time and man-effort required will be made in relation to specific survey constraints. For the remaining analysis in this thesis, 20 cameras will be used in all camera trapping survey simulations, being the optimum number to minimize effort, and a middle ground between the number required to minimize either the time or cost required.
4.3 Method comparison

4.3.1 Precision

4.3.1.1 Cost to generate specified precision

Using the survey designs optimum for obtaining the best precision in RA estimates for both linear and recce transects and camera trapping surveys, the minimum cost of obtaining a CV of 25% for each species was calculated using each of the six datasets.

Recce transects were shown to be the most cost-effective method to achieve RA estimates with a CV of 25%, followed closely by linear transects (Figure 4.3). RA estimates with the required level of precision for all focal species can be obtained for under $2500 using both transect methods. Camera trapping was shown to be the most expensive method of data collection, able to achieve the RA estimates with a CV of 25% for three of the focal species for under $5,000 using data collected in Pallisco, and not for any focal species using data from SFID.

There is a noticeable difference in the variation in costs between species for each monitoring method. For all transect surveys, the five species with the lowest monitoring costs are all four duiker species and red river hog. For camera trapping, the cheapest species for which to obtain RA estimates with the required level of precision varied with area, but included duiker species, as well as both species of great ape. Despite the differences in ranked costs of species between methods, the five cheapest species to monitor are also the five species estimated to be most abundant by each survey, demonstrating the link between precision of RA estimates and estimates of species relative abundance. This is supported by the results presented in Table 4.2.
The results from the repeated linear transect surveys carried out in SFID and Pallisco show little difference in the cost required to monitor each species between the areas. However, results from the camera trapping surveys carried out in either area have noticeable differences in the cost of achieving RA estimates with a CV of 25% for each species, particularly for Peter’s duiker, chimpanzee, red river hog and sitatunga. For all these species, camera trapping in Pallisco appears able to achieve RA estimates with the required precision for far less expense than when trapping in SFID.

![Graph showing the cost of generating RA estimates with a CV of 25% for different species and survey methods.](image)

**Figure 4.3** Minimum cost (US$) required to generate RA estimates with a CV of 25%. Y axis plotted on a logged scale. Horizontal lines drawn at survey cost of $2500, $5000, $10,000
4.3.1.2 Precision for a specified budget

For a budget of $10,000 estimates of RA for all focal species would have a much lower CV if generated using transects rather than camera trapping surveys (Figure 4.4). Comparing the two transect methods, it is evident that for all species, recce transects are able to generate RA estimates with a lower CV, as the lower number of field staff required for data collection will enable a higher number of transects to be carried out for the same budget. Transect surveys of this value are able to generate RA estimates with a CV lower than 10% for all focal species (with the exception of estimating the RA of elephants in SFID). In comparison, camera trapping surveys would be able to generate RA estimates with a CV of less than 10% for blue duikers, RA estimates for the remaining seven species would have CVs ranging between 11-32% (Figure 4.4).

![Chart showing CV for RA estimates generated by each monitoring method when survey costs are limited to $10,000.](image)

**Figure 4.4** CV for RA estimates generated by each monitoring method when survey costs are limited to $10,000. Horizontal line drawn at CV of 10%
4.3.1.3 **Time and Effort**

Transect surveys (using either linear or recce lines) were shown to be the more efficient monitoring method, able to generate RA estimates with a CV of 25% for all focal species in less than a year (Figure 4.5). Comparatively, only RA estimates for blue duikers would achieve this level of precision from camera trapping surveys lasting less than a year.

![Bar chart showing minimum time (days) required to generate RA estimates with a CV of 25%.

Y axis plotted on a logged scale.
Horizontal line drawn at survey time of one year.

**Figure 4.5** Minimum time (days) required to generate RA estimates with a CV of 25%.

The effort required to complete transect surveys to achieve RA estimates with CV of 25% is less than that required by camera trapping for seven of the nine focal species (Figure 4.6). The effort required to monitor chimpanzees in Pallisco by camera trapping is less than by linear transects, and similarly the effort required to monitor gorillas in SFID using camera trapping is less than that required by linear transects. However, for both species the most efficient method to use in order to generate RA estimates is recce transects.

For all focal species, the difference in the man-effort required by the different monitoring methods is less than the differences in the total time taken to complete data collection by each method. This is due to the length of time cameras are left out in the field trapping, not requiring any human effort to collect data.

Figure 4.6 Minimum man-effort (man-days) required to generate RA estimates with a CV of 25%. Y axis plotted on a logged scale.
4.3.1.4 Detection accuracy

Calculating the combined CV using both empirical and detection CV for RA estimates of great apes demonstrates the effect that imperfect detection has upon the precision of RA estimates. For a survey consisting of 100 x 2km transects, the empirical CV is estimated at between 15-20% for both species of great apes in both study areas, whereas including the detection function into precision estimates raises the CV by between 27-32%.

**Figure 4.7** Comparison of the empirical CV and combined empirical and detection CV calculated for RA estimates from surveys of 100 2km transects for Gorillas and Chimpanzees using data from repeated linear transect surveys in SFID and Pallisco.
4.3.2 Cost projection

Method costing can be broken down into one off start up and long term running costs (see Table 3.1). Once start-up costs have been covered during the first survey, continuing to monitor with repeat surveys will only require running costs to be paid. Projecting a running total of the cost of collecting data capable of producing RA estimates with a CV of 25% per survey over 20 repeats (Figure 4.8) demonstrates that over time camera trapping will not become the most cost-effective method to use. The cost of man-effort required to carry out camera trapping surveys exceeds that of transect surveys, thereby increasing the running total cost of a monitoring program at a faster rate.

Camera trapping is a more efficient method to use (in terms of man-effort) to monitor chimpanzees in Pallisco than linear transects (Figure 4.6). As a result of this, after five repeat surveys, camera trapping becomes more cost-effective method to survey this species in Pallisco than linear transects. Recce transects, however, remain the most cost effective method overall for surveying this species in both logging concessions.
Figure 4.8 Projection of the cost to generate RA estimates with a CV of 25% over 20 repeat surveys.
4.4 Focal species comparison

To determine which species have the greatest effect on survey cost for each method, the cost of achieving RA estimates with specified precision for all species in select groups using transects each method was explored (Figure 4.9). When collecting data using transects, minimum survey cost can be achieved by focusing monitoring on duiker species alone. For camera trapping, the most cost-effective group of species to achieve a minimum CV of 25% for RA estimates for was the great apes.

![Survey length required to achieve RA estimates with CV no greater than 25% for all species in given groups for different combinations of focal species: (a) for transect surveys and (b) for camera trapping.](image-url)

*Figure 4.9* Survey length required to achieve RA estimates with CV no greater than 25% for all species in given groups for different combinations of focal species:
- a) for transect surveys
- b) for camera trapping
4.5 Camera trapping costing

4.5.1 Effect of using rechargeable batteries

Purchasing batteries for camera trapping surveys will be an expensive addition to survey costs for this monitoring method. In an attempt to reduce this expense, the cost-effectiveness of investing in rechargeable batteries was explored. Comparing the survey cost for varying number of battery changes when using disposable and rechargeable batteries reveals how many battery changes are required to make a saving from using disposable ones (Figure 4.10). The use of rechargeable batteries will lead to a saving after five sets of disposable batteries have been used. As the majority of species require more than five camera placements (lasting one battery life each) in order to achieve RA estimates with the required levels of precision, it is likely that the use of rechargeable batteries will reduce costs of camera trapping within the first survey.

![Figure 4.10](image-url) Comparison of camera trapping survey cost for varying number of battery changes per survey using disposable and rechargeable batteries.
4.5.2 Effect of camera cost

Costs of technical equipment can be highly variable, and prices can fall dramatically over time. To consider the effect that a drop in cost of camera traps would have on overall survey cost of camera trapping, and therefore on overall suitability of this method, the cost of achieving RA estimates with a CV of 25% was modeled using varying prices of camera traps (Figure 4.11). In comparison to the cost of other methods, even a decrease in camera equipment cost by 90% does not make camera trapping the most cost-effective method choice, demonstrating the power of the influence that man-effort expenses have on the overall survey cost. Reducing the cost of camera equipment will however enable RA estimates with a CV of 25% for up to four focal species to be calculated for under $5000, which is not achievable for any species for full equipment costs. Additionally, a 90% reduction in camera cost will enable RA estimates to be calculated with a CV of 25% for Blue duikers for under $2,500.

**Survey method**
- **Linear – SFID repeated**
- **Linear – Pallisco repeated**
- **Recce – paired survey**
- **Camera Trapping – SFID**
- **Camera trapping – Pallisco**

**Figure 4.11** Cost of achieving RA estimates with CV 25%, camera trapping survey costs calculated using:

- a) Actual camera cost
- b) 50% reduction in camera cost
- c) 75% reduction in camera cost
- d) 90% reduction in camera cost

Horizontal lines drawn at survey cost of $2500, $5000 and $10,000.
5. Discussion

5.1 Correlation between Relative Abundance and Coefficient of Variation

For all monitoring methods, species encounter rate and the precision of relative abundance (RA) estimates were found to be significantly correlated (Table 4.2). The strong negative correlation demonstrates that the more abundant a species is, the lower the effort required to achieve a specified level of precision in RA estimates. In order to be able to have the power to detect the smallest changes in population size for the least effort, the most cost-effective species to include in monitoring programmes are therefore the most abundant ones (Walsh & White, 1999). However, it is likely that the species most in need of monitoring will be the rarest, rather than the most abundant (MacKenzie, 2005a). Consideration of the potential impact species abundance may have upon the precision of results, or on the length of survey that may be required, will be important to ensure that it is feasible to monitor certain species within the constraints of the study (Field et al., 2005).

5.2 Optimum survey design

5.2.1 Transects

The most efficient survey design to achieve precise RA estimates using transects is to visit each transect only once, avoiding repeat visits along the same transect lines (Table 4.3), demonstrating that large numbers of repeat visits can be of little value (Brashares & Sam, 2005, Field et al., 2005). This is true when using both linear and recce methods, and when considering efficiency in terms of cost, time (days) or man-days required (Figure 4.1). However, to carry out a linear transect survey with this design would require a new path to be cut before each and every transect could be walked. For a long term study not only will this be labour intensive and time consuming (Walsh & White, 1999), but could cause considerable disturbance and damage to the area being monitored (Plumptre & Cox, 2006). The extra effort required to continuously cut new transects has the potential to increase the man-effort required and time taken to carry out a survey, and therefore would have an impact on the overall cost of this method of monitoring. This was not taken into
consideration in Figure 4.1 due to the ambiguity of the size of the effect this may have, but it could lead to a reduction in the differences between survey designs. The increase in CV for RA estimates collected using a survey with 100 transects, compared to one with 50 transects repeated twice is minimal (ranging between <1 and 5% depending on transect method, area and species), and unlikely to markedly affect the power to detect a change in population. Reducing the number of transects by 50% for a survey could therefore be an adequate trade-off between increases in effort required and survey cost.

This is only likely to be an issue when surveying rare species, as the low encounter rate along the transect (Tobler et al., 2008) will mean a large number of transects and therefore an increase in effort will be required per survey. The reverse is true for abundant species, such as duikers. This issue is not relevant when using recce transects, as there is no requirement to cut a path through the forest (Fitzpatrick et al., 2009), and consequently the effort demanded is not affected by the number of repeat visits made to each transect during a survey. This illustrates a clear advantage that recce transects have over linear transects.

5.2.2 Camera trapping.

The most efficient survey design for camera trapping, in order to obtain precise estimates of RA, is to use cameras to trap for short periods of time (approximately one month) in a large number of different placements (Table 4.4). Conversely, the cheapest survey design for camera trapping is to trap for an extended period of time in one place, providing a clear trade off between efficiency of obtaining precision in RA estimates, and the cost required to do so. The resource constraints of any survey will often dictate the number of placements possible, but the importance of precision in the data collected also requires consideration, in order that confidence can be expressed in this data. Using fewer, longer placements per camera will cost less as revisiting cameras in the field to replace batteries will entail less effort than moving cameras to an alternative placement, but will take longer to achieve the same level of precision.
5.2.2.1 Camera number

Deciding upon the number of camera traps to use in a survey will be the end product of a three-way trade-off between the survey cost, the time taken (in days) to achieve the required survey length, and the man-effort required. To optimise cost, a low number of cameras are required; to optimise time taken, a large number. Effort can be minimised by using a medium number of cameras, thereby compromising between the number of cameras to distribute for each placement, and the number of placements required. The decision will depend entirely on the constraints and objectives of a specific survey, but in order to minimise all factors as far as possible, using a number of cameras that achieve a middle ground between the extremes favoured for cost and time will be optimum.

5.3 Method comparison

5.3.1 Cost

For all focal species, the most cost-effective method to generate RA estimates with a CV of 25% is transect surveys (Figure 4.3). There is little difference between the cost of the two transect methods; recce transects will cost less per transect due to the need for only three observers as opposed to the five needed for linear transects, but for some species a higher number of recce transects than linear transects are required to achieve the required CV for RA estimates.

Examination of the cost-effectiveness of different methods for a particular budget is perhaps a more realistic way to compare the relationship between cost and precision. Transects would clearly be the optimum method choice for a budget of $10,000 (Figure 4.4), and able to generate data that would have the power to detect a 30% or smaller change in a species population over time (Walsh & White, 1999). The ability to detect this level of change, particularly for rare species with small population sizes, will be of great advantage when monitoring the impact of extractive activities, such as logging, on wildlife. Sensitivity to population change is essential in order to determine the success of management initiatives and to enable impacts to species to be carefully monitored. The ability to detect smaller changes in
population size will enable a rapid response to any detected threats, preventing further damage.

5.3.1.1 Reducing camera trapping costs.
The high cost of camera trapping, commonly attributed to the need to purchase expensive cameras (Treves et al., 2010), in comparison to transect methods can deter the use of this method for wildlife monitoring (Trolle & Kery, 2003, Franco et al., 2007). Simulating the cost of achieving RA estimates with a CV of 25% for different camera costs revealed that even if costs were reduced to 10% of the current price, camera trapping would still not be the most cost effective method of obtaining precise RA estimates for any of the focal species (Figure 4.11). However, lower start up costs do reduce the difference in cost between camera trapping and transect surveys. Lower survey costs will make this monitoring method more widely accessible, thereby make investing in camera trapping for these reasons more justifiable.

5.3.1.2 Rechargeable batteries
A large number of batteries are required to sustain cameras in the field over long surveys. Continuously purchasing batteries is expensive, and so the feasibility of investing in rechargeable batteries was explored. The cost of investing in enough rechargeable batteries and battery chargers to cover all cameras is approximately equal to six sets of normal batteries per camera (budgetbatteries.co.uk, 2010). The use of rechargeable batteries will therefore become profitable during a camera trapping survey after six battery lives (Figure 4.10). To obtain RA estimates with a CV of 25% the majority of species will require a camera trapping survey of lasting longer than this, resulting in a profit being made within the first survey. The use of rechargeable batteries will not reduce camera trapping costs to the extent that it becomes a more cost effective method to use in comparison to transects, but it may make the method more affordable and therefore more suitable for use.
5.3.2 Time and Effort

Together with the cost of monitoring, it is important to consider the time a survey will take to complete, and the man-effort required to carry out data collection. Rapid surveys requiring little man-effort will not only be advantageous in terms of being low in cost, but will also require only a small time commitment per survey. For all focal species, transect surveys were shown to collect data required to generate RA estimates with a CV of 25% more efficiently than camera trapping (Figure 4.5).

The effort required for camera trapping is commonly considered to be low (Rovero & Marshall, 2009) due to the ability to leave cameras trapping in the field independently. However, the number of trap-days required to generate RA estimates with the required level of precision necessitates extra visits to the field in order to change batteries or move cameras to a new placement, increasing the level of man-effort required. Surveying chimpanzees in Pallisco, and Gorillas in SFID were the only instances where transects required more man-effort than camera trapping (Figure 4.6), demonstrating that differences between species will affect their detection (Harmsen et al., 2010).

5.3.3 Cost projection

For a monitoring programme to be sustainable, following the first survey, the running costs will need to remain on budget for each repeat survey. Projecting the cost of surveying the nine focal species to generate RA estimates with CV 25% for 20 repeat surveys revealed that even after start up costs are covered camera trapping will not become the most cost effective method to survey with (Figure 4.8). It is evident therefore that the high costs of camera trapping surveys are not solely down to large start up costs, but also the high level of effort and extended survey time needed to obtain adequate precision in RA estimates.

Examining the cost projection for chimpanzees in Pallisco further emphasizes this. The man-effort required to monitor chimpanzees in Pallisco was shown to be lower for camera trapping than linear transect surveys (Figure 4.6). Following initial high start-up costs the difference in cost between these two methods will decrease and
after six repeat surveys the cost of using linear transects to monitor chimpanzees is shown to exceed that of camera trapping surveys.

Maintenance costs were not factored into the cost projection, due to the uncertainty of the number and extent of problems that could occur, but are likely to have a greater effect on the cost of camera trapping over transect surveys, further increasing the difference in cost between methods. An indication of the likelihood of maintenance is shown in the hardiness of camera traps during data collection; after the 36 day survey in SFID 5% of cameras were already in need of maintenance and unable to be used in the Pallisco survey. Having to pay an additional 5% of the total start up costs every year would further reduce the likelihood that monitoring with camera trapping over multiple years would be suitable or sustainable.

5.3.4 Detection accuracy

No method of monitoring will have perfect detection (MacKenzie, 2005, Gardner, 2010). The large influence that inaccuracy in detecting all signs of species presence along a transect line can have on the precision of RA estimates (shown in Figure 4.7, (Buckland et al., 2005, Kery et al., 2009)), enforces the importance of not considering an estimate of empirical CV ($CV_e$) as an exact measure of precision. The size of the detection CV ($CV_d$) is directly affected by a species encounter rate; the larger the number of observations, the smaller the $CV_d$. Obtaining an adequate number of observations (estimated at a minimum of 40 for transect sampling (Bennun et al., 2004)) is therefore essential to minimise both $CV_e$ and $CV_d$, and ensure a high level of precision in RA estimates.

Obtaining as precise RA estimates as possible is essential in order to ensure that monitoring programmes have the power to detect changes in populations over time (Field et al., 2005). Unequivocally, transects are the optimum method to use for monitoring the focal species in this survey, minimising cost, survey time and man-effort required to generate RA estimates for all focal species with a CV of 25% (Figures 4.3, 4.5 and 4.6).
5.3.5 Focal species

The selection of species on which to focus monitoring will have a great impact the overall survey length in order to ensure precision RA estimates for all species (Figure 4.9) (MacKenzie, 2005b). The species requiring minimum effort (in terms of survey length, measured in transects or camera placements) varied with monitoring method, again demonstrating the difference in species detection between methods. Careful selection of the species to focus on will impact on the suitability of a method for monitoring in different situations.

Transects are shown to be highly efficient at monitoring common species such as duikers and red river hog, but survey cost and effort required are heavily affected by the inclusion of rarer focal species (Figure 4.9, (Field et al., 2005)). The importance of monitoring other rare species, such as elephant and sitatunga, needs to be carefully considered in relation to survey objectives, due to the effect these species will have on overall survey costs. The most expensive species to achieve RA estimates with CV of 25% using camera trapping are Peter's duiker and red river hog; species that are relatively common and therefore can be monitored for a fraction of the price using transect surveys (Figure 4.3). Chimpanzees and gorillas are some of least expensive species to monitor using camera trapping, and so are more suitable to be the focus of a camera trapping survey.

5.4 Method benefits and limitations

5.4.1 Data collection and analysis

Camera trapping methodology is highly simplified in comparison to transects, with less need for highly experienced field workers (Rovero & Marshall, 2009), and the use of cameras to collect data reduces potential for observer bias in data collection (Rovero & Marshall, 2009, Treves et al., 2010). Minimal training will be required prior to setting cameras up in the field, increasing potential for the involvement of untrained non-scientific people in field teams. However, reliance on technology to collect data can be problematic, and success subject to the performance of cameras (Dajun et al., 2006). Following data collection, photographs must then be analysed and species identified. Depending in the species studied and number of photographs
taken, this could be a long process, further extending the time taken to generate outputs using this method.

The implementation of transect surveys requires a far greater level of expertise, efficiency and ability will vary with training and experience (Sorensen et al., 2002). The need for a larger degree of training prior to data collection will therefore increase costs associated with this method (Fitzpatrick et al., 2009). Correct identification of species from indirect signs can be very difficult, especially for species with similar track or dung patterns (Dajun et al., 2006).

Indirect species observations are commonly recorded for species that are not detectable from the transect line, or in an attempt to increase the encounter rate and number of observations collected Stephens et al., 2006, Cromsigt et al., 2009). However, an increase in species detection rates using indirect observations can present a disadvantage, and lead to species observations being heavily pseudo replicated. The transect data used for analysis was found to not be spatially independent, and in order to correct this, sampling units along the transect line had to be redefined to be in line with the estimated home range of each species (Kendall et al., 1992, Lunt et al., 2007). Adjusting the data sets accordingly did not affect the overall trends seen in results – transects remained as the most cost effective method of data collection for all species. The use of indirect observations along transect lines is likely to have contributed to the vast differences seen between the effort requirements (both in survey length and man-days) of the two different methods.
5.4.2 Alternative outputs

Inferences on the cost, time and effort required to reach a specified level of precision are all drawn in relation to collecting data in order to calculate relative abundance. Both transect surveys and camera trapping have the potential to produce alternative outputs, the use of which may affect overall method suitability and therefore selection for use in some situations. It is therefore important to widen inferences made concerning the focal monitoring methods to include such outputs, and the advantages they may generate.

In addition to calculating RA estimates, absolute density estimates for certain focal species can be calculated from both linear transects and camera trapping (Dillon and Kelly, 2008, Marshall et al., 2008). Calculating absolute abundance from indirect observations can be very inaccurate, requiring the estimation of production and decay rates of signs (Matthews, 2004, Plumptre and Cox, 2006). As a result, the calculation of density estimates using transects will be far more advantageous when using direct sightings of species presence.

The use of camera trapping as a method for monitoring wildlife go beyond estimating species abundance; photographs taken during surveys can also be used to map species occupancy and home ranges (Tobler et al., 2008), create species activity budgets (Azlan & Sharma, 2006) and improve the extent of information on rare species (Rovero et al., 2005, De Luca & Rovero, 2006). The ability to detect rare and cryptic species that may never be recorded using transect surveys (Dajun et al., 2006), together with a similar efficiency in detection both diurnal and nocturnal species (Silveira et al., 2003) means camera trapping is an effective way to create a species inventory for a study area (Trolle, 2003, Rovero et al., 2009). Photographs taken during camera trapping surveys provide a proof of species existence (Dajun et al., 2006, Tobler et al., 2008), and can have many benefits following data analysis. Reporting results from surveys will receive more attention if combined with photographs of the species in question, raising awareness of conservation and monitoring programmes, and potentially generating income as a result, which could be fed back into reserve management and monitoring.
5.5 **Recommendations**

5.5.1 **Recommendations to the Wildlife Wood Project**

This study has demonstrated that transects are the optimum method of monitoring in order to minimise the cost, time and effort required while taking into consideration the precision of RA estimates. Camera trapping can be a valuable monitoring method, with advantages for measuring species abundance (O’Brien, 2008, Rowcliffe et al., 2008). However, the high start up costs, together with the high cost of carrying out long enough surveys to generate precise RA estimates make this method far less suitable for wildlife monitoring in timber concessions. Managing and sustaining camera trapping over extended periods will require a high level of commitment in terms of time and effort from logging companies, and as a result is unlikely to be successful.

In order to be succeed in encouraging timber companies to take responsibility for their forest and carry out monitoring on a regular basis, the recommended survey design needs to be simple, low cost and quick to implement. This is exactly what transect surveys present, and the opposite of what camera trapping surveys would entail. For the specific purpose of wildlife monitoring in timber concessions, it is therefore strongly believed that transects are the optimum method to use.

Throughout all the comparisons of precision, cost, time (days) and effort (man-days) linear and recce transects have remained very similar, due to the cost of man-days in Cameroon being very low. Were the daily rate for a tracker to increase, or if monitoring were being implemented in an area where man-day cost was higher, the difference between the two methods would be more noticeable, and recce transects would become more cost-effective in comparison to linear transects. Despite the potential for bias (Plumptre & Cox, 2006), recce transects are recommended overall as the optimum method to use, due to the reduction in damage to the forest, lower effort requirements and increased speed of surveying in comparison to linear transects (Plumptre & Cox, 2006).
Should adequate funds be available, the use of a combined recce and camera trapping survey is also recommended. During camera set up and collection, it should be possible to carry out transects when walking between each camera, at no extra expense. The results from the two methods are complimentary (Lyra-Jorge et al., 2008), enabling precise estimates of RA to be generated from transect data along with the added benefit of hard evidence to confirm the presence of all focal species (Rovero & Marshall, 2009). The integration of transects within camera trapping removes the need to keep cameras trapping for such extended periods, thereby controlling the time and cost of a survey, while enhancing the overall quality of data collected (Lyra-Jorge et al., 2008).

5.5.2 Recommendations for wider use

The optimum method of wildlife monitoring is likely to change with variations in budget, survey objectives and time constrains (Joseph et al., 2006). Camera trapping should be retained as a monitoring method option, and be kept in consideration when planning a monitoring programme. This method can offer a great deal of advantages, and, if the budget can cover the cost, is a viable method of monitoring wildlife (Franco et al., 2007).

The conclusions drawn from this study have potential to be used in other situations, but must be done so with care. It is highly likely that the minimum levels of effort recommended to obtain precise RA estimates will change not only for alternative focal species, but also when monitoring in different areas and habitats, and so should be used more as a relative comparison between methods than absolute values to adhere to. There is a clear need for an increase in studies comparing the cost effectiveness of different monitoring methods (Franco et al., 2007), and examining the precision of estimates of abundance they return. As stated by Franco et al (2007), the more comparison studies available the easier it is to generalise conclusions to assist decision making in any situation.
5.6 Study Limitations and Further work

The comparisons carried out during this study have concluded recce transects to be the most cost efficient monitoring method, and therefore recommends the use of this methodology where appropriate is recommended. However, the current inability to collect perpendicular distances from a recce transect line to species observations greatly restricts the level of analysis when using this method. Further work into potential ways to overcome this limitation and accurately collect distance data for all species observations would greatly improve the suitability of this method, and enable a wider implementation of the technique.

In order to improve the wider implications of the conclusions drawn in this thesis it would be advantageous to be able to generalise more on method performance over a range of species, and indifferent areas. Currently large differences, especially in camera trapping, can be seen in the minimum requirements to achieve RA estimates with a precision of CV 25% between the two timber concessions surveyed (Figures 4.3, 4.5, 4.6). The reasons for this are unknown, and as a result it is difficult to make general inferences concerning method performance in relation to focal species in other areas. Further work to ascertain the reasons for these differences may then highlight additional factors affecting the suitability of different methods in different areas.

The comparisons made between camera trapping and transect surveys are likely to have been affected by the different signs of species presence used by either method. Camera trapping presents data on direct species observations, whereas for this study transects had been used to record indirect signs of species presence. It is likely the number of observations per species will differ as a result. This could be a reason behind the differences in RA estimates for each species between the methods, and the differences in relative cost to monitor species with each method. It would be interesting to compare the methods when both are used to collect record of direct sightings, as this will undoubtedly increase the effort required from transect surveys, and may make the two methods more comparable.
References


Appendix A: Focal species home range estimates

To address the issue of non-spatial independence in indirect counts of species presence from transect surveys, estimates of home range were used to adjust sampling units along the transect to be closer to being independent. Home ranges were assumed to be circular, and therefore the estimate of home range diameter, rounded to the nearest 50m, was taken as the sampling unit, in an attempt to estimate the total distance covered in a day. Where home range data was unavailable, home ranges of other species were adapted in line with relative body size.

Table A1: Home range estimates and sampling units used for each focal species.

<table>
<thead>
<tr>
<th>Species</th>
<th>Home range</th>
<th>Sampling Unit</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bay duiker</td>
<td>70,000m (Bowland and Perrin, 1995)</td>
<td>300m</td>
</tr>
<tr>
<td>Blue duiker</td>
<td>7,000m (Bowland and Perrin, 1995)</td>
<td>100m</td>
</tr>
<tr>
<td>Chimpanzee</td>
<td>70,000m (Newton-Fisher, 2003)</td>
<td>300m</td>
</tr>
<tr>
<td>Elephant</td>
<td>310,000m (Shannon et al., 2006)</td>
<td>600m</td>
</tr>
<tr>
<td>Gorilla</td>
<td>150,000m (Doran-Sheehy &amp; Boesch, 2004)</td>
<td>450m</td>
</tr>
<tr>
<td>Peter’s duiker</td>
<td>70,000m (Bowland and Perrin, 1995)</td>
<td>300m</td>
</tr>
<tr>
<td>Red river hog</td>
<td>280,000m Adapted from information in Bowland and Perrin, 1995</td>
<td>600m</td>
</tr>
<tr>
<td>Sitatunga</td>
<td>10,000m Adapted from information in Bowland and Perrin, 1995 together with AWF.org, 2010</td>
<td>100m</td>
</tr>
<tr>
<td>Yellow-backed duiker</td>
<td>182,000m Adapted from information in Bowland and Perrin, 1995</td>
<td>500m</td>
</tr>
</tbody>
</table>
References


Appendix B: Example R code

An example of the R code used to resample datasets in order to calculate the coefficient of variation of relative abundance estimates for varying survey design.

data<-read.delim("linear independant.txt")
transect<-data$Transect
tmax<-15 # number of different transects in dataset
subsample <- function(trans,n,dat,index)
    { sample(dat[which(index==trans)], n, replace=T) }
CV <- function(dat)
    { sd (dat) / mean(dat) }
RA<-function (dat,n,e)
    { sum(dat)/(n*e) }

#start
obs<-data$BAD # species to focus on
eff<-2
m<-1
tsample<-sample(1:tmax,1,T)
res<-sapply(tsample,subsample,m,obs,transect)
RAres<-RA(res,1,eff)
allRA<-c(RAres)
allRA
for (samplecounter in 2:1000)
    {
        tsample<-sample(1:tmax,1,T)
        res<-sapply(tsample,subsample,m,obs,transect)
        RAres<-RA(res,1,eff)
        allRA<-c(allRA,RAres)
    }
RA.CV<-CV(allRA)
RA.CV
all.RA1<-c(RA.CV)
for (transectcounter in 2:500)
{
    tsample<-sample(1:tmax, transectcounter, T)
    res<-sapply(tsample, subsample, m, obs, transect)
    RAres<-RA(res, m*transectcounter, eff)
    allRA<-c(allRA, RAres)
}

for (samplecounter in 2:1000)
{
    tsample<-sample(1:tmax, transectcounter, T)
    res<-sapply(tsample, subsample, m, obs, transect)
    RAres<-RA(res, m*transectcounter, eff)
    allRA<-c(allRA, RAres)
}

RA.CV<-CV(allRA)
all.RA1<-c(all.RA1, RA.CV)
}

all.RA1
l.bad<-all.RA1
#end
Appendix C: Modeling detection function

Following estimation of the detection function coefficient of variation ($CV_d$) using DISTANCE analysis for varying sample size, the results were plotted and fitted with an exponential curve. The equation to this curve was then able to be used to estimate $CV_d$ for any given sample size.

**Figure C1** The effect of number of observations on the size of $CV_d$ for

a) Gorilla nest detection

b) Chimpanzee nest detection

Exponential curve line fitted to the points is shown in red, equation for the curve shown on each graph.
Appendix D: Relative abundance estimates

Relative abundance estimates were calculated for all focal species using the total number of observations collected during each monitoring survey. The coefficient of variation was then calculated for each estimate of relative abundance by resampling datasets 1000 times for the survey design used during each survey.

Relative abundance estimates for transect surveys are a measure of species observations per 1km of transect, and for camera trapping are a measure of species observations per camera per trap day.

Table D1. Relative abundance and coefficient of variation estimates from the repeated linear transect survey in SFID.

<table>
<thead>
<tr>
<th>Species</th>
<th>Relative abundance</th>
<th>Coefficient of variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bay duiker</td>
<td>2.63</td>
<td>4.35%</td>
</tr>
<tr>
<td>Blue duiker</td>
<td>7.52</td>
<td>3.57%</td>
</tr>
<tr>
<td>Chimpanzee</td>
<td>0.27</td>
<td>19.73%</td>
</tr>
<tr>
<td>Elephant</td>
<td>0.02</td>
<td>65.17%</td>
</tr>
<tr>
<td>Gorilla</td>
<td>0.18</td>
<td>17.80%</td>
</tr>
<tr>
<td>Peter's duiker</td>
<td>2.77</td>
<td>3.39%</td>
</tr>
<tr>
<td>Red river hog</td>
<td>1.28</td>
<td>8.31%</td>
</tr>
<tr>
<td>Sitatunga</td>
<td>0.15</td>
<td>33.38%</td>
</tr>
<tr>
<td>Yellow backed duiker</td>
<td>1.65</td>
<td>3.39%</td>
</tr>
</tbody>
</table>

Table D2. Relative abundance and coefficient of variation estimates from the repeated linear transect survey in Pallisco.

<table>
<thead>
<tr>
<th>Species</th>
<th>Relative abundance</th>
<th>Coefficient of variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bay duiker</td>
<td>2.01</td>
<td>6.32%</td>
</tr>
<tr>
<td>Blue duiker</td>
<td>4.42</td>
<td>5.17%</td>
</tr>
<tr>
<td>Chimpanzee</td>
<td>0.18</td>
<td>27.67%</td>
</tr>
<tr>
<td>Elephant</td>
<td>0.21</td>
<td>26.93%</td>
</tr>
<tr>
<td>Gorilla</td>
<td>0.14</td>
<td>26.27%</td>
</tr>
<tr>
<td>Peter’s duiker</td>
<td>2.34</td>
<td>4.89%</td>
</tr>
<tr>
<td>Red river hog</td>
<td>1.33</td>
<td>6.68%</td>
</tr>
<tr>
<td>Sitatunga</td>
<td>0.61</td>
<td>20.23%</td>
</tr>
<tr>
<td>Yellow backed duiker</td>
<td>1.10</td>
<td>7.10%</td>
</tr>
</tbody>
</table>
### Table D3. Relative abundance and coefficient of variation estimates from the paired linear transect survey in Pallisco.

<table>
<thead>
<tr>
<th>Species</th>
<th>Relative abundance</th>
<th>Coefficient of variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bay duiker</td>
<td>2.75</td>
<td>7.05%</td>
</tr>
<tr>
<td>Blue duiker</td>
<td>5.13</td>
<td>8.04%</td>
</tr>
<tr>
<td>Chimpanzee</td>
<td>0.30</td>
<td>38.64%</td>
</tr>
<tr>
<td>Elephant</td>
<td>0.28</td>
<td>25.36%</td>
</tr>
<tr>
<td>Gorilla</td>
<td>0.25</td>
<td>34.04%</td>
</tr>
<tr>
<td>Peter's duiker</td>
<td>3.05</td>
<td>5.07%</td>
</tr>
<tr>
<td>Red river hog</td>
<td>1.37</td>
<td>10.31%</td>
</tr>
<tr>
<td>Sitatunga</td>
<td>0.37</td>
<td>30.55%</td>
</tr>
<tr>
<td>Yellow backed duiker</td>
<td>1.17</td>
<td>11.24%</td>
</tr>
</tbody>
</table>

### Table D4. Relative abundance and coefficient of variation estimates from the paired recce transect survey in Pallisco.

<table>
<thead>
<tr>
<th>Species</th>
<th>Relative abundance</th>
<th>Coefficient of variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bay duiker</td>
<td>2.85</td>
<td>5.07%</td>
</tr>
<tr>
<td>Blue duiker</td>
<td>5.67</td>
<td>6.81%</td>
</tr>
<tr>
<td>Chimpanzee</td>
<td>0.28</td>
<td>39.24%</td>
</tr>
<tr>
<td>Elephant</td>
<td>0.37</td>
<td>25.65%</td>
</tr>
<tr>
<td>Gorilla</td>
<td>0.15</td>
<td>32.67%</td>
</tr>
<tr>
<td>Peter's duiker</td>
<td>3.13</td>
<td>4.78%</td>
</tr>
<tr>
<td>Red river hog</td>
<td>1.52</td>
<td>6.89%</td>
</tr>
<tr>
<td>Sitatunga</td>
<td>0.20</td>
<td>31.43%</td>
</tr>
<tr>
<td>Yellow backed duiker</td>
<td>1.40</td>
<td>8.38%</td>
</tr>
</tbody>
</table>

### Table D5. Relative abundance and coefficient of variation estimates from the camera trapping survey in SFID

<table>
<thead>
<tr>
<th>Species</th>
<th>Relative abundance</th>
<th>Coefficient of variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bay duiker</td>
<td>0.007122507</td>
<td>32.39%</td>
</tr>
<tr>
<td>Blue duiker</td>
<td>0.041310541</td>
<td>26.26%</td>
</tr>
<tr>
<td>Chimpanzee</td>
<td>0.011396011</td>
<td>68.93%</td>
</tr>
<tr>
<td>Elephant</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Gorilla</td>
<td>0.007834758</td>
<td>46.85%</td>
</tr>
<tr>
<td>Peter's duiker</td>
<td>0.000712251</td>
<td>102.96%</td>
</tr>
<tr>
<td>Red river hog</td>
<td>0.001424501</td>
<td>107.35%</td>
</tr>
<tr>
<td>Sitatunga</td>
<td>0.001424501</td>
<td>94.81%</td>
</tr>
<tr>
<td>Yellow backed duiker</td>
<td>0.004273504</td>
<td>43.54%</td>
</tr>
</tbody>
</table>
Table D6. Relative abundance and coefficient of variation estimates from the camera trapping survey in Pallisco

<table>
<thead>
<tr>
<th>Species</th>
<th>Relative abundance</th>
<th>Coefficient of variation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bay duiker</td>
<td>0.038006757</td>
<td>44.67%</td>
</tr>
<tr>
<td>Blue duiker</td>
<td>0.134290541</td>
<td>23.93%</td>
</tr>
<tr>
<td>Chimpanzee</td>
<td>0.006756757</td>
<td>34.86%</td>
</tr>
<tr>
<td>Elephant</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Gorilla</td>
<td>0.001689189</td>
<td>67.84%</td>
</tr>
<tr>
<td>Peter's duiker</td>
<td>0.032939189</td>
<td>36.02%</td>
</tr>
<tr>
<td>Red river hog</td>
<td>0.005912162</td>
<td>62.95%</td>
</tr>
<tr>
<td>Sitatunga</td>
<td>0.004222973</td>
<td>62.17%</td>
</tr>
<tr>
<td>Yellow backed duiker</td>
<td>0.002533784</td>
<td>70.47%</td>
</tr>
</tbody>
</table>
Data presented below is in addition to Table 4.3 in Results, demonstrating the effect that the number of transects and repeat visits to transects per survey has upon the CV of RA estimates calculated.

**Table E1.** Showing the effect of different combinations of transects and repeat visits on CV of RA estimates per species using data collected during the repeated linear transects survey in Pallisco.

<table>
<thead>
<tr>
<th>Survey design</th>
<th>Coefficient of Variation (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Transects</td>
<td>Visits</td>
</tr>
<tr>
<td>100</td>
<td>1</td>
</tr>
<tr>
<td>50</td>
<td>2</td>
</tr>
<tr>
<td>25</td>
<td>4</td>
</tr>
<tr>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>5</td>
<td>20</td>
</tr>
</tbody>
</table>

**Table E2.** Showing the effect of different combinations of transects and repeat visits on CV of RA estimates per species using data collected during the paired linear transects survey in Pallisco.

<table>
<thead>
<tr>
<th>Survey design</th>
<th>Coefficient of Variation (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Transects</td>
<td>Visits</td>
</tr>
<tr>
<td>100</td>
<td>1</td>
</tr>
<tr>
<td>50</td>
<td>2</td>
</tr>
<tr>
<td>25</td>
<td>4</td>
</tr>
<tr>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>5</td>
<td>20</td>
</tr>
</tbody>
</table>

**Table E3.** Showing the effect of different combinations of transects and repeat visits on CV of RA estimates per species using data collected during the paired recce transects in SFID.

<table>
<thead>
<tr>
<th>Survey design</th>
<th>Coefficient of Variation (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Transects</td>
<td>Visits</td>
</tr>
<tr>
<td>100</td>
<td>1</td>
</tr>
<tr>
<td>50</td>
<td>2</td>
</tr>
<tr>
<td>25</td>
<td>4</td>
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<tr>
<td>10</td>
<td>10</td>
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<tr>
<td>5</td>
<td>20</td>
</tr>
</tbody>
</table>