Challenges in implementing payments for environmental services for biodiversity conservation in a developing country context

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Abstract

Payments for environmental services (PES) have emerged over the past decade as an umbrella term for approaches that provide conditional positive incentives for ecosystem management. These incentives may compensate those who presently supply an environmental service or incentivise those who would otherwise not provide a service. In this dissertation, I investigate both the social and ecological dimensions of implementing PES for biodiversity conservation using the case study of a community-based payment scheme in Menabe, Madagascar.

I begin by examining limitations in the predominant definition for PES and suggest a refined framework for defining PES as a set of management tools. This revised PES approach relies on the "transfer of *positive incentives* to environmental service providers that are *conditional* on the provision of a service, where successful implementation is based on a consideration of *additionality* and varying *institutional contexts*." Next I examine the motivational impact of payments on influencing individual behaviours. In the case study, monitoring appears to be more important than payments in influencing individual behavioural change. I then explore the distribution of costs and benefits within a community-based PES scheme and outline the advantages and limitations of in-kind benefits, particularly the challenge of addressing variable opportunity costs. If payments are to be conditional on the state of a system or specific actions, then meaningful indicators must be developed. Using the case study's monitoring scheme as a framework, I show that the sampling of relevant indicators requires unrealistically high effort.

Those initiating community-based PES would benefit from a greater consideration of how payments impact individual and community decision-making processes, as well as of their ability to detect change in the indicators required for payment conditionality. These issues need to be addressed to improve the likelihood of management success in community-based PES for biodiversity conservation.

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List of Acronyms

AIC	Akaike Information Criterion
ANOVA	Analysis of Variance
AUC	Area Under the ROC Curve
CBNRM	Community-Based Natural Resource Management
CIFOR	Centre for International Forestry Research
COBA	Communauté de Base
CRES	Compensation and Rewards for Environmental Services
CRP	Conservation Reserve Program
FIRMS	Fire Information for Resource Managers
GCF	Gestion Contractualisée des Forêts
GELOSE	Gestion Locale Securisé
GLM	Generalised Linear Model
GLMM	Generalised Linear Mixed Effects Model
ICDP	Integrated Conservation and Development Project
IPES	International Payment for Ecosystem Services
MES	Markets for Ecosystem Services
MODIS	Moderate Resolution Imaging Spectroradiometer
NASA	National Aeronautics and Space Administration
NGO	Non-Governmental Organisation
NTFP	Non-Timber Forest Product
PES	Payments for Environmental Services (also, ecosystem or ecological services)
PRA	Participatory Rural Appraisal
PSA	Pagos por Servicios Ambientales
REDD	Reduced Emissions from Deforestation and Forest Degradation in Developing Countries
ROC	Receiver Operating Characteristic
USFWS	US Fish and Wildlife Service

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Chapter 1. Introduction

1.1 Background

Ecosystems provide valuable services to local, regional and international communities (Costanza et al., 1997; Millennium Ecosystem Assessment, 2005). However, traditional economic markets are underdeveloped or lacking for many environmental services such as watershed benefits, biodiversity conservation and carbon sequestration. As a result, decisions to convert or alter natural habitat towards market-based agricultural or timber activities generally fail to take into account the total costs of service loss (Westman, 1977; Hanley, 1992; Loomis et al., 2000). Where these services are of direct, indirect or non-use value to neighbouring or distant communities, the internalisation of these external values may tip the scales in favour of environmental service provision, particularly if competing resource uses are only marginally profitable (Pearce and Moran, 1994; Pagiola et al., 2004).

Payments for environmental services (PES; also ecological or ecosystem services) has been used as an umbrella term for approaches that provide positive incentives to environmental service providers, contingent on service delivery (Landell-Mills and Porras, 2002; Wunder, 2007). PES approaches have been widely implemented to provide carbon capture and storage benefits (Pollini, 2007), watershed services, including erosion control (Kosoy et al., 2007; Bennett, 2008) and water quality (Asquith et al., 2008; Wunder and Alban, 2008), as well as landscape benefits (Robertson and Wunder, 2005).

Despite the applications of PES interventions, its definition has been for the most part implicit. While a recent definition by Wunder (2005, 2007) has been widely used, even Wunder acknowledges the limitations of his definition in dealing with the full variety of interventions that have been labelled PES approaches (Wunder et al., 2008).

While PES has been used for biodiversity conservation, particularly in the case of US and European agri-environment payments (Wu, 2000; Glebe, 2007; Dobbs and Pretty, 2008), there is less documented experience with the paradigm for biodiversity conservation in developing countries. A disproportionate number of the world's vulnerable terrestrial species are located in developing countries (Bailie et al., 2004) and hundreds of millions of dollars are spent annually on efforts to conserve this biodiversity (James et al., 1999; Balmford et al., 2003). A number of authors have highlighted the potential for PES to deliver biodiversity benefits in developing countries (Simpson and Sedjo, 1996; Ferraro, 2001; Ferraro and Kiss, 2002), as the variety of incentive structures for biodiversity conservation that have been trialled over recent decades in developing countries have produced mixed results (Wilshusen et al., 2002; Kiss, 2004; McShane and Wells, 2004; Naughton-Treves et al., 2005). The exisiting biodiversity conservation PES interventions in developing countries are just beginning to reach maturity and there have been few evaluations published of their effectiveness (Clements et al., 2009; Ferraro and Gjertsen, 2009). As a result, there is a need for researchers to evaluate the social and ecological effectiveness of biodiversity PES case studies in order to inform the design of future interventions.

Many academics and practitioners have been concerned with the quality of evaluations of the effectiveness of conservation interventions (Newmark and Hough, 2000; Agrawal, 2001; Sutherland et al., 2004; Stem et al., 2005; Bertzky and Stoll-Kleemann, 2009). This can be partially attributed to the lack of baselines, limited investment in long-term monitoring of project indicators and lack of well-defined goals (Kremen et al., 1994; Kleijn and Sutherland, 2003; Borgerhoff-Mulder et al., 2007; Kapos et al., 2008). Additionally, organisations implementing interventions have not historically had great incentives to critically evaluate their projects publically for fear of revealing failures (Redford and Taber, 2000). As a result, many of the evaluations of conservation interventions that have emerged have been criticised for being overly qualitative and lacking statistical rigour (Agrawal, 2001; Kleijn and Sutherland, 2003). In response to these criticisms, there have been calls for evaluations of new paradigms like PES to apply methods from other social science disciplines, particularly the use of control groups or matching (Salafsky and Margoluis, 2003, Ferraro and Pattanayak, 2006). Ferraro (2009) argues that simply noting change from a baseline within an intervention is not sufficient to demonstrate what would have occurred in the absence of an intervention (the counterfactual). These calls for evaluation underscore the need for PES researchers to define frameworks for how incentive structures function and how specific interventions are expected to influence the behaviours of participants (Margoluis et al., 2009).

Success of many biodiversity conservation initiatives relies on influencing the natural resource-use decisions of individuals in local communities (Barrett and Arcese, 1998; Adams and Hulme, 2001). Managers generally implement positive incentives, such as subsidies ("carrots"), or negative incentives, such as fines or imprisonment ("sticks") to help influence local behaviours (Emerton, 1999). While a conservation paradigm may be labelled based on a single positive or negative incentive; in practice, most interventions rely on a collection of incentives. For example, protected areas may use guards to monitor non-compliance and distribute fines, resulting in the label, "fortress conservation" (Brockington, 2002). Yet many protected areas have concurrent programmes or employment opportunities that provide positive incentives (Heinen, 1996). There is thus a need to better understand how the variety of incentives that influence individuals' decisions interact in conservation interventions, and PES in particular (DeCaro and Stokes, 2008). Many past interventions have evaluated the degree to which communities support an intervention as the primary measure of success (Mehta and Heinen, 2001; Infield and Namara, 2001; Baral and Heinen, 2007). But, while attitudes are important for the sustainability of interventions, they do not necessarily reflect changed behaviours or successful conservation outcomes (Holmes, 2003). Thus, understanding the role of incentives in conservation interventions requires both a consideration of attitudes and behaviours.

In addition to using monitoring to evaluate how incentives influence behaviours and attitudes, the use of monitoring to define payments poses practical challenges to PES implementation. Monitoring is explicitly built into PES interventions, as payments are conditional on service provision (Ferraro, 2001), but there are many options with respect to the selection of indicators and decisions on how monitoring results will be used to define payments. Given that there is uncertainty in monitoring, particularly of rare indicators, a certain amount of effort is required, alongside appropriate monitoring design, to ensure that differences between samples represent true differences rather than random variation (Yoccoz et al., 2001; Field et al., 2007). There is thus a need for all conservation interventions, and PES in particular, to consider how they can establish monitoring frameworks that provide cost-efficient and meaningful results (Legg and Nagy, 2006).

1.2 Aims and objectives

In this thesis I aim to increase understanding of how incentive structures influence the implementation and impact of conservation interventions by examining challenges in the design and implementation of PES interventions for biodiversity conservation in developing countries. I address a number of identified gaps in the literature concerning: definitions of PES; rigorous evaluations of the effectiveness of interventions; the relative impact of incentives on behaviours; and the role of indicators and monitoring. I use a long-running, community-based PES for biodiversity conservation in Menabe, Madagascar as a case study. I isolate the impact of the intervention based on reported behaviours with control communities that had not been included in the PES. The differences among the eight communities participating in the intervention allow me to evaluate distribution and fairness based on institutional differences among villages, as well as the characteristics of individuals within villages. The scheme's existing monitoring framework provides an opportunity to examine the effort required to monitor different aspects of biodiversity service provision.

My specific objectives in this thesis are:

- To clarify the conditions under which an intervention may be considered PES;
- To evaluate the success of a PES intervention at influencing biodiversity conservation behaviours;
- To examine the role of positive and negative incentives in influencing individual biodiversity conservation behaviours;
- To evaluate how the distribution of benefits from PES interventions is related to attitudes and participation; and
- To evaluate the strengths and limitations of ecological monitoring approaches in providing conditionality for biodiversity PES interventions.

1.3 Thesis structure

Chapter 2 examines limitations in the current predominant definition for PES and presents a novel framework for defining whether an intervention should be considered a PES. I suggest that positive incentives and conditionality are the defining criteria of a PES, but highlight the importance of considering additionality and institutional contexts. Chapter 3 places PES for biodiversity conservation in the context of past biodiversity conservation interventions in developing countries, examines reasons for the limited implementation of PES for biodiversity in developing countries, and presents background information on the case study and methodological approaches used in the rest of the thesis. Chapter 4 evaluates whether the Menabe PES intervention has achieved conservation benefits in terms of actual behavioural change of resource users and explores the relative importance of payments and other components of the conservation intervention in influencing the resource management decisions of individuals. Chapter 5 identifies factors that influence whether individuals perceive themselves or their community to have benefited from the Menabe intervention and whether the distribution is considered fair. Chapter 6 examines the effort required to monitor biodiversity indicators in the Menabe and questions whether basing payments on biodiversity monitoring outcomes is feasible. This is important both because the conditionality of PES payments rests on reliable monitoring of service provision and because budgets are limited, leading to a trade-off between action and monitoring. Chapter 7 highlights the key findings of this research and offers both recommendations for implementing biodiversity conservation PES interventions in developing countries and recommendations for future research to improve our understanding of how PES interventions can be used to incentivise conservation behaviours.

Chapter 2. A revised conceptual framework for payments for environmental services

2.1 Introduction

In this chapter, I examine a recent widely accepted definition of payments for environmental services (PES), highlight its strengths and weaknesses, and propose a new framework to evaluate whether an intervention should be considered a PES. This will provide a framework for examining a PES for biodiversity conservation in subsequent chapters. Despite the emergence of PES as a common resource management approach, the definition of PES has been for the most part implicit. However, a recent definition by Wunder (2005, 2006, 2007) has received widespread acceptance from the academic community (articles cited more than 150 times between January 2005 and December 2008 according to Google Scholar), as well as from practitioners of payment interventions, such as the Center for International Forestry Research (CIFOR) and the Katoomba Group's Ecosystem Marketplace. Wunder classifies a PES as a:

- voluntary transaction where
- a well-defined environmental service
- is being 'bought' by a minimum of one environmental service buyer
- from a minimum of one environmental service provider
- if and only if the provider secures environmental service provision (conditionality)

Those who have evaluated PES interventions in practice have found that many interventions attempting to implement PES do not meet all of these criteria. For example, perhaps the most well known tropical PES, Costa Rica's countrywide Pagos por Servicios Ambientales (PSA), would not be considered a pure-PES by Wunder's definition, given that payments from some user groups are non-voluntary (Pagiola, 2008). Swallow et al. (2007) present their discomfort with the current definition:

> "(W)e do not challenge the Wunder definition of PES per se, but do doubt its usefulness for describing and analysing the range of interesting and important mechanisms that are being negotiated for managing interactions between people with diverse interests in ecosystem management and ecosystem services."

In order to deal with these various contexts, there have been a number of attempts to develop alternative vocabulary to PES, such as *Compensation and Rewards for Ecosystem Services* (CRES) (Swallow et al., 2007), or sub-categories of PES, like *Markets for Ecosystem Services* (MES) (Pagiola and Platais, 2002) and *International Payments for Environmental Services* (IPES) (UNEP et al., 2006). Nevertheless, PES remains the most widely used and recognised term.

There is wide acknowledgement that because of the variety of local institutional contexts surrounding natural resource management, pure-PES approaches that fulfill all of the criteria may not always be possible, or even preferable (Wunder, 2005; Corbera et al., 2007; Engel et al., 2008). As a result, whether a project can be considered a PES is frequently unclear. For example, Wunder et al., (2008) describes South Africa's Working for Water programme as a 'PES-like' programme due to the fact that those being paid are not necessarily the *de jure* or *de facto* land owners. In contrast, the same article considered China's Sloping Lands Conversion Program a PES programme despite participation frequently being involuntary (Wunder et al., 2008; Bennett, 2008).

> "It becomes a judgement call as to whether several individual programs should be considered 'PES with qualifications,' or 'non-PES with PES-like characteristics'…Even among us three editors, there is thus some disagreement over where exactly the line between PES and non-PES should be drawn." (Wunder et al., 2008)

There is a risk that the use of terms such as 'PES with qualifications' or 'PES-like' implicitly suggests that interventions not fulfilling all of the definition's criteria are inferior. I believe that PES is best seen as an umbrella term for a set of resource management tools that are based on the philosophy of implementing conditional positive incentives in a wide variety of institutional contexts. I therefore suggest a revised framework for PES that focuses on two core criteria forming a definition of PES and two additional principles to guide planning and implementation.

2.2 A revised framework

I propose a definition of PES that does not relax Wunder's (2005) definition, but rather refines and refocuses it on the two criteria that I believe make PES a unique and powerful set of approaches for ecosystem management. I define PES as approaches that aim to:

- Transfer *positive incentives* to environmental service providers that are
- *conditional* on the provision of the service

where successful implementation is based on a consideration of:

- *additionality* and
- varying *institutional contexts.*

Table 2-1 Comparison of PES criteria from this paper and **Wunder's definition.** This also presents justifications for the changes made in this paper.

This paper	Wunder	Justification for differences between definitions
Criteria	Criteria	
Conditionality	Conditionality	Methodological core of both definitions that incorporates monitoring and the definition of the service goal
Positive incentives		Ideological core to both definitions and implicit in Wunder's; is also implicit in the use of the term *payment* as opposed to *fines*, though positive incentives are not limited to monetary transfers
Considerations		
Additionality		Reflective of the social or ecological goals of the intervention and allows wider impact to be measured. Discussed by Wunder but not a criterion
Institutional context		Practical implementation issues that will vary among PES interventions, implicit in Wunder's discussion of buyers, service providers, voluntary and PES-like interventions
	Voluntary	Falls within a consideration of institutional context, as the voluntary criterion may be met to varying degrees within a conditional positive incentive
	Well-defined service	Implicit in the concept of conditionality
	Service buyer	Implicit in the idea of a transfer and fits better as a component of the institutional context
	Service provider	Same as for buyers

I consider four of Wunder's five criteria to be present within my framework, yet only the conditional criterion is common to both (Table 2-1). I do not argue that a well-defined service, buyers, and service providers are not important to PES, but rather that they are implicit within the criterion of conditionality and they are best considered as part of the institutional context. The one criterion that I believe is not crucial to all PES interventions is that it must be voluntary. While I agree that PES is voluntary at the level of the transaction (i.e. service providers can decide whether or not to accept payment), service providers do not necessarily have the choice whether or not to provide the service, such as in cases where land-use change is illegal. Instead I propose that the extent to which a PES is voluntary belongs within the discussion of institutional contexts. I believe my framework offers the flexibility for practitioners and academics to develop specific tools for varying situations, but retains a grounding in the spirit and methodology of PES. I use the remainder of the paper to elaborate on the two core criteria, evaluate policy tools based on the two criteria and finally explore the roles of additionality and institutional contexts in the implementation of PES.

2.3 Criteria for PES

2.3.1 Criterion 1 – Positive incentives

The use of positive incentives, including but not limited to payments, is the core ideology of PES. In this section, I differentiate between positive and negative incentives and discuss the use of positive incentives in PES to impact both behaviours and attitudes. I also elaborate on the need to consider the distributional impacts of implementing positive incentives at a range of scales, the role of negative incentives and the overemphasis on monetary payments in current PES literature.

Incentives are factors that influence a decision maker's motivation to engage in an action. I classify incentives as positive or negative based on whether a decision maker perceives a gain or loss from their baseline. A PES scheme should aim to provide net-gain for participants through the use of positive incentives. Most commonly this involves a material compensation or reward for individuals based on opportunity costs incurred by stopping a behaviour that is detrimental to service delivery, or for taking actions to increase or maintain service delivery.

In the most straightforward PES approach, individuals possess legal control over service provision (e.g. have the right to carry out certain land uses changes which would change service provision) and incentives are transferred to influence the decision to produce the service. Indeed, in this case, the implementing organisation or user group is constrained to using positive incentives, as there may be no legal justification for negative incentives. This context is particularly common when individuals are paid to implement certain farming practices, such as to develop or maintain hedgerows under agri-environment schemes (Dobbs and Pretty, 2008). This is also the case in most user-financed PES schemes where downstream water users create incentives for upstream landowners to safeguard water quality through particular land management practices.

Positive incentives may also be used in a PES to influence attitudes towards a regulation or a change in legal enforcement. For example, Pagiola (2008) suggests that a primary reason behind the establishment of Costa Rica's PSA was to make a legislative ban on deforestation on private lands more palatable to landowners and to entice them to cooperate. In these circumstances, the PES system does not drive the change in behaviours. Nevertheless, it is important for achieving the social support that may ultimately strengthen compliance with the anti-clearance law (Chapter 4).

The issue of the scale at which positive incentives need to be felt for an intervention to be a PES needs careful consideration. For example, if payments are made to a regional or central government, rather than to individuals making the resource use decisions, then positive incentives may not be driving the provision of service on the ground though they drive the decision of the government to participate. This is the case within the nationally managed system of Conservation International's Guyana conservation concessions, where payments are made to the government to offset logging opportunity costs (Hardner and Rice, 2002). While this may be considered a PES at the national level, the government may use a variety of negative incentives to ensure local compliance. In such situations, a PES may appear to local people little different from traditional law enforcement approaches. This issue may become particularly relevant as the methodologies for Reduced Emissions from Deforestation and Forest Degradation (REDD) are put into practice under the UN Framework Convention on Climate Change, as emissions credits are likely to accrue at the national level. Even where incentives are targeted at the community level, distributional issues within the villages may mean that some community members do not receive benefits but that compliance is obtained by coercion from within the community (Grieg-Gran et al.; 2005, Pagiola et al., 2005). Such examples muddy the consideration of Wunder's voluntary criterion and project funders may have to decide whether it is possible or even preferable to be concerned with the ultimate distribution of benefits once the payments are made. Nevertheless, it is clear that positive incentives drive the participation at the level of the transaction, and that for a scheme to reflect the spirit of PES there should be an explicit attempt to transfer these positive incentives down to the those who control provision of the service. Indeed such concerns are being addressed in the development of REDD methodologies, as safeguards are likely to be implemented to ensure that REDD has local level benefits (Peskett et al., 2008).

A dedication to positive incentives does not imply the absence of negative incentives within a PES intervention. When a PES scheme is used to influence attitudes in a regulatory environment, regulation acts as a negative incentive. Furthermore, the existence of conditionality can be perceived as a negative incentive, as there are repercussions for breaking a contract or agreement. Negative incentives may emerge from PES systems through coercion to participate due to social pressures from other community members. Nevertheless, by definition, the positive incentives received from participation should outweigh the negatives for those participating in a PES.

In the design of PES interventions, the role of monetary payments as positive incentives needs to be considered. Payments have been shown to act under some circumstances as negative incentives, as small payments insult participants and thus lower individuals' motivation, or payments can "crowd out" other pre-existing forms of motivation such as altruism (Gneezy and Rustichini, 2000; Frey and Jergen, 2001). Similarly there is cause for concern in terms of diminishing returns on motivation through time from the repeated use of positive incentives (Benebou and Tirole, 2003). For example the positive incentives may become perceived over time not as incentives but as entitlements and thus lose their motivational force.

Finally, there has been an overemphasis on monetary payments within the general PES discourse, which frequently settles on the term "payments," and the monetary transfers this implies. Since the aim of PES is to influence the behaviours of those who have some control over service provision, the impact of positive incentives other than payments should be considered. Indeed, individuals rarely act as pure profit-maximisers (Frank, 1987). Rather, social cooperation, local norms or religious beliefs also influence behaviour (Deci and Ryan, 1985; Heinrich et al., 2001; Ajzen and Fishbein, 2005). Positive incentives may therefore come from social impacts such as tenure legitimacy and pride, in addition to monetary transfers. This highlights the need to evaluate the roles and interactions of a range of potential positive and negative incentives throughout an intervention's lifespan. For example, the complex interactions of monetary payments with other positive and negative incentives have been apparent in PES schemes when individuals choose to provide services at payments lower than their opportunity costs (Wunder, 2005; Kosoy et al., 2007).

2.3.2 Criterion 2 – Conditionality

Conditionality is the core method for motivating service provision, as it creates a consequence of not providing the service. The use of conditionality also makes the definition of the service, the monitoring regime and enforcement explicit to the buyer and service provider. This term thus subsumes Wunder's criteria of conditionality and a well-defined service and places a focus on the issue of monitoring.

Whether to make incentives conditional on measurements of the service itself or of the actions taken by providers is an important decision for those designing PES interventions (Engel et al., 2008). Which approach to take relates primarily to the technical challenges and costs of monitoring. Because of the difficulties of measuring changes in environmental services, payments are often conditional on ecological indicators with assumed relationships to service provision, rather than based on the flow of the service itself. For example, carbon service provision may be estimated by monitoring coarse changes in habitat that can be observed remotely by satellites (Sanchez-Azofeifa et al., 2007). Alternatively, since PES interventions seek to change behaviours, payments conditional on specific actions of service providers are also common. European agri-environmental biodiversity payments are based on the assumed relationships between actions and environmental outputs, such as the creation of hedgerows and biodiversity, rather than payments for delivery of directly measured environmental benefits (Glebe, 2007; Dobbs and Pretty, 2008). Such action-based agreements increase the farmer's and policy maker's certainty concerning what they will pay and receive. However, they increase uncertainty associated with ecological outcomes (Kleijn et al., 2004). Thus, the most appropriate monitoring to employ depends on the capacity to observe the service, the capacity to observe the actions or effort of the service providers, and the strength and consistency of the relationship between the providers' actions and service provision (Chapter 6).

It is not only what is monitored that impacts the efficacy and cost effectiveness of conditionality, but also who monitors. The most cost effective schemes are likely to have a structure where the service provider has an incentive to truthfully monitor and report their own actions (Laffont and Martimort, 2001). Alternatively, in systems where service delivery is contingent on multiple resource management units, monitoring by peers may be effective. The most challenging monitoring regimes force service buyers to monitor diffuse services where individual shirkers cannot be easily identified. This is the case for community based PES and in these circumstances, for a scheme to adequately incentivise individuals, an entire group of service providers may have to bear the cost of a single individual's non-compliance (Meijerink, 2007). These costs may mean that payments stop flowing to the service provider, or that the provider incurs a fine. Social disapproval may also play an enforcement role in group schemes. While it is critical to understand how various structures of monitoring and enforcement impact intervention success, the wide range of these issues are best dealt within a consideration of institutional contexts. Nevertheless, conditionality is undoubtedly a central criterion that is critical to the functioning of PES schemes.

2.4 Evaluation of incentive structures

My framework does not create an unambiguous line separating various policy tools. While interventions that rely on negative incentives, such as fines and regulation, clearly do not fall under PES, some policy tools fulfil the PES criteria more often than others. Policies that rely on positive incentives for service provision, such as integrated conservation and development projects (ICDPs), ecological certification, quota-based trading systems, transferable development rights and government subsidies, may require a nuanced evaluation to determine whether they qualify.

ICDPs are one of the most common approaches in international conservation, but they generally do not fit the PES criteria. This is primarily because the payments are not explicitly conditional on provision of service benefits (Ferraro, 2001). Furthermore, making development benefits, such as access to health and education or construction of facilities for public use, conditional on service provision may be morally unacceptable.

I do not consider certification or eco-labelling to constitute a PES, as there is no assured relationship between certification and a positive incentive (Sedjo and Swallow, 2002). Though an incentive in the form of higher price for certified goods is the objective, it is not guaranteed and depends on a market being available. In contrast, schemes that provide a guaranteed price premium to service providers for the provision of an environmental service would be considered a PES.

Many markets for environmental services, like the European Trading System for carbon, or tradable harvest quotas for fish or game species would not be classified as PES interventions. Here, an environmental service is provided by a regulation that caps emissions or harvesting and, in these cases trading becomes a mechanism to ensure an efficient distribution of rights (Tietenberg, 2003). However, individual payments within such a cap may be consider PES.

Within the international climate regime, the likely implementation of REDD will act as a PES where governments become the primary buyers and providers (Ebeling and Yasue, 2008). As implementation on the ground is likely to be devolved in many countries to NGOs, they may also use REDD funds from the government to implement local level PES schemes. However it is not clear whether the national and sub-national level activities under REDD will necessarily fulfil PES criteria. In contrast, the Clean

Development Mechanism and joint implementation mechanisms fulfil PES criteria in theory, as incentives are transferred based on discrete, measurable and additional reductions in carbon emissions (Streck, 2004).

Conservation easements, and indeed any intervention where one-time property rights are transferred to another group, would not likely be considered a PES, as once the rights are transferred; the new owner of the rights becomes the service provider. However, there may be an implied relationship between two policy interventions. For example conservation easements are frequently implemented along with preferential tax assessments whereby preferential taxes are given to land that has easements on it. A preferential tax assessment based on environmental service provision would be an unambiguous PES system, but the presence of an easement simply represents a way of targeting landowners who are more likely to provide a service. A similar example can be observed in the US Fish and Wildlife Service (USFWS) Conservation Banking programme. Conservation banking systems typically use trust funds to finance annual management of habitats for environmental services and would thus be considered a PES (USFWS, 2009). However, similar to preferential tax assessments, the USFWS incorporates an eligibility prerequisite that land must be under a development easement.

Governments are frequently purchasers of environmental services through subsidies for environmental service production, for example in EU agri-environment schemes (Dobbs and Pretty, 2008). In these cases, payments are linked to actions undertaken by farmers. Yet the role of governments in PES interventions may be complicated because the range of government resource management policies may or may not be perceived as separate from one another. This is particularly acute in programmes such as the US Conservation Reserve Program (CRP) where one of the explicit goals, alongside ecological goals of preventing erosion and creating habitat, is to "provide needed income support for farmers" (Reichelderfer and Boggess, 1988). Indeed, because positive incentives drive my framework for examining PES, whether or not an incentive scheme acts as a PES may be contingent on how a government portrays the policy.

2.5 PES Considerations

While conditional positive incentives are the defining characteristic of PES, in practice there are additional considerations that influence the ultimate success of an intervention. In particular, a consideration of additionality provides assurance to investors that an intervention will have a measurable impact, while a consideration of the institutional context surrounding implementation ensures that the specific design of a PES is appropriate. It is not within the scope of this chapter to fully elaborate on these considerations. However, I will draw out their relevance to PES and highlight areas that are particularly controversial.

2.5.1 Consideration 1 – Additionality

Whereas conditionality allows one to demonstrate the impact of an intervention (i.e. has the service provider met the conditions of the agreement?), additionality is the measure of outcomes in relation to what would have occurred in the absence of the intervention. Additionality is therefore of equal or greater interest to funders and the wider community, as it is essential for assessing intervention impact (Engel et al., 2008). A number of PES evaluations have determined *post-hoc* that they have achieved relatively little, if any, behavioural or ecological additionality (Sanchez-Azofeifa et al., 2007; Munoz-Pina et al., 2008). Wunder (2007) highlights the importance of additionality in his discussion of PES though he does not include it into his defining criteria. While I agree that additionality is not a defining criterion for an intervention to be a PES, it should be an aspiration for all environmental management interventions.

Additionality is a central criterion of carbon offset markets, the most developed environmental service markets (Pfaff et al., 2000; Niesten et al., 2002). Additionality is also frequently used as an indicator of PES effectiveness (Engel and Palmer, 2008; Wunder et al., 2008). Unfortunately it is extremely difficult to demonstrate additionality due to the methodological and practical challenges of estimating baselines, measuring the service itself, and identifying leakage (Aukland et al., 2003). With respect to leakage, Wu (2000), for example, demonstrated that for every 100 acres of cropland taken out of production by the CRP, 20 acres were brought into production, resulting in fewer benefits than advertised. I suggest that the spatial and temporal scales at which an intervention will be additional should be considered in advance of PES interventions, allowing investment opportunities to be compared. Nevertheless, it is clear that, due to the challenges of establishing additionality, in many cases these estimates will be qualitative at best.

There may be cases where ecological additionality may be a secondary objective for a PES. Because of targeting difficulties, or due to social equity or political goals, it may not be possible to pay only those whose supply of the service is likely to depend on the payment. Information on the likely supply of a service in the absence of payments may be difficult or costly to obtain. There are techniques to target payments in the absence of such information (Ferraro, 2008; Wunscher et al., 2008; Barton et al., 2009), but this is costly and often requires large amounts of data. Some concern has been raised that targeting payments too narrowly on those likely to produce true additionality may create perverse incentives (Pirard and Karsenty, 2009). For example, if payments are made only to landholders who are thought likely to convert their land, this may encourage increased land conversion by others. These issues should be explicitly acknowledged in the planning stages to avoid unrealistic expectations of the ecological or social impacts of a PES. Although I consider additionality to be unfeasible as a defining criterion of PES schemes, it must be a guiding consideration to ensure that it does not become acceptable to infuse substantial funds into communities as incentives without concern for what has been achieved (Ferraro and Pattanyak, 2006).

2.5.2 Consideration 2 – Institutional context

While my two criteria define the essence of a PES intervention, it is the institutional structure that guides the practice and ultimate effectiveness of an intervention (Engel and Palmer, 2008; Corbera et al., 2009). PES opportunities do not only exist within the narrow confines of voluntary transactions between buyers and providers. Instead there are a wide variety of situations in which environmental service suppliers and buyers can operate. As a result, interventions frequently diverge from Wunder's criteria based on these constraints. The institutional context may be organised generally in terms of characteristics of the buyers, service providers and the relationship between the two and expressed through a series of generalised dichotomies (Table 2-2). This chapter does not elaborate on each of the contexts, but rather highlights key issues and alerts the reader to case studies where specific institutional issues have been considered.

Table 2-2 Non-exhaustive list of institutional contexts for PES implementation. Characteristics on the left side of the options are generally (although not always) easier contexts for implementing PES. These choices often rest within a continuum rather than a dichotomy.

Characteristics	Institutional context	Options		
Service provider	Governance type	Democratic	vs	Authoritarian
	Type of provider	Individual	vs	Community
	Property tenure	Private property	vs	No tenure
	Legality of behaviours	Legal	vs	Illegal
	Opportunity costs	Homogenous	vs	Variable
Buyer	Buyer's funding	Secure	vs	Insecure
	Buyer goals to trade-off	Economic efficiency	vs	Equitable distribution
	Additional buyer goals to trade-off	Social	vs	Ecological
Relationship	Threats to system	Internal	vs	External
	Distance between buyer and provider	Local	vs	International
	Relationship between buyer and provider	One-on-one	vs	Intermediaries
	Negotiations	Market-based	vs	One-off/ project-based negotiation
	Participation constraints	Voluntary	vs	Regulated

Buyer and Service Provider characteristics

We consider the existence of buyers and service providers of environmental services to be implicit within the two defining criteria. Yet the wide variety of buyers and providers and their potential relationships are important to consider in the design of a PES. Choosing the most appropriate provider may be of interest, for example, when multiple providers have legitimate claims to a single service. This may be the case when local community members have customary rights on government property. As a result, it is not always immediately clear whether payments should be made to individuals, the community, or government. Within the PES case study in the Central Menabe, Madagascar that I will elaborate on in Chapter 3, Durrell Wildlife Conservation Trust (Durrell) makes payments to communities with forest management rights (Durbin, 2002), whereas in Guyana payments are made to influence national government priorities (Hardner and Rice, 2002). In each case the service buyer made a decision as to which service provider it believed to be the most effective institution to negotiate with. Had Durrell decided to negotiate with the central government it is unlikely that management information and benefits would have effectively trickled down to the local communities, and the PES been effective (R. Lewis, personal communication).

Relationship characteristics

Within an agreement between two or more parties, there are a wide variety of characteristics that can describe the relationship. These can relate to spatial, temporal, legal and power components of the relationships. While the voluntary nature of an agreement has been used as a defining characteristic of a PES, this criterion is frequently not met in interventions commonly considered as PES (Robertson and Wunder, 2005) and there are many examples where payments are used to alter behaviours that are already illegal.

A non-voluntary PES approach may also be applicable where national level enforcement of laws is limited and third party organisations have a presence. In these cases, individuals can engage in PES voluntarily. However they do not have the right to break the law. In the Central Menabe implementing a community-based biodiversity PES has given Durrell a rationale for monitoring illegal forest-use behaviours. This has both acted as a direct deterrent and has stimulated actions by local forest monitoring councils to influence individual forest use behaviours (Chapter 4). This approach has the added benefit of filling an institutional void left by limited government presence in the region. It thus demonstrates the capacity for PES to complement the deficiencies in existing enforcement regimes.

Finally, the concept of a voluntary transaction is complicated in cases where PES is negotiated between collective groups at scales from small organisations to national governments. In such cases, the views of disadvantaged minorities within the community may be discounted (Corbera et al., 2007). As a result, although a PES intervention may be overtly voluntary, in reality certain subgroups may be participating due to coercion. Whether all those affected by a PES participate voluntarily depends on who can most effectively exert control over the services of interest and who is in a position to negotiate (Engel and Palmer, 2008).

How PES characteristics can be adapted to the institutional context

The issues highlighted here describe some of the institutional characteristics that project developers may encounter. Practitioners have choices in structuring the positive incentives, conditional methodology, and consideration of additionality that allow them to adapt to these institutional constraints and opportunities. The PES discipline has begun to explore many of structural choices based on early experience with PES tools. However, a great deal of applicable research has already been performed in the disciplines of economics, sociology and management theory to guide this learning process (Laffont and Martimort, 2001; Table 2-3).

Table 2-3 List of examples of non-exhaustive characteristics of PES design choices. These may be modified in order to address local institutional constraints.

Potential design characteristics	Criterion/ consideration	Example			PES studies that raise these design characteristics	Studies from other literatures that examine these characteristics
Incentive type	Positive Incentive	Cash	vs	In-kind	Wunder, 2005; Asquith et al., 2008; Engel et al., 2008	Currie, 1994; Currie and Ghavari, 2008
Contract type	Conditionality	Formal contract	vs	Implied agreement		Levin, 2003
Payments based on	Conditionality	Defined actions	vs	State of system	Centre for Rural Economic Research, 2002; Musters et al., 2001; Engel et al., 2008	Baker et al., 1988; Holmstrom and Milgrom, 1991
Size of payment based on	Conditionality	Performance relative to others	vs	Individualised specific criteria		Nalebuff and Stiglitz 1983; Malcomson, 1984
Monitoring (personnel)	Conditionality	Local agents	vs	Hired agents	Pagiola, 2008	Holmstrom, 1979; Frey, 1993; Cowen and Glazer, 1996; Gibson et al., 2005
Monitoring (method)	Conditionality/ Additionality	On-ground	vs	Remotely		
Payment time horizon	Conditionality/ Additionality	Annually	vs	End of agreement	Marland et al., 2001; Wunder, 2005; Peskett et al., 2008	
Openness of incentives (spatial)	Additionality	Inclusive	vs	Targeted	Watzold and Dreschler, 2005; Wunscher et al., 2008; Barton et al., 2008; Hartig and Dreschler, 2009	
Openness of incentives (participation)	Additionality	Inclusive	vs	Targeted	Ferarro, 2008	van de Walle, 1998

2.6 Conclusions

PES is widely promoted as a novel set of tools for environmental management. Wunder's definition has helped structure academic and practitioner thinking about this novel approach to conservation. This modified framework develops Wunder's definition and focuses on the two principles that define the PES approach: positive incentives and conditionality. This highlights the novelty of the PES approach while being inclusive of a wide range of situations. A commitment to positive incentives in motivating resource-use decision making is the ideological basis for PES, while the methodological core is based on making these incentives conditional on monitored provision of a service. The demonstration of benefits additional to those that would have occurred without the intervention represents an aspiration for PES interventions, while the variety of institutional contexts informs planning and implementation. By making the definition of PES more inclusive and representative of on-the-ground realities, I focus attention on the core principles of this approach. This revised framework provides a useful point of departure for future theoretical and practical work for PES.

Chapter 3. Biodiversity conservation through payments for environmental services in developing countries: context, challenges and a case study

3.1 Introduction

In the last chapter I presented a framework for better understanding which types of interventions may be considered a payments for environmental services (PES) scheme and a discussion of the considerations that influence successful implementation. Here I examine PES in the context of biodiversity conservation in developing countries, my case study from Central Menabe, Madagascar and my general methods. First, I provide a brief overview of the challenges of managing the commons and compare the advantages and disadvantages of PES for biodiversity conservation to other incentive structures used in biodiversity conservation. Next, I examine the challenges for successful implementation of PES for biodiversity conservation in developing countries. I then introduce the Durrell Wildlife Conservation Trust PES case study from Central Menabe, Madagascar, and finally I provide an overview of the approach to data collection and statistical analysis used in this study.

3.2 Resource management and biodiversity conservation approaches in developing countries

3.2.1 Managing the commons

Researchers from a variety of disciplines from the social and biological scientists, including economists, geographers, anthropologists and conservation biologists, have examined the challenges faced by communities in resource management through a variety of lenses. One of the most common approaches of geographers has been to classify the role of communities in resource management as a tragedy of the commons scenario, whereby in the absence of clear property rights, rational individuals will deplete a resource as they can absorb the all the benefits of using a resource but share only a portion of the costs of depletion (Hardin, 1968). Authors have since posed this community resource management challenge as a collective action problem and have clarified the distinction between open access resource rights and common property (Berkes et al., 1989; Bromley 1992). Others have further defined common property resources as areas where exclusions of beneficiaries is costly and the use of the resource by one individual limits the potential use by others (Ostrom et al., 1994). Ostrom (1990) examined in particular where common property has been managed effectively and identified, *inter alia*, small homogenous communities; locally adapted rules; monitoring; low cost conflict resolution; and decision making authority for all actors as important preconditions to successful management of commons.

A classical economics approach to Hardin's tragedy of the commons problem is to define it as a problem of unclear rights and to advocate a clarification of rights, often through privatisation of the resource (Baland and Platteau, 1996). Privatisation of rights has been met with some success in marine fisheries, for example through individual transferable quotas (Eythorsson, 1996). In terrestrial systems, it has had limited success, in part because in many cases communities only possessed traditional customary use rights rather than full property rights over common land. Furthermore, land-tenure reform tends to be emotive and politically challenging, particularly in the numerous cases where it has led to the exclusion of local communities (Colchester, 1994; Cernea and Schmidt-Soltau, 2006). Privatisation in the context of conservation can also be challenging if it does not match local livelihood needs, for example seasonal migration for pastoralists or farmers (Reid et al., 2003).

Other social scientists have focused their lens on the relationship between local communities and conservation, rather than simply community resource management. The concept of the ecological noble savage examined the proposition that local communities or individuals are better stewards of ecosystems than national governments (Redford 1991). Alvard (1993) demonstrated that a number of South American communities did not show any restraint in their hunting behaviours when populations became locally rare, and highlighting the importance of separating the state of the system from the actual behaviours of individuals. Despite this cautionary tale, the concept that local communities manage resources more sustainably than central governments has received increased interest as scientists explore potential community based approaches to contribute to climate change mitigation through reduced emissions from deforestation and forest degradation (REDD). Recent studies have argued that communities manage forests more effectively than government led top down approaches (Agrawal and , 2010; Nepstad et al., 2009). However, such conclusions continue to beg the question of whether communities are actively managing such forests, whether they simply lack the technological capacity to exploit the resources, or have not yet reached the resource.

Other social scientists have examined the social impacts of implementation of community-based conservation strategies (Wilkie et al, 2006). Some argue that many of these strategies ignore the primary needs and perspectives of local communities, for example through limiting access to traditionally accessible resources while creating inadequate and poorly targeted benefits (Harper, 2002).

Conservation interventions have increasingly taken on approaches that require wide community engagement. The potential role of local communities has been further explored as communities have been brought into a wider array of management activities, for example in monitoring. Monitoring is an integral and costly component of conservation interventions. Some recent efforts have attempted to engage local communities in monitoring to increase local understanding and ownership of projects. These efforts have been met with mixed success. On the one hand, encouraging or paying communities to monitor the state of local resources has the potential to increase awareness of trends and threats, thus creating a case for community support for interventions (Danielsen et al., 2005). However, monitoring is not a costless activity and in some cases the effort required to gain useful information from monitoring may be extremely high and thus not reasonable for a community to undertake on its own (Hockley et al., 2006). Community monitoring also acts as a tool for compliance and can have an important role in detecting non-compliance with rules. In other cases though if local monitoring regimes do not have the support of regional and national law enforcement, or the wider support of the local community, it may not be sustainable in the long-term. While community-based approaches have the potential to contribute to the success of individual projects, they require a great deal of engagement between the organisation implementing the project and the local community in order to ensure understanding and common objectives.

3.2.2 Payments for biodiversity conservation services in developing countries

PES for biodiversity conservation in developing countries emerged in the conservation literature as a response to perceived limitations in existing strategies for biodiversity conservation (Ferraro and Kiss, 2002). During the second half of the 20th century, protected area management in developing countries followed the North American model of exclusion of local populations through the use of fines and regulations (Brockington, 2002; West et al., 2006). However, this reliance on negative incentives proved both practically and ethically untenable in many situations, as the economic costs of regulation fell on local land-users who tended to be poor and had limited political power (Misha, 2003; Moyle, 2003). The unequal relationship between protected area authorities and local communities has often resulted in tensions that in some cases escalated to violent conflict (West et al., 2006).

In response to these limitations, newer models for conservation emerged, including Integrated Conservation and Development Projects (ICDP), Community-Based Natural Resource Management (CBNRM) and ecotourism, which were all popularised in the 1980s and 1990s. These approaches recognised the limitations of using negative incentives in developing countries both from a community relationship perspective and in terms of capacity to enforce regulations. Like PES, they focus on delivering positive incentives to local communities.

However, ICDPs, ecotourism and CBNRM have also received criticism. ICDPs have been criticised for failing to create incentive structures that result in conservation benefits (Barrett and Arcese, 1995; Newmark and Hough, 2000). CBNRM, where management is devolved to local communities, is limited to locations where sustainable management of wild resources is more profitable than alternative land uses, for example through trophy hunting (Lewis and Alpert, 1997; Lindsey et al., 2007). With respect to ecotourism, the market is limited in any region; the extent to which benefits reach local communities varies dramatically and tourists may only need a very small patch of quality habitat or assured sightings of a few charismatic species to justify their visit (Yu et al., 1997; Bookbinder et al., 2008; Kiss, 2004; Kruger, 2005). Despite these critiques, many authors note that protected areas, ICDPs, CBNRM and ecotourism are appropriate approaches under specific circumstances (Campbell et al., 1999; Wunder, 2000; Newmark and Hough, 2000).

PES offers advantages to some of the limitations presented by these previous intervention types. Like ICDPs, CBNRM and ecotourism, PES relies on positive incentives to influence behaviours. It also creates a strong link between service provision and the benefits that communities or individuals receive, as PES explicitly links the level of payments to conservation outcomes (Ferraro, 2001). PES is able to target incentives to any location of conservation interest (Wunscher et al., 2008), and is not reliant on tourist infrastructure or on local management paying for itself. These advantages have the potential to expand the coverage of biodiversity conservation management. PES can avoid some of the challenges of unrealistic expectations and unfair distribution of benefits experienced by past interventions (Fritzen, 2007), as payments and structure are established in agreements between buyers and providers. Furthermore, monitoring is an explicit component of PES systems and as a result offers a framework for managers to evaluate the impact of the incentives. This can facilitate adaptive management, which academics have long called for to be a part of conservation practice (Holling, 1978; McCarthy and Possingham, 2007).

3.3 Challenges for delivering biodiversity services

Biodiversity conservation represents just one of many types of services that can be provided through PES interventions. Despite the theoretical advantages of PES over alternative conservation intervention approaches outlined above, and support for the approach in the literature (Simpson and Sedjo, 1996; Ferraro, 2001; Pagiola et al., 2002), many of the PES interventions that address biodiversity conservation are either in their infancy (Robertson and Wunder, 2005) or offer biodiversity conservation as a co-benefit alongside another primary service, such as watershed or carbon storage services (Pagiola et al., 2005; Asquith et al., 2008; Engel et al., 2008; Pagiola, 2008). As these cases studies emerge, it is important to examine the challenges facing the implementation of PES for biodiversity conservation schemes. Some potential challenges include a lack of common metrics for biodiversity services, barriers between those providing and using the service, the provider's limited ability to influence service provision, and monitoring costs.

3.3.1 Lack of a common metric

First, the lack of a common unit for measuring biodiversity poses challenges to commodifying biodiversity services (Kosoy and Corbera, 2009). Carbon storage services are traded in the unit of tons of carbon dioxide equivalents, and this standardisation facilitates global trade as it reduces uncertainty in the product and its price for both the buyer and seller (Tietenberg, 2006). The discussion of a unit of biodiversity in PES is complicated by the variety of metrics for biodiversity, including the presence of threatened species or habitats and the number of species found in an area. Neither of these metrics provides a clear unit to measure service provision, nor is an adequate proxy for biodiversity. Despite much theoretical and practical discussion, no consensus methodology for defining a comparable unit of biodiversity for threatened species or habitats (Purvis and Hector, 2000) or quantifying the economic value of that biodiversity unit (Pearce and Moran, 1994; Nunes and van den Bergh, 2001) has emerged. In addition to the conservation of threatened habitats and species, the protection of ecosystems with greater numbers of species has been discussed as an ecosystem service. Diverse communities have not for the most part been shown to provide ecosystem services any more reliably than less diverse communities (Loureau et al., 2001). While high diversity can result in ecosystems that are more resilient to exotic species invasions and other disturbances, the marginal value of individual species varies and the mechanisms behind resilience are poorly understood (Chapin et al., 2000, Hooper et al., 2004). As a result of these limitations for basing payments on a specific unit of biodiversity, PES for biodiversity conservation is typically constrained to focusing on individually negotiated contracts to protect specific populations of interest, as illustrated in the cases of snow leopard (*Uncia uncia*) (Mishra et al., 2003) or sea turtle conservation (Ferraro and Gjertsen, 2009). Alternatively, interventions may focus on habitat conservation in general with the assumption that biodiversity will thus be conserved (Pagiola, 2008).

3.3.2 Spatial dynamics of users and providers

A second challenge to biodiversity PES is the relationship between service providers and service users. Because biodiversity is a public good, it faces the problem of free-riders, and it can be challenging to identify the users of the services (Engel et al., 2008). There is certainly a willingness to pay, primarily at the international level, for the conservation of tropical biodiversity (James et al., 2001). Yet most of this willingness to pay from individual donors is distributed through intermediaries such as international conservation NGOs or government. This creates a challenge of communication and coordination between service providers and users. This contrasts sharply with the development of watershed service payments, particularly in Latin America, which have been facilitated by the proximity of downstream users and upstream providers and the excludable nature of watershed services (Pagiola et al., 2002; Asquith et al., 2008). These spatial dynamics of watershed services reduce coordination costs and facilitate the negotiation of agreements, as well as monitoring and enforcement for compliance (Williamson, 1981).

3.3.3 Ability for providers to influence service provision

Two additional challenges for biodiversity PES are the sellers' capacity to influence the presence and abundance of biodiversity on their lands and the ability of buyers to monitor services provision. While carbon and watershed services can be provided by stationary species, generally plants, much investment in biodiversity conservation is based on efforts to conserve charismatic vertebrate animal species; though habitat conservation is a common means to species conservation. Landowners and managers generally have a greater capacity to control the presence/absence of a habitat, rather than presence/absence and population dynamics of animal species that occupy or pass through their land. Yet, the relationship between biodiversity service provision and habitat may not always be direct, for example if habitat degradation occurs through over harvesting of forest products or selective harvesting of high value timber (Wilson et al., 2005). Thus simply ensuring the presence of habitat may not be adequate for true biodiversity conservation service provision.

3.3.4 Ability of users to monitor service provision

Due to the difficulty and costs involved in monitoring abundance of rare mobile species, cost-effective options for monitoring biodiversity are needed (Hauser et al., 2006; Field et al., 2007). One option is to make payments based on the quality and availability of habitat, an indirect indicator of success. Indeed, watershed and carbon payments are most frequently linked to land cover change that occurs over a relatively large scale of cleared habitat. These changes can be monitored with confidence using transects or remotely sensed imagery. Because habitat may be devoid of much of its biodiversity well before observable clearance occurs (Redford, 1992; Peres, 2000; Wilson et al., 2005), indirect measures must be coupled with some form of on the ground species-monitoring programme (Pagiola et al., 2004; Brooks et al., 2004). Alternatively, PES systems for biodiversity conservation in Europe and the United States have relied on rewarding easily and cheaply monitored management behaviours that promote biodiversity, such as leaving fields fallow for migrating birds rather than attempting to monitor the birds themselves (Parks and Shorr, 1997). Similar examples are emerging in developing countries where individuals are paid for protecting successful nesting sites (Clements et al., 2009; Ferraro and Gjertsen, 2009). Any such approach that reduces monitoring costs will thus increase the viability of PES, but will lead to a more indirect relationship between the quantity being monitored and the biodiversity that is the ultimate object of the conservation intervention.

3.4 Challenges to implementing PES in developing countries

A few PES interventions in developing countries have achieved widespread notoriety, such as the Costa Rican Pago por Servicios Ambientales (PSA) programme (Pagiola, 2008), and Pimampiro and PROFAFOR in Ecuador (Wunder and Alban, 2008). However, there has been some concern that many PES initiatives in developing countries remain incipient (Wunder and Ibarra, 2005; Wunder, 2007) and that there has been little dialogue between initiatives in developing and developed countries (Engel et al., 2008). The factors that challenge the successful implementation of PES in developing countries are for the most part the same institutional factors that influence the success of many conservation interventions in developing countries, particularly where resources are common property. These include issues such as a lack of property rights, inequitable distribution of benefits among and within communities, hesitancy to distribute cash benefits, weak institutions, and addressing drivers of poverty. These factors create challenges to establishing a PES intervention and influence the extent to which the approach is likely to act as an incentive to service providers.

In developing countries, natural resource managers frequently face the challenge of improving governance of areas that are either *de facto* or *de jure* managed by small groups or communities (Ostrom, 1990; Schlager and Ostrom, 1992; Balland and Platteau, 2006). One challenge of situations where those holding *de facto* property rights differ from the *de jure* property rights is that the identification of the service provider is not necessarily simple. Those with legal rights to the land (often government) may be entirely different from those who are managing land (often local people with customary rights over the land) (Mendelsohn, 1994). If land is owned by a government unable to effectively control how local people use it, it may not be clear with whom negotiations should proceed when developing a PES scheme (Balland and Platteau, 2006; Chapter 2).

Secondly, the distribution of benefits within the community influences whether PES systems can work as a behaviour changing incentive. Costs and benefits of engaging in a conservation intervention are rarely equitably distributed throughout a community (Berkes, 2004; Chapter 5), and in some cases the conservation of biodiversity may be in direct conflict with livelihood-sustaining activities, for example hunting when bushmeat is the primary protein source for a community (Godoy et al., 2000; Fa, 2003), or when forest goods are critical for seasonal food shortages (Cavendish, 1999). Individuals within communities have varying opportunity costs and it is not clear whether PES systems have a solution for avoiding the local distributional issues that have hampered many ICDPs (Peters, 1998; Hughes and Flinton, 2001).

The form of the incentive can also impact the efficacy of PES interventions. There has been hesitancy to distribute cash benefits from development projects in developing countries due to the perceived dangers of corruption (Wunder, 2005). Traditionally, benefits from conservation interventions have been in-kind through development projects or physical goods of the community's choosing. The use of development projects as contingent payments may not be appropriate, as resentment can develop in situations where a community believes that funds or projects are a right, not a reward (Frey and Oberholzer, 1997). As a result, for PES to be successful in developing countries buyers must move beyond the paternalistic reluctance to distribute cash benefits (Wunder, 2005).

Weak institutions at local, regional or national levels can undermine all conservation interventions. Institutional failures have been blamed for weak enforcement of protected area regulations (Moyle 2003), poor development and conservation gains in ICDPs (Barrett et al., 2001), and collapse of sustainable management in some CBNRM programmes (Campbell et al., 2001). Barrett et al. (2001) suggest that institutions managing for biodiversity conservation need the authority and ability to restrict access and use; the wherewithal to offer incentives; the capacity to monitor ecological and social conditions; and the flexibility to alter incentives and rules. PES places the responsibility for managing land on the service provider. Management support beyond monitoring for payments is not an explicit component of the PES framework, but may be required in areas with weak institutional capacity. As a result, the capacity for local institutions to restrict access and use may be one of the most important prerequisites for successful PES implementation. This challenge highlights that in locations where PES is implemented following the failure of other conservation paradigms due to weak institutions, success may be extremely hard to achieve.

The principal goal of PES is not poverty reduction but rather, natural resource management (Pagiola et al., 2005). Yet the potential role of PES as a tool in poverty reduction has received a great deal of attention (Landell-Mills and Porras, 2002; ABCG, 2005; Iftihkar et al., 2007; Wunder, 2008). Within a local context, those who control large portions of land are likely to be the wealthier or more socially dominant individuals. Early PES systems in the tropics, like EcoMarkets (Sills et al., 2005), were frequently administered over a large area and were open to any landowner whose land met basic criteria. This approach favoured wealthier, well-educated landowners who could absorb transaction costs and were privy to the advertisement of the programme (Miranda et al., 2003; Zbiden and Lee, 2005). An example of this is the way in which land title requirements have limited the extension of PES to poorer farmers with customary rights in Costa Rica (Pagiola, 2002). In this example, oversubscription and limited funding led to the prioritisation of land for PES based on quality of services and cost of provision (Balmford et al., 2000; Wilson et al., 2005). This focus of PES interventions on service provision limited the potential for prioritisation of participation of poor individuals within the scheme. Nevertheless, programmes have attempted to make PES pro-poor (Sherr et al., 2004; Iftihkar et al., 2007), or explicitly target PES towards the rural poor by focusing on particularly poor regions (Rewarding Upland Poor for Environmental Services-RUPES; http://rupes.worldagroforestry.org/).

This review suggests that the challenges facing implementation of PES in the context of biodiversity conservation in developing countries are not insurmountable, but they demonstrate that PES does not offer a panacea for natural resource management. Instead, like past conservation interventions, PES is most appropriate under specific circumstances.

3.5 Case study: Menabe, Madagascar

The case study that I use in this thesis is a long-standing PES intervention strictly for biodiversity conservation in a developing country (Durrell Wildlife Conservation Trust 2004-2008). I use this case study to explore some of the key issues that face those implementing PES, as outlined in sections 3.3 and 3.4.

3.5.1 Ecology of the Menabe

The Central Menabe, hereafter Menabe, region of Madagascar extends approximately 75km north to south between the Morondava and Tsiribihina rivers and approximately 40km from the coast to the easternmost patches of dry deciduous forests. The forests of the Menabe are an international conservation priority (Kremen et al., 2008). They provide the sole habitat for three endangered endemic vertebrate species: the flat-tailed tortoise (*Pyxis planicauda*), the giant jumping rat (*Hypogeomys antimena*) and Berthe's mouse lemur (*Microcebus berthe*) (IUCN, 2009), and critical habitat for numerous other endemic species of plants and animals, many of which are threatened.

The development of expansive sisal and sugar plantations in the 1970s, as well as prior forestry concessions, led to large-scale immigration of individuals from the Tandroy and Korao ethnic groups to the region. This, alongside the needs of the local Sakalava ethnic group, has resulted in substantial forest clearance (Scales, 2008). Agricultural expansion linked to inhabitants of the sisal plantation still threatens to split the Menabe forest into isolated blocks that could result in genetic isolation for some species (L. Schäffler, personal communication). At the southern extent of the forest, immigration related to the development of the industrial sugar plantation, Siranala, and pressure from the urban population centres of Morondava and Mahabo has led to the loss of tens of thousands of hectares of forest since 1970 (Scales, 2008). In the late 1960s and mid-1980s international oil companies searched for oil in the Menabe, establishing a grid of roads that made the interior of the forest accessible to selective harvesting of wood, grazing of animals, and hunting (Genini, 1996; Sandy, 2006). Despite these threats, approximately 100,000 hectares of contiguous forest remains in this rural region, which is populated by almost 5,000 people (Figure 3-1).

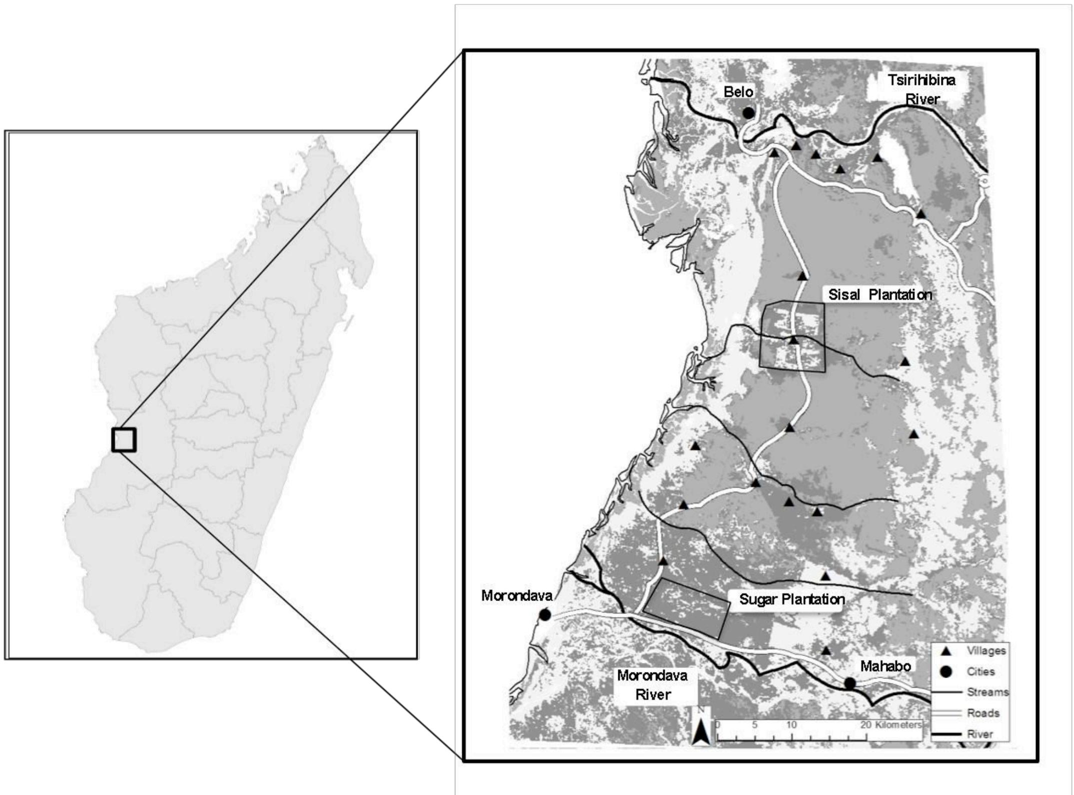


Figure 3-1 Map of the Central Menabe.
Light gray represents forested areas as of 2000. Dark grey represents area deforested between 1970 and 2000. The map also presents regional cities and communities, water sources, transportation routes and two large plantations (Data from Harper et al., 2008)

3.5.2 Livelihoods

Agriculture is the basis for local livelihoods in the Menabe, and the principal driver of forest loss. In communities with access to streams or rivers to irrigate fields, rice is the primary subsistence crop. On the banks of the Tsiribihina River, communities also grow cash crops of beans during the dry season when floodplains are exposed. There are few options for acquiring new wet fields along water bodies in the Menabe, as most of these areas have already been cultivated (personal observation). As a result, most interest in acquiring new land is focused on expansion of fields into the dry deciduous forests for maize for subsistence and groundnuts as a cash crop.

In the expansion process, individuals clear small trees and vines from June to October, and burn the dried debris to release nutrients into the soil. Land that is undergoing this slash and burn process is known locally as *hatsake* (Genini, 1996). Individuals plant maize by digging small holes with spades in the *hatsake* during the first year before the start of the rainy season in December. In subsequent years, individuals continue to cut and burn the remaining trees and eventually remove the stumps of most trees (Scales, 2008). In some communities, when the stumps are removed, farmers prepare the soil with ox-pulled plough. Historically, subsistence farmers would abandon their property after two to three growing seasons and continue to expand into mature forests due to rapidly falling maize yields. The abandoned land would then start a secondary succession process, which is dominated by a single species, *Ziziphus mauritania*, locally known as *monkanazy* (Genini, 1996). Unfortunately, there is no evidence that the *monkanazy* forests revert to forests with species composition similar to primary forests (V. Razafintsalama, personal communication).

Figure 3-2 The process of forest clearance for agriculture in the Menabe.
a. During the first year, farmers clear the land and burn to release nutrients to create *hatsake*, leaving some large dead trees standing. b. During subsequent years, the land is cleared further. c. Maize is planted in individual holes among the stumps. d. The remaining stumps are burned from the ground. e. After three or four years, most stumps are removed, the land can be ploughed in rows, and it is known as *monka*. f. Only baobab trees (*Adansonia sp.*) survive in fully cultivated *monka*.

Land that is fully cleared for agriculture is known as *monka* (Figure 3-2). The Cooperation Suisse has been successful in many communities in introducing groundnuts as a crop to follow maize, thus keeping *monka* productive, aiming to reduce the desire of farmers to expand into new forest (Laurent, 1996; P. Raonitsoa, personal communication). Groundnuts are grown as a cash crop rather than for subsistence, and can be grown over many successive years on a single piece of land. A number of other products are grown for subsistence on this land, most notably cassava, a starchy tuber.

In all communities, individuals use the forest for supplemental income and personal consumption. Uses include timber for building, hunting, and collection of non-timber forest products (NTFPs; Dirac, 2009). The collection of honey and hunting of tenrecs (*Tenrec ecaudata, Echinops telfairi* and *Setifer setosus*) and lemurs (*Propithecus verreauxi, Eulemur fulvus* and *Lepilemur ruficaudatus*) are important for a large portion of the population particularly during the lean season before the agricultural harvest. These activities also have cultural significance in local rituals and traditions (Fauroux, 1997). Hunting of lemurs however is taboo for individuals from the Tandroy ethnicity (Lingard et al., 2003).

3.5.3 Conservation history

International conservation and development organisations have been active in the Central Menabe since the late 1970s (Cabalazar, 1996). The largest efforts were made by the Cooperation Suisse, which focused on developing national capacity in sustainable forestry through a forest training centre in the region. However, as the extent of deforestation and impacts of agricultural expansion became apparent, the efforts of the Cooperation Suisse began to centre on sustainable development and the reduction of deforestation drivers (Laurent, 1996).

Under Malagasy law, all forested land belongs to the government (Raik, 2007). Legal land tenure for individuals is established in the Menabe by applying and paying for forest clearance permits from both local and national authorities. Rather than participating in this process, most individuals rely on customary tenure, and only apply for legal title after they have cleared the land and cultivated it for a number of years (J. Randrianarijaona, personal communication). While this is technically illegal, there is very little enforcement.

In the mid-1990s, as Madagascar began receiving international support for its three phase National Environmental Action Plan, it became apparent that its focus on centrally run protected areas was not adequate to ensure sustainable natural resource management (Raik, 2007). As a result, the government began to support community-based resource management policy (Antona et al., 2004). This led to the creation of the Gestion Locale Securisé (GELOSE) law in 1997, which gives management responsibilities to local associations (communauté de base, or COBA) for defined community forest areas. These local forest associations are then responsible for setting and enforcing rules alongside the right to use managed areas for a variety of subsistence activities (Antona et al., 2004). Under this scheme, rules and local capacity for enforcement are written into traditional customary law, known as *dina* (Henkels, 2001). Early experience with GELOSE proved to be overly bureaucratic, and in 2000, the Gestion Contractualisée des Forêts (GCF) law was introduced to offer a simplified management transfer process. These transfers under GELOSE/GCF are legalised through contracts between the state and forest associations, and the process has often been supported by international conservation organisations (Raik, 2007). While GELOSE/GCF contracts give communities the rights to subsistence use of forest resources, it does not transfer property rights, and the contracts must be renewed every three to ten years (Kull, 2002). Thus, while some subsistence use rights are legally transferred, the laws have placed potentially costly responsibilities on communities for monitoring and enforcement, which may exceed local capacities (Antona et al., 2004).

Durrell began working in the Menabe in 2000 and has focused on community-based conservation, conservation education and population monitoring of flagship species. Durrell has facilitated the transfer of management responsibility of government forest to local forest associations. These community-managed forests are split into multi-use and strictly protected areas that are planned ultimately to act as a buffer zone for a new Central Menabe Protected Area (P. Raonintsoa, personal communication). Though many communities where Durrell works have not formally completed their management transfer, the forest associations have established their local rules through forest management documents. These management documents define permit fees, prohibited behaviours and punishments.

In each of the communities, the forest associations that manage community forests are made up of local members, who must pay an annual fee and are then given use rights to the multi-use forest. Local association boards take responsibility for monitoring their forest, liaising with Durrell and the national forest service, and distributing permits for activities in the multi-use forest. Despite the existence of these management transfers in over 400 communities across Madagascar, few communities have actually implemented their management plans (Antona et al., 2004; Raik and Decker, 2007).

3.5.4 Durrell scheme

In order to create incentives for active local community forest management in the Menabe, in 2003 Durrell began implementing annual payments to communities based on the state of biodiversity indicators and threats within protected forest, as well as forest governance indicators. Communities receive payments from Durrell, which they use to purchase in-kind incentives such as diesel generators, community cooking equipment, bicycles and supplies for community buildings (Durrell, 2004-2008). These payments fulfil the criteria for PES under the framework in Chapter 2, as they are conditional on the state of an indicator and deliver positive incentives. They also fulfil the criteria for PES as defined by Wunder (2006), as they represent *voluntary* transactions between a *buyer* (Durrell) and *seller* (communities) *contingent* on provision of a *well-defined service* (governance and presence of biological indicators). Individuals within communities can choose their level of participation by becoming forest association members, actively participating in management or simply by using the in-kind incentives that Durrell provides. While the scheme is not voluntary at the individual level, communities decide whether to participate. I thus consider the intervention to be a PES at the community level. Only communities that are in the process of establishing community forests under the GEF or GELOSE contracts participate in the scheme, and not all communities in the Menabe have been approached to participate in either the community forest contracts or the PES scheme (F. Rakotombololona, personal communication).

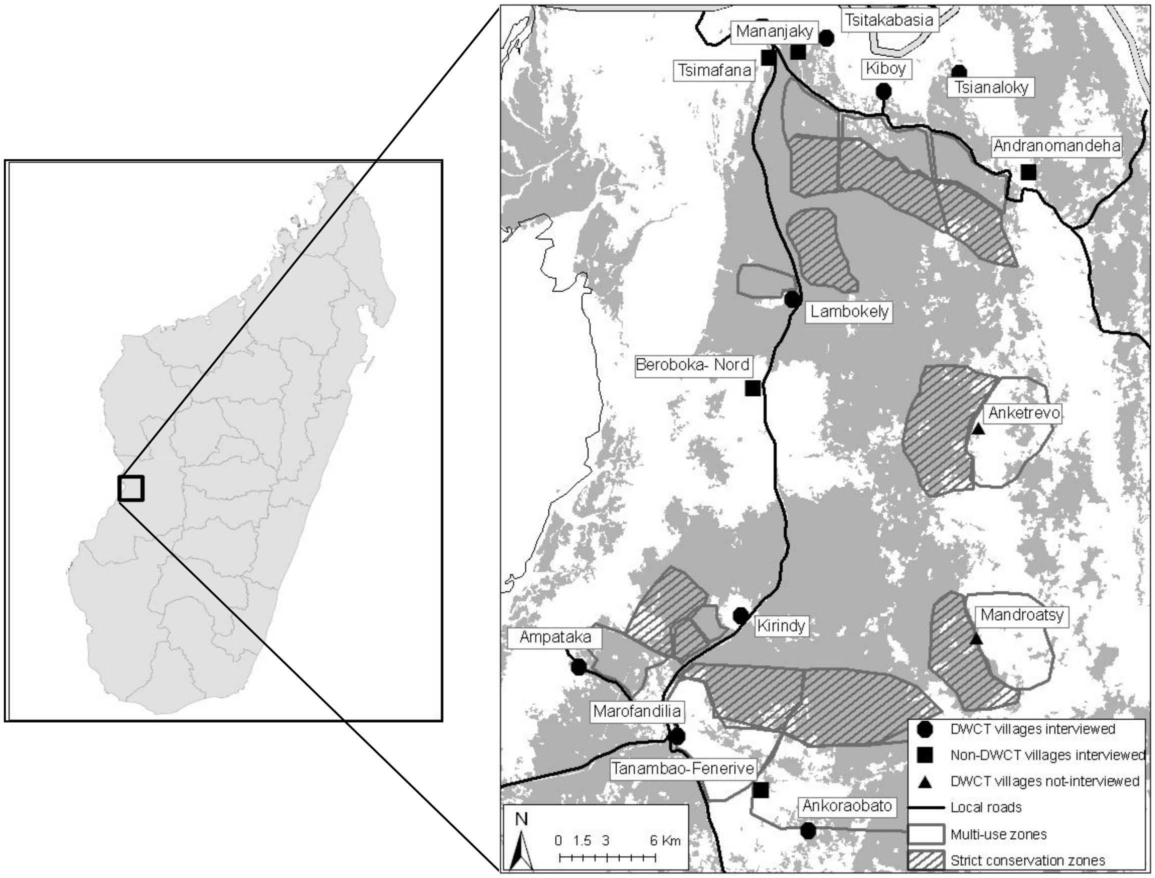


Figure 3-3 Map of the surveyed PES participating and non-participating communities.

The scheme began in 2004 with three communities in the north (Tsitakabasia, Kiboy and Tsianaloky) (Figure 3-3). In 2005, it expanded to two additional communities in the south (Marofandilia and Ankoraobato). In 2007, five more communities were added (Lambokely, Kirindy, Ampataka, Anketrevo and Mandroatsy).

Payments are contingent on both the state of the strictly protected forest (the number and abundance of species of interest) and on actions that impact the system (forest governance indicators and monitored threats), scored during an annual assessment carried out by Durrell in collaboration with community members (indicators are in Appendix 1).

There are a number of forest-use activities that can impact a community's payment, either by altering habitat or posing threats to species. These include clearing forest for grazing, agriculture or paths, selective cutting of trees for timber or for accessing honey, and hunting of lemurs. Each forest-use behaviour has varying legality under national laws. If individuals are caught engaging in some of the activities inside the strictly protected forest they could face local and national fines or even prison sentences (Chapter 4).

Table 3-1 Scores and total award for 2008 DWCT PES for communities surveyed.

	Ampataka	Ankoraobato	Kiboy	Kirindy	Lambokely	Marofandilia	Tsianaloky	Tsitakabasia
Positive biodiversity points	24	22	39	17	30	11	40	25
Positive governance points	11	3	16	15	15	24	18	8
Negative threat points	-10	-18	-14	-4	-9	-25	0	0
Total score	25	7	41	28	36	10	58	33
Size of forest (ha)	1608	1370	2214	372	1912	1575	1543	1136
Percent of total forest area	0.10	0.09	0.14	0.02	0.12	0.10	0.10	0.07
Award amount ($)[a]	741	177	1673	192	1269	290	1649	691
Money rank	5	10	2	9	4	7	3	6
Score rank	7	10	3	6	4	8	1	5

[a]Awards are calculated by creating an adjusted score based on the proportion of total forest area controlled by each community in the scheme. The total award is then divided among communities based on the adjusted scores.

In the scoring for payments, positive indicators from the state of the system and forest governance are scored alongside negative indicators of threat to create an aggregate score for each community. The scores are subsequently adjusted proportional to the size of each forest to reflect the greater amount of service provided by larger forests (Table 3-1; Durrell 2004-2008). A particularly unique component of the system, in relation to other

PES schemes, is that the payment levels for each community are relative to how well the other communities in the scheme perform. This reflects the assumption that competition among communities will act as a positive incentive (J. Durbin, personal communication).

At present ~US$9,500 is distributed between the ten participating communities annually. Payment amounts and scores are announced in each community through an annual "forest party" sponsored by Durrell. Payments are distributed to the communities via the local forest associations and each community has a different procedure for sharing the benefits with the wider community (Figure 3-4).

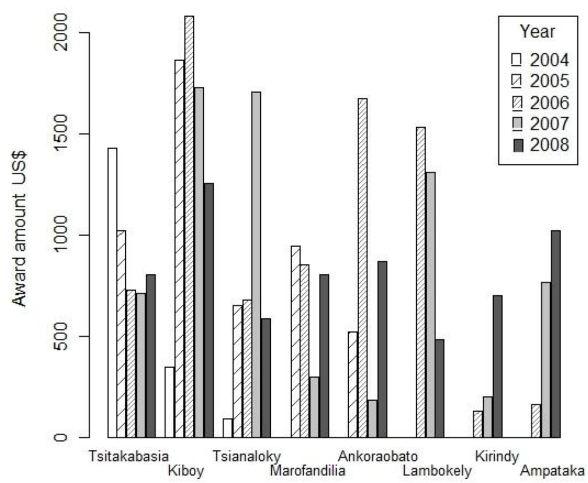


Figure 3-4 **Awards for interviewed communities in Durrell's PES from 2004**-2008. (From Durrell 2004-2008).

Though the monetary value of the payment to each community is announced annually, it is not distributed in cash, but instead used to purchase in-kind incentives. The board members of each community forest association, with the permission of at least 80% of the association, decide what they would like to purchase with the Durrell payment. The subsequent distribution of these items differs between communities. In some communities, direct access to the items purchased is open to the entire community; while in others, they are only available to association members or for the facilitation of community-wide events.

The costs of engaging in the intervention vary by individual and by community. Individual costs vary to the extent that individuals had previously been involved in controlled activities. Each community association has an annual membership fee. Membership gives individuals the right to apply for permits to use resources in local multi-use managed forest and in some communities, membership is a prerequisite for taking advantage of Durrell's incentives. Though Durrell did not directly introduce new rules or laws into the community forest management system, their independent species monitoring programmes unrelated to the PES throughout the region have created a perception of enforcement of existing laws that was not previously present, thus placing an increased perceived cost on non-compliance.

3.6 Data collection methodology

3.6.1 Field team and approach to data collection

We (my field assistants and I) spent two field seasons in the Menabe from March to June 2007 and October to May 2008. Permission to conduct research in Madagascar was granted by the Ministry of Environment, Water, and Forests in Antananarivo, Madagascar with institutional support from Durrell. We worked in eight of the ten communities that are participating in the PES, as well as five communities that are not participating (Table 3-2). We collected social data in 13 communities and ecological data in 5 community forests (Table 3-3, map in Figure 3-3). I had previously spent thirteen months living and performing research in rural communities in Madagascar over the course of five years, and started my research with basic language skills and a strong knowledge of culture and sensitivities. The first field season gave me a chance to improve my language skills in Malagasy. I am fluent in French and my primary field assistant is fluent in Malagasy, French and English.

Table 3-2 Description of key features of the 13 interviewed Menabe communities.
Adult population estimates from participating villages are from Durrell PES documents, estimates from non-participating villages were based on educated guesses.

	Community	Ethnicity	Age of intervention (years)	Age of community (years)	Individuals interviewed	Estimated adult population	Percent of adult population interviewed
1	Tsitakabasia	Sakalava, Korao	5	75+	79	266	27
2	Kiboy	Sakalava, Korao	5	75+	81	516	16
3	Tsianaloky	Sakalava	5	75+	82	381	22
4	Marofandilia	Korao, Tandroy, Sakalava	3	~75	82	193	42
5	Ankoraobato	Korao, Tandroy	3	~75	83	271	31
6	Lambokely	Tandroy	2	15	97	272	36
7	Kirindy	Tandroy	2	25	63	74	85
8	Ampataka	Sakalava, Tandroy	2	75+	84	366	23
9	Andranomandeha	Sakalava	Na	75+	59	~500	~11
10	Tsimafana	Sakalava	Na	75+	40	~1000	~4
11	Beroboka-Nord	Tandroy	Na	~35	36	~200	~18
12	Mananjaky	Sakalava	Na	75+	60	~150	~40
13	Tanambao-Fenerive	Tandroy	Na	~20	18	~200	~9

3.6.2 Communities

Ethnicities and Geography

I interviewed six villages in the northern region of the Central Menabe, three communities that participate in the Durrell interventions (Tsitakabasia, Kiboy and Tsianaloky) and three communities that do not participate (Manajaky, Tsimafana and Andranomandeha). Each of these communities is primarily composed by individuals from the Sakalava ethnic group with a small number of immigrants from the Tandroy and Korao ethnic groups. The northern communities of the Menabe have long been related to the historical capital of the Sakalava ethnic group in Belo-sur-Tsiribihina (Belo). Local histories describe the establishment of each of the communities along the riverbank and subsequent movements over the past century. For example, the name of Tsitakabasia (where the guns will not reach) dates to the French colonization of Madagascar when the community fled from the village to an overlook above the river, as the French forces advanced. The communities are each located in close proximity to the Tsiribihina River so that during the dry season when the river's floodplains are exposed, most families can grow cash crops of beans. In the wet season most families grow subsistence maize and cassava inland from the river on hatsake. More recently, with the arrival of Tandroy immigrants, many families have begun to grow groundnuts during the rainy seasons on these inland dry soils following two years of growing subsistence maize.

Table 3-3 Development indicators of communities interviewed in the study.

	Community	Ethnicity	Rice	Maize	Groundnut	Beans	Primary School	Hospital	Water Course	On Main Road	Well (2008)
1	Tsitakabasia	Sakalava, Korao	high	low	Low	High	x		x		x
2	Kiboy	Sakalava, Korao	high	low	Low	High	x		x		x
3	Tsianaloky	Sakalava	high	low	Low	High	x	x	x		
4	Marofandilia	Korao, Tandroy, Sakalava	medium	medium	Medium	None	x	x	x	x	x
5	Ankoraobato	Korao, Tandroy	medium	medium	medium	None	x		x		x
6	Lambokely	Tandroy	none	high	High	None				x	
7	Kirindy	Tandroy	none	high	High	None					x
8	Ampataka	Sakalava, Tandroy	none	low	Low	None	x		x		x
9	Andranomandeha	Sakalava	high	low	None	High	x		x	x	x
10	Tsimafana	Sakalava	high	low	Low	High	x	x	x	x	x
11	Beroboka-Nord	Tandroy	none	high	High	None				x	
12	Mananjaky	Sakalava	high	low	Low	High			x		x
13	Tanambao-Fenerive	Tandroy	none	high	High	None					

I interviewed four communities that are primarily composed of individuals from the Tandroy ethnic group (Beroboka-Nord, Tanambao-Fenerive, Lambokely and Kirindy). Each of these communities has a similar community structures and farming approach. None has a community centre, but rather individual's homes are distributed throughout their fields. Many households stay in these communities only during the wet season, and move to seasonal wage employment in other regions of Madagascar during the dry season, or in some cases farm their own land in communities on the Tsiribihina or Tomisy Rivers (Scales, 2009). Each of these communities relies on maize and cassava for subsistence and groundnuts as a cash crop. None of these communities has access to irrigation and thus their agricultural opportunities are limited.

The Tandroy communities were established due to migration related to the establishment of a failed sisal plantation in the 1960s and 1970s. The migrants and plantation owner (DeHeulme family) initially established a Beroboka within the sisal plantation. With the abandonment of the production of sisal by the Deheulme family in the 1970s, the town of Beroboka continued to grow as new Tandroy immigrants arrived and expanded their maize production into the primary forest surrounding the plantation boundaries. Beroboka-Nord represents the forest frontier in the area that had been cleared over the past 50 years. In contrast, Kirindy and Lambokely were each established by families from Beroboka who applied for agricultural permits within the primary forest to the north and south of Beroboka in the 1980s and 1990s. Small areas were granted to a few families. Due to limited enforcement of permits over the past two decades, a large number of immigrants relocated to Kirindy and Lambokely and expanded their agricultural lands into the primary forests surrounding the original claims. This expansion has continued until the present and halting future expansion of these fields has been a major priority of conservation organizations and to a much lesser extent the national government.

In contrast to this expansion into primary forest, the community of Tanambao-Fenerive was established by Tandroy immigrants in the early 1990s seeking to grow groundnut crops on land that had been previously abandoned by individuals who had grown maize and exhausted the land's fertility. Because of this dynamic of individual families reclaiming dispersed abandoned land, Tanambao-Fenerive is much less of a distinct community than the other communities. Tanambao-Fenerive does not have well-defined boundaries and as a result, they are administratively linked with the Korao community of Ankoraobato.

Each of these Tandroy communities is directly adjacent to the National Route 8 that runs between Morondava and Tsimafana. None of these communities has a primary school due to these dynamics of seasonal migration and recent establishment. Instead, children tend to walk to the closest community with primary education or live with relatives in communities with schools.

Three additional communities, Marofandilia, Ampataka and Ankoraobato, in the southern portion of the Central Menabe share a common history due in part to the river Tomisy and a former forest concession. Marofandilia was established as a logging community for a forest concession along Route National 8 in the early 1900s, and as a resting point for Sakalva foot travellers. The concession attracted Korao immigrants and given the community's location along the Route National, it has attracted a mixture of Sakalava, Tandroy and Korao ethnic groups. However, following the closing of the concession in the 1950s, individuals focused on irrigated agriculture along the Tomisy River. The river inexplicably dried between 1960 and 2004, and the community members who remained in the area altered their livelihoods to be primarily based on slash and burn agriculture for maize. This maize boom was accompanied rise in the international price of maize, which led to extensive deforestation. Since the return of water to the Tomisy, many individuals have started irrigating rice again and conversion of forest has slowed.

Ankoraobato was established by immigrants from the Korao (or Antesaka) ethnic group who came to the Menabe in the 1920s to work in the sawmill in Marofandilia. Individuals in Ankoraobato have a stronger connection with other Korao communities to the south, than they do with the other Central Menabe communities in this study. Ankoraobato sits alongside the Tomisy River upstream from Marofandilia (Lindenmann, 2008).

Ampataka is a unique community in the Durrell Scheme, as community members rely on fishing on the ocean and collection of crabs in mudflats in addition to limited agriculture of cassava. This represents a change over the past decade due to a growing demand for fish and crabs from Morondava (Scales, 2007). The community is close to the Tomisy River, and grows some crops along its banks, but generally individuals focus on fishing and crabs, which are collected biweekly by collectors from Morondava.

Accessibility

Each of the communities studied are accessibly by road from Morondava to the south. Due to Morondava's port and road access to the capital, most trade in the Menabe passes through Morondava. The six northern communities on the banks of the Tsiribihina River also have limited connection to trade along the river and the former capital of the Sakalava Kingdom in Belo-sur-Tsiribihina (Belo). Route 8 from Morondava to Tsimafana is the main artery for trade in the Central Menabe and five of the communities (Marofandilia, Kirindy, Lambokely, Beroboka-Nord and Tsimafana) are located directly on this road. Daily bush taxis run between Morondava and Tsimafana throughout the year (90km, 3-12 hours), though the road can be impassable following rainstorms.

The additional northern communities are connected to Tsimafana by a trade road from to the river crossing town of Serinam (40km), which has sporadic bush taxis. Of these northern communities, only Andranomandeha is directly on the road. Mananjaky, Tsitakabasia, Kiboy and Tsianaloky are each located on the riverbank within 2km of the road.

In the south, Ampataka is connected by a road to Marofandilia, though individuals often travel by *lakana* (traditional canoes with sails) to Morondava (20km). Similarly, a rarely travelled road connects Marofandilia, Tanambao-Fenerive and Ankoraobato. Tanambao-Fenerive and Ankoraobato have established trade connections and a number of roads with the town of Mahabo to the south.

Education and health

There are a number of primary schools in the Central Menabe. Of the intervention communities, only Kirindy and Lambokely do not have elementary schools. In the control communities, Mananjaky, Tanambao-Fenerive and Beroboka-Nord do not have elementary schools, in part due to their small size and proximity to communities with schools. For secondary school, students either have to travel to and live in Morondava or Belo. In terms of access to health, both Tsimafana and Marofandilia have small functioning hospitals, though sick individuals often travel to Morondava or Belo for treatment. A European Union funded hospital was also completed in Tsianaloky in 2007, but by 2009, neither the community nor government had funded medical staff or equipment.

3.7 Data collection and statistical analysis

3.7.1 Approaches to data collection

While most of the social data I report on in the chapters is based on conventional surveys with individuals, a great deal of my understanding of the communities studied comes from participatory rural appraisal (PRA) principles and techniques (Chambers, 1994a). Researchers and practitioners in sociology, anthropology, health, development, ecology and natural resource management have used PRA to advance their work (Catley and Aden, 1996; Beebe, 1995; Wright and Walley, 1998). My initial approach to working with communities was based on the PRA principles of participating in local activities, using flexible methods, learning on site, and breaking the stereotype of research that finds answers and then moves on (Chambers, 1994b).

The quality of my social data was reliant on individuals telling the truth about illegal and controlled behaviours. Therefore, it was critical to earn the trust of local communities. During the first field season, I spent 8 to 14 days on each field visit with a single field assistant living in a central community in the north and south aiming to become part of the local community. On these initial visits, we explained why we were studying in the area to as many individuals as possible, but did not ask questions related to our research to most individuals.

Table 3-4 Social and ecological data collected during fieldwork.

Dataset	Individuals or groups interviewed	Comments
Social data		
Structured and semi-structured interviews on forest use behaviour and impact of incentives	849	13 communities- 8 PES participating and 5 non-participating
Impact of distribution of incentives	645	8 PES participating communities
Semi-structured interviews with members of forest associations	55	8 PES participating communties
Semi-structured interviews with individual imprisoned for agricultural expansion	11	All from Lambokely
Forest wood use interviews*	25	From Tsitakabasia and Kiboy
Daily tracking of honey and tenrec collection*	18	Tracked over 2 months in 2 communities.
Ecological data		
Forest PES transects (Species and threats)	180 transects	From 5 strictly protected community forests
Tree survey transects*	420 transects	From 2 communities, multi-use and strictly protected forests

*These datasets were not used in the body of the dissertation.

Initially, we travelled by local bush taxis, bike or ox cart. We based ourselves in central communities and would take frequent trips to neighbouring communities to meet with community elders and forest association members. We accompanied friends on daily trips to the field or forest and participated in manual labour. During breaks or less physical activities we would interview these individuals informally, using traditional PRA methods, such as drawing community maps in groups, creating community timelines, developing seasonal calendars and ranking the importance of various livelihood options (Chambers, 1994a; Freudenberger, 1999). These learning tools gave way on subsequent visits to testing question formulations for our structured questionnaire. We attempted to visit each village frequently enough so that our presence/arrival was not an “event.” In the evenings we interviewed community elders regarding local history and landscape changes.

Throughout the research we performed semi-structured interviews with forest association board members, forest monitors and village elders. These interviews were not analysed statistically, but rather they informed the research questions, hypothesis and subsequent discussions. We also interviewed 11 individuals from Lambokely who had been imprisoned for expanding their land into the forest without a permit. In addition to generally informing out understanding of the case study, I used these responses to validate the self-report data.

During the second field season, my field assistant and I piloted our structured surveys in Tsitakabasia. Interviews from individuals in the pilot surveys were not subsequently used. During the pilot surveys, I compiled a list of the most common responses to open questions and categorised answers in the main survey. When individuals gave multiple responses to open questions, the interviewer asked for further clarification on primary and secondary responses. The detail of the approaches used and questionnaires is given in the relevant chapters.

We then assembled a team of four new field assistants for the social surveys, who were all from the Menabe, spoke the local Sakalava dialect, and all were accustomed to working in rural communities. We trained for two days in Morondava and then began field interviews. We rented a 4x4 vehicle to drop us off in each village and pick us up four to seven days later.

We also hired local community members each day to help facilitate our interview process. We did not choose individuals with authority who might influence responses, but rather selected young well-respected men with whom we had built a rapport. These individuals acted as "gatekeepers" by introducing us to the interviewees (Bernard, 2002). We selected interviewees by walking through the village and agricultural fields, and choosing a different section of village or fields to interview each day. All sections of the village and fields were covered during the research period. Within a section, individuals were approached opportunistically, with the aim of covering a broad range of age and social statuses. Males were preferentially selected, as they engage in forest-use activities within the community managed forests more frequently than women. As our survey teams always travelled in twos, one member would engage in labour to assist the interviewee while he/she responded to questions. This would frequently take the form of picking or preparing agricultural products, such as maize and groundnuts. We gave individuals who participated in interviews a small gift of coffee and sugar, or beans as thanks. While the impacts of compensation for participation in interviews have been debated in the literature (Singer and Kulka, 2002), there was an expectation of compensation in many of the communities and as a result small gifts were socially appropriate.

3.7.2 Biases and validation

Numerous forms of bias can enter into data collection. I was initially concerned that individuals would identify my research with Durrell or another NGO working in the area. As a result, we declined transport with NGOs or government, and took public transport, bicycles or walked when possible. These actions allowed many individuals in the communities to identify us as independent researchers, and presumably reduced bias in their responses. In order to minimise the impact that our relationship with Durrell had on the responses to questions, we spent a significant amount of time in each community getting to know community members and helping individuals understand the purpose of my research in order to gain trust and limit bias. Nevertheless, some individuals would inevitably look upon our interviews with suspicion. Yet this suspicion cannot be solely attributed to my team's relationship with Durrell. Suspicion of outsiders is common in the area and while Durrell's team visits each community in the intervention a number of times throughout the year, these visits are typically for a half day at a time and are very formal. In contrast, my interviews were extremely informal, often in the agricultural fields and I tended to stay in each community for a week at a time. These dynamics further distanced my research from Durrell's day-to-day work in the region.

My relationship with Durrell staff members and their institutional hosting of my research in Madagascar posed no conflict of interest in my critical evaluation of their interventions. Indeed, members of Durrell's global and national leadership teams made it clear to me that they wanted the brutal truth and they were open in describing both the successes and failure that they perceived in the intervention. I do not believe that I was biased in my analyses and conclusions due to a desire not to offend my host institution. This does not mean that I took a confrontational approach, but rather I used constructive language to level critiques and suggestions. This approach will hopefully give my recommendation more weight within the organization. It does however stand in constrast to a recent MSc thesis from the Menabe, which criticized the work of Durrell as well as other NGOs in the region (Lindenmann, 2008), based on an anthropological analysis of conservation in the region.

Throughout the interview process we maintained a self-critical awareness and attempted to minimise each individual team member's impact on responses. Prior to implementing the surveys widely and following the pilot in Tsitakabasia, all field assistants practiced posing the questionnaires to one another. I rotated among field assistants as they performed interviews to ensure that the assistants asked questions in a consistent manner. Each evening I examined questionnaires to ensure that there were no inconsistencies, and in the event of discrepancies, we returned to the interviewee the following day for clarification.

Because many of our results were based on self-reports, we used triangulation with quantitative data to examine the veracity of many of the responses we received (Jick, 1979). Self-reports have a long history in the criminal justice literature. They have generally been found to be useful for discerning behaviours within a population (Thornberry & Krohn, 2000), though criticism of self-reports due to the possibility of falsification and recall errors remains (Kirk, 2006). There are several methods that can be used to validate results of self-reports, including a comparison of self-reports with official criminal records, as well as the use of key informants (Hindelang et al., 1979). We attempted to validate the behavioural self-reports by asking 3-4 key informants in each community about the actions of all the individuals who we interviewed. When all key informants were consistent in their description of an individual's behaviour, it was compared to the individual's self-report.

3.7.3 Control vs treatment

In this study, responses of individuals from Andranomandeha, Mananjaky, Tsimafana, Beroboka-Nord, Tanambao-Fenerive acted as control communities. While this represents a larger relative influence of Sakalava communities than in the treatment sample, and does not include any coastal communities with similar dynamics to Ampataka, the use of random-effects models in the analyses of control and treatment communities accounts for this inter-village variability.

Communities in the Menabe have been subject to different development trajectories and interactions with government and NGOs over the past decade. For example, Japanese development projects have brought hand pumped wells to five of the ten intervention villages and three of the five non-intervention communities. The local NGO Fanamby, which focuses on agricultural development, has been involved in both the control and treatment communities. Nevertheless, these differences are not divided simply between control and treatment communities. Thus, while development interventions have been received at differing rates among communities, I am aware of no biases that would impact the conclusions from this dissertation.

3.7.4 Explanatory variables

In Chapters 4 and 5, I test the relationship between a variety of explanatory variables and forest-use behaviour, fairness of benefit distribution and perceptions of family and community benefit. I used standard variables such as age, family size, marital status, wealth and ethnicity. I also included variables reflecting engagement with the intervention and representing components of the opportunity cost of changing behaviours.

I used position in the forest association as a proxy for the elite, as those individuals who are members of the board have knowledge and likely greater access to the benefits of the intervention. I also examined relative wealth by asking individuals if they had more, less or the same amount of three types of agricultural fields, rice, corn/peanuts and bean fields as other individuals in the community. In addition, I asked whether the production from these fields was enough, more than enough or not enough to satisfy family requirements. I expected that the wealthy and elite would be more likely to give up forest use activities because they have less subsistence need for forest goods, and that they would receive greater benefits from the intervention than others due to their influence on the community. I did not include variables such as household items or capital because of the differing values placed on these possessions by the different ethnic groups included in the study, and because interviews did not always occur within individual's primary residence.

Proximity of land to the forest and size of family were used as variables to represent different measures of opportunity costs of not expanding land or using forest resources, and I expected that individuals with land next to the forest would be less likely than others to perceive a benefit from the intervention and would be less likely to change their behaviours. Variables representing knowledge of Durrell interventions acted as a proxy for how engaged individuals were with the intervention. I expected individuals with knowledge of the intervention to be more likely to change behaviours and perceive the intervention as beneficial and fair.

3.7.5 Statistical Approach

I performed all analyses in R2.7 (R Development Core Team, 2005). I used the base package and a number of additional packages (tree, lme4, ROCR; epicalc, pwr). I used generalised linear models (GLM) and generalised mixed effects models (GLMM) to examine my social data in two chapters.

This thesis seeks to make contributions to both the applied economic and ecological sciences. However, these two disciplines have differing approaches to regression. Economists tend to value conformity of models with a-priori theories of relationships between explanatory and response variables, while ecologists prefer to develop parsimonious models that are refined from initial maximal models (Armsworth et al., 2009). This thesis follows the ecological approach to model development and simplification because I was interested in examining the influence of a wide range of explanatory variables on behaviour change, and fairness and distribution of benefits.

3.7.6 Generalised linear models and mixed effects models

Standard linear regression assumes a normal distribution of errors, constant variance, a lack of measurement error in the explanatory variable and all unexplained variation within the response variable. Linear regression subsequently uses the best fit of a straight line that minimises the sum of squares from the straight line. Generalised linear models (GLM) relax these assumptions by allowing regressions to be fitted when response variables error follow non-normal distributions and variances are not constant. GLMs achieve this through the use of a linear predictor and link function (Crawley, 2007). The first step in developing a GLM is defining the error structure of the model. In the examples in this thesis, the data exhibited binomial errors. Next, a linear predictor is used to define a structure of the model and subsequently provide a sum of the terms for each of the model parameters. Models are typically fitted using maximum likelihood approach which selects the model parameters that are most likely to produce the data. Finally, the linear predictor is transformed into a response variable value through the use of a link function. In this thesis my response variables primarily exhibited binomial distributions and thus I relied on a logit link to transform the linear predictor value.

Generalised linear mixed effects regression is an extension of GLM models that incorporate the use of both fixed and random effects as explanatory variables. Random effects are a category of explanatory variables that help to explain the covariance in the data, for example when observations are not independent (Pinheiro and Bates, 2000; Bolker et al., 2009). Random effects provide grouping structure within models. For example, in the case of observations of individuals from two different communities, responses may not be independent from others inside a community due to shared histories. Alternatively in the case of time series data, multiple observations from a single individual are not independent. The use of community or individual as a random effect isolates this impact of grouping by accounting for the different error variances in each community. In Chapter 4, community as a random effect controls for differences among communities within treatment groups and thus allows for the examination of differences between intervention and non-intervention communities. Because fitting generalised linear mixed effects models requires integrating potential parameter values over all possible values for random effects, an inefficient process, I used Laplace approximation to measure likelihood because I had binomial data and a low number of random effects in each model (Bolker et al., 2009).

3.7.7 Model simplification

In initial maximal models, interactions between variables were not included in order to avoid over over-parameterisation of the models. Correlations between variables were checked. In the case of over parameterised maximal models due to low sample sizes, such as when evaluating rare behaviours, I used regression trees to select a subset of variables for the initial model (Crawley, 2007).

I used a backwards stepwise approach to iteratively eliminate non-significant explanatory variables and I compressed factor levels that did not significantly differ from one another during model simplification (Crawley, 2007). Critics of stepwise elimination argue that variables are sometimes eliminated prematurely, and that Akaike Information Criterion (AIC), which allows for ranking a selection of models, is a more appropriate approach to model simplification (Burnham and Anderson, 2002). However, in my case the backward stepwise approach presented advantages by offering an intuitive approach to grouping factor levels. Furthermore, the use of ranking or model weighting using AIC, when there are a number of categorical variables with multiple factor levels, requires developing an extremely large number of candidate models, which was not practical for this study.

Following each iteration of variable elimination using the backwards stepwise approach, I compared models using an ANOVA to evaluate whether the simplified model was significantly worse than the previous model (Crawley, 2005). The minimum model was selected when removing the next least significant explanatory variable led to a poorer model. I assessed the significance of fixed effects with a Wald statistic (Pinheiro and Bates, 2000). However, there is substantial debate over the value of test statistics from mixed-effects models due to unresolved questions regarding the appropriate number of degrees of freedom to consider (Bolker et al., 2009). Minimal models only are presented in the results.

For mixed effects models, the decision to include the random effect in the final model was made using a log-likelihood test between the minimum adequate mixed effect model and a generalised linear model without the random effect (Crawley, 2007; Bolker et al., 2009). When random effects were included, I used a variance components analysis to describe the importance of the random effect within the final model (Pinheiro and Bates, 2000; Borger et al., 2006). I assessed the significance of fixed effects by comparing the Wald statistic of each fixed effect to the chi-squared distribution.

The discrimination ability of the final GLM and GLMM models with response variables with binomial distribution was represented by the values of Area Under the ROC (Receiver Operator Characteristic) Curve (AUC). Models with AUC values greater than 0.7 were considered to demonstrate reasonable discrimination ability, though the utility of particular AUC values is context dependent (Pearce and Ferrier, 2000). These statistical approaches represent the current state of the art in ecological social statistical analysis and provide powerful tools for isolating the impact of interventions.

3.8 Conclusions

While PES offers advantages to previous biodiversity conservation intervention approaches, it is not a cure-all, and indeed it creates new challenges to implementing interventions in developing countries. My case study from the Menabe provides a timely example of PES and the following chapters use the case study and methods above to explore challenges related to the implementation and evaluation of PES for biodiversity conservation in developing countries. This will offer insight into how PES influences individual behaviours and will identify the conditions under which PES is most likely to be successful.

Chapter 4. The impact of a community-based payments for environmental services intervention on forest-use behaviours

4.1 Introduction

Payments for environmental service (PES) schemes transfer positive incentives to individuals or communities contingent on the supply of well-defined environmental services, with the aim of maintaining or increasing the supply of those services (Wunder, 2007, Chapter 2). Although PES for biodiversity conservation is well established in the context of American and European agro-environment schemes (de Koning et al., 2007), there is less understanding of the efficacy of PES in supplying biodiversity services in developing countries despite a growing body of case studies (Landell-Mills & Porras, 2002; Engel et al., 2008; Chapter 3).

PES is based on the assumption that payments to service providers will change behaviour in a way that will improve service provision compared to what would have occurred in the absence of an intervention (Chapter 2). Land managers can only be considered service providers if they have control over service provision. A variety of actions can be taken to maintain or improve service provision, including changing resource use behaviours, carrying out monitoring and enforcement to reduce external threats on their land, or by actively restoring biodiversity services on degraded land. Alternative approaches to conservation may need to be considered if land managers are unable to directly influence service provision. There are a variety of additional factors related to the individual characteristics of service providers and users and their relationship that influence whether PES is an appropriate conservation tool, including the security of funding, whether payments amounts are based on existing markets or one-off negotiations, and whether the providers and users are individuals, communities, or loosely defined groups (Chapter 2).

However in cases where PES is appropriate, payments are not the only incentive from the intervention that impact individuals' decisions. Natural resource management interventions, including those based on the PES paradigm, are multi-dimensional, incorporating both positive and negative incentives within a larger socio-political framework (Koontz, 2001). Few evaluations have examined how specific components of conservation interventions influence individuals' behaviours in relation to the wider context of the other incentives impacting their decisions (Borgerhoff-Mulder et al., 2007). This is particularly important in the case of community-based PES schemes in developing countries where a complex set of incentives, including payments, outreach and the legal framework, as well as local, regional and national monitoring, influence the decision to engage in activities.

This study seeks to assess how the Durrell PES intervention for biodiversity conservation in the Menabe has impacted local behaviour. In particular, I address the following questions:

- What is the relative importance of payments compared to other conservation intervention activities in affecting individual resource management behaviours and attitudes?

- What factors have influenced whether or not individuals stopped behaviours; and is there a difference between communities that have participated and those that have not participated in the PES intervention?
- How do individuals' reasons for stopping behaviours relate to their desire to return to the behaviours?

We rely on self-reports from individuals of whether they have changed their involvement in a range of natural resource use behaviours, their stated desire to return to these behaviours and their reported reasons for changing.

4.2 Methods

4.2.1 Behaviours of interest in the Menabe PES

In the Menabe system, payments are contingent on both the state of the strictly protected forest (the number and abundance of species of interest) and on actions that impact the system (forest governance indicators and monitored threats), scored during an annual assessment carried out by Durrell in collaboration with community members (Chapter 3). There are a number of forest-use activities that can impact a community's payment, either by altering habitat or posing threats to species. These include clearing forest for grazing, agriculture or paths, selective cutting of trees for timber or for accessing honey and hunting of lemurs. Each forest-use behaviour has varying legality under national laws, though if individuals are caught engaging in regulated activities inside the strictly protected forest they could face local and national fines or even prison sentences (Table 4-1).

Table 4-1 Description of the legality of the behaviours that were tracked in the study. Also includes the extent of national enforcement and likely consequence of being caught.

Behaviour	Legality	Realistic national enforcement	Realistic consequence	Subsistence importance	Profitability	% report engaging in behaviour[a] Durrell	Non-Durrell
Agricultural expansion	Illegal	Limited	Fine & prison	High	High	76	69
Lemur hunting	Illegal	None	Fine	Low	Low	13	5
Canoe building	Subsistence use legal	Limited	Fine	Low	High	17	8
Tenrec collection	Seasonally legal	None	None	Medium	Medium	62	53
Honey collection	Legal, though the normal collection method is illegal	None	None	Medium	Medium	58	45
Tuber collection	Legal	None	None	High	Low	77[b]	na
						n=388	n=182

[a] Percent of the male population who have ever engaged in each activity from Durrell and non-Durrell communities.
[b] Tubers have n=628 as women are also involved in collecting these

Where land is formally titled, identifying those holding land tenure is important for identifying service providers (Chapter 2). However in locations where few people have formal tenure and communities have established customary management rights to land, community-based PES interventions may be appropriate (Engel and Palmer, 2008). In some cases, PES interventions can help facilitate transfer of legal property rights to individuals or communities (Rosales, 2003), though there has also been some concern that the prospects of a PES could lead to property right conflicts by individuals or communities aiming to access benefits from the intervention (Wendland et al., 2010). Such conflict is unlikely in the Menabe, as the customary boundaries of each community forest were established well before the PES began and the goal of the PES is to incentivise communities to implement their existing management plans (Antona et al., 2004).

Positive indicators from the state of the system and forest governance are scored alongside the negative indicators of threat to create an aggregate score for each community. The scores are subsequently adjusted proportional to the size of each forest to reflect the greater amount of service provided by larger forests. Payment amounts and scores are announced in each community through an annual "forest party" sponsored by Durrell. Payments are distributed to the communities via the local forest associations and each community has a different procedure for sharing the benefits with the wider community (Chapter 3).

4.2.2 Data collection

We used interview data from individuals from eight communities receiving payments (n=645) and five communities that did not receive payments (n=204). The interviews addressed forest-use behaviours in the past and present, as well as future desire to use the forest. Questions focused on six behaviours (agricultural expansion, timber harvesting, collection of honey and tubers and hunting of tenrecs and lemurs, questionnaire in Appendix 2). Few individuals reported starting the behaviours rather than stopping them since 2000, so only factors influencing the decision to stop each behaviour were included in statistical analyses.

We asked both open-ended and closed questions to ascertain whether and why individuals continued to engaged in or changed specific behaviours. The activities of Durrell were not explicitly mentioned in this part of the interview to avoid introducing bias. We also asked about respondent's knowledge of rules, participation in community activities, attitudes towards conservation organisations, as well as their perception of conservation and government interventions in the region. Individuals reported wealth in terms of relative size of fields (small, average or large) and sufficiency of production for family needs (not enough, enough, and more than enough) for three types of agricultural land (wet rice fields, riverbank bean fields and slash and burn maize/groundnut fields).

We performed closed ranking exercises in communities participating in the PES on the perceived effect of conservation related activities in changing the respondents' forest-use behaviour (Durrell monitoring, government monitoring, community monitoring, environmental outreach and Durrell payments). Individuals then separately ranked how four activities that are specifically implemented by Durrell alone (monitoring, environmental outreach, payments, and parties) impacted their behavioural choices and attitudes.

We also performed semi-structured interviews with key informants, including the presidents of each community association, 2-3 members of each local forest patrol, and with 11 individuals who had been imprisoned for agricultural expansion.

Triangulation of self-reports

I attempted to validate behavioural self-reports by asking 3-4 key informants in each community about the actions of all the individuals who we interviewed. When all key informants were consistent in their description of an individual's behaviour, it was compared to the individual's self-report. Of the 575 records with agreement out of over 2000 responses between key informants and self-reports that behaviours had been carried out in the past, there was 78% agreement between self-reports and key informant views on whether there had been behavioural change. These results are consistent with similar self-report studies of delinquent behaviours that used direct observation for validation (Bernard et al., 1984). As expected, there was under reporting of engaging in behaviours and over reporting of leaving behaviours. Yet given the imperfect knowledge of key informants, there is no clear evidence that key informant responses represent the true accounts. I have no reason to believe that responses to motivational questions were consistently biased.

4.2.3 Analysis

Analysis of behavioural change

Closed rank data were analysed using the non-parametric Kruskal-Wallis rank test followed by multiple comparisons with a Bonferroni correction. Effects of intervention and socio-economic characteristics on reports of stopping behaviour were modeled using generalised linear mixed effects models (GLMM), with community as a random effect and a binomial link function (Table 4-2). Regression trees were used to select a subset of variables where models were overparameterised due to low sample sizes (lemur hunting and canoe building). A backward stepwise procedure was used for model simplification (Chapter 3).

Table 4-2 Explanatory variables to predict changes in behaviours in GLMM.

Socio-economic explanatory variables	Response
Community	13 communities
Intervention community	2 level factor (yes/no)
Sex	2 level factor (M/F)
Age	Continuous
Ethnicity	3 level factor (Tandroy, Sakalava, Korao)
Married	2 level factor (yes/no)
Number of children	Continuous
Immigrant	2 level factor (yes/no)
Years in community	Continuous
Agricultural wealth (size and sufficiency) for three field types	6 level factor (summed for wet, dry and rice fields)
Frequency of forest-use	5 level factor (never, seasonally, monthly, weekly, daily)
Durrell related explanatory variables	
Member of association[a]	2 level factor (yes/no)
Member of association board[a]	2 level factor (yes/no)
Knowledge of Durrell	2 level factor (yes/no)
Knowledge of national forest service	2 level factor (yes/no)
Knowledge of payments	2 level factor (yes/no)
Knowledge of competition	2 level factor (yes/no)
Size of Household	Continuous
Agriculture next to the forest	2 level factor (yes/no)
Attitudes	
Attitude towards Durrell (Likert scale)	5 level factor
Attitude towards national forest service (Likert scale)	5 level factor

[a]Represents variables only investigated within Durrell intervention communities

Chi-squared and Fisher tests were used to examine combined primary and secondary reasons for stopping behaviours and compare between Durrell and non-Durrell communities. Open responses on why individuals stopped carrying out behaviours were placed into broad categories of personal or social reasons, reasons related to fear, and biogeographic reasons related to the abundance and distribution of the species (Table 4-3). A GLMM modeling approach was also used to look specifically at whether the reasons why individuals stopped a behaviour influenced their inclination to return to this behaviour.

Table 4-3 Summary of reasons individuals reported to have stopped carrying out a behaviour.

Fear
Afraid of local community
Afraid of Durrell
Afraid of national forest service
Afraid of other (plantation owner, wasps, lost)
Biogeographic
Too far to go
There are many close by
Look somewhere else
There are not many
There is not enough place to look
Social/ personal
Not used to it
Too physically difficult
Have enough/don’t need it
No money to invest
Prefer to buy
Not enough time
Have become too old

4.3 Results

4.3.1 Role of payments on behaviours and attitudes

The intervention activities carried out in the communities had significantly different impacts on individuals' reported decisions to reduce their forest-use behaviours ($K=977.75$, $df=4$, $p<0.001$, Figure 4-1a). Durrell's monitoring and environmental outreach and local monitoring ranked higher than government monitoring and payments in their impact on individuals' behaviours. Of the interventions specific to Durrell, monitoring had the greatest impact on reported reduction in forest-use behaviours, followed by environmental outreach, payments and parties respectively ($K=427.1$, $df=3$, $p<0.001$, Figure 4-1b). The effect of these activities on local attitudes was precisely the opposite, with parties producing the most positive attitudes followed by payments ($K=403.0$, $df=3$, $p<0.001$).

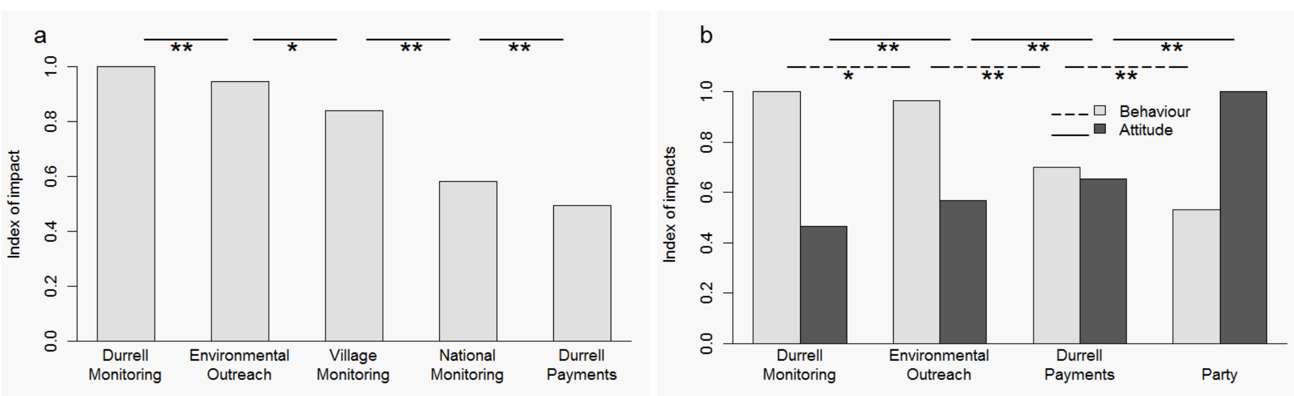


Figure 4-1 Influence of intervention activities on forest use behaviours and attitudes.
Indexes of influence of a) each intervention activity on individuals' forest-use behaviours in Durrell intervention communities (n=645) **and b) each Durrell intervention activity on individuals' forest**-use behaviours and attitudes in intervention communities (n=645). The indexes are based on the sum of ranks a) (1-5), b) (1-4) where 1 represents the largest impact. Horizontal lines with asterisks above represent a significant difference between the reported influence of two intervention activities on behaviours or attitudes. * represents a $p<0.05$, ** $p<0.01$ from non-parametric post-hoc multiple contrasts.

4.3.2 Correlates of behavioural change and reported reasons for change

Agricultural expansion and tuber collection were the most frequently cited forest-use behaviours, reflecting their importance for subsistence livelihoods (Figure 4-2). Canoe construction and lemur hunting were the least common behaviours, as expected due to the specialised skills required, as well as the ethnic aversion to lemur hunting for the Tandroy and the limited need for canoes for those living far from large water bodies.

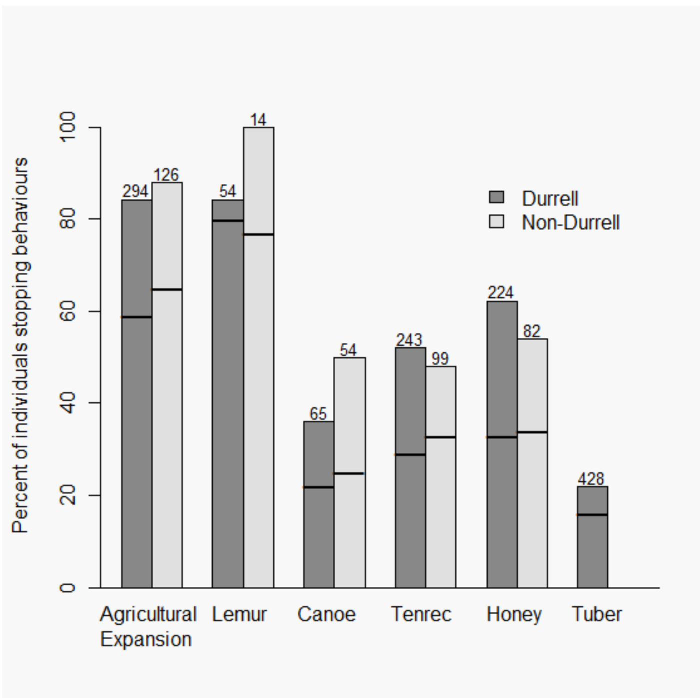


Figure 4-2 The percentage of individuals stopping behaviours since 2000.
The horizontal line within each bar represents the proportion of individuals who reported stopping a behaviour, but who would like to restart. Individuals in non-Durrell intervention communities were not surveyed regarding tuber collection. Sample sizes are presented on top of the bars.

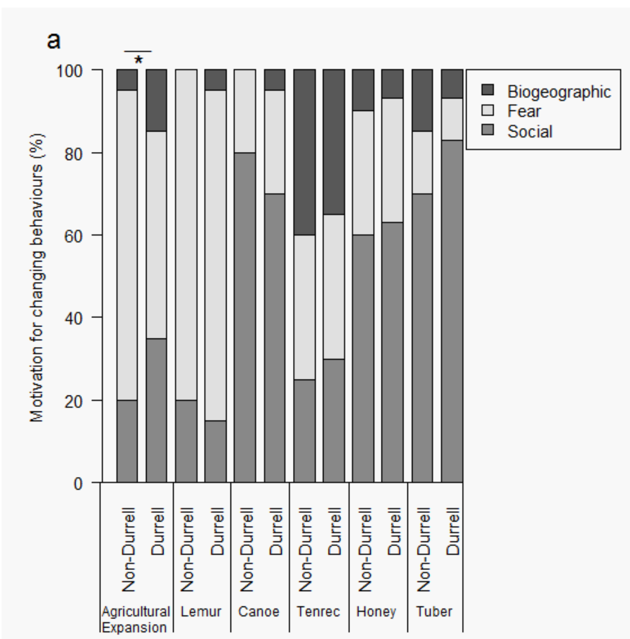
Individuals in both Durrell and non-Durrell communities reported reductions in their forest-use behaviours since 2000 when Durrell interventions began. The most reduced behaviours were agricultural expansion and lemur hunting, reflecting the severe consequences of being caught and the clear legal status of these behaviours. Predictably, age was positively associated with changed forest behaviours, as these activities are particularly physically challenging, while those who were heavily reliant on forest use during the rainy season were less likely to move away from agricultural expansion, canoe building, and tenrec and honey collection (Table 4-4). The other fixed effects of behavioural change were either inconsistent in their direction across behaviours or were only correlated with isolated behaviours. For all behaviours, the use of community as a random effect explained a large portion of the variance, highlighting the differences between villages in behavioural change.

Table 4-4 Final GLMM summaries for individuals who stopped behaviours.

Explanatory Variables [a]	Agricultural expansion	Lemur hunting	Canoe building	Tenrec collection	Honey collection
Age		+ [b,c,d]		+++	+++
Married				--	
Land next to forest			-		
Rainy season forest use	-		--	---	---
Wealth of maize/groundnut fields	+		+		-
Knowledge of Durrell activities		+			
Korao ethnic group					-
Sakalava ethnic group	++			+	
Random effect: Community [e]	59%	99%	65%	61%	47%
AUC [f]	0.813	0.973	0.855	0.852	0.845
N	432	62	82	362	325

[a] The full list of variables tested is in Table 4-2
[b] Shaded variables were included in the minimum adequate model for that behaviour.
[c] Positive and negative signs reflect the impact of the explanatory variables on the changed behaviour, with positive signs meaning individuals were more likely to stop the behaviour.
[d] Significance is expressed through the number of +/- signs (+,-: p<0.05; ++,--:p<0.01; +++,---:p<0.001)
[e] The percent of variance explained by the random effect is presented under the random effect.
[f] AUC is a discrimination index that measures the predictive power of the model based on the probability that the predictions and the outcomes are in agreement, with values >0.7 representing a reasonable fit.

Surprisingly, payments were not mentioned as a primary reason for stopping behaviours in open questions on the reason for specific behavioural changes. Instead, fear was the most prevalent reason for reported reductions in agricultural expansion and lemur hunting, reflecting the illegality of these behaviours (Figure 4-3a). The reduction in tuber collection was the least driven by fear, and most driven by social/personal reasons such as getting older. Interestingly, the reduction in tenrec collection was strongly motivated by biogeographic factors with individuals commonly commenting that tenrecs have become increasingly difficult to locate.



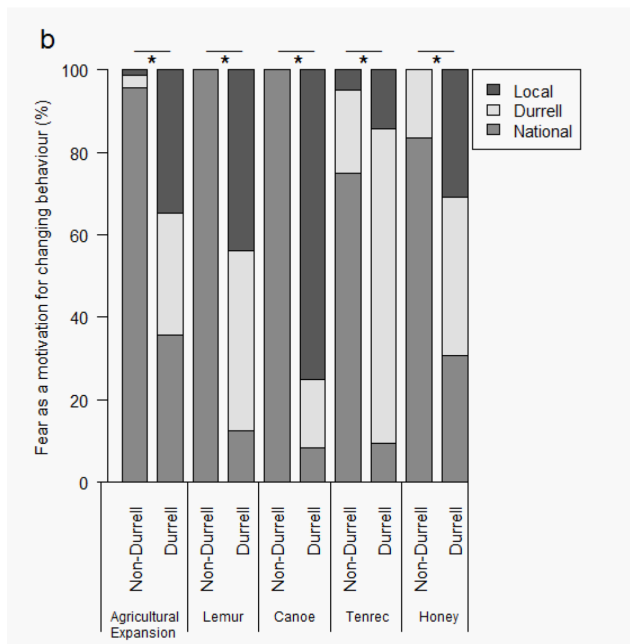


Figure 4-3 Reasons individuals have stopped forest-use behaviours.
For a) overall reasons b) fear only. Horizontal bars with * represents p<0.05 for chi-squared tests between Durrell and Non-Durrell communities within each behaviour.

These broad-scale factors influencing the decision to leave behaviours were similar between Durrell and non-Durrell communities. However, when the responses were divided into fear of local, national and international institutions, fear of the local forest association, and of Durrell instituting fines and punishments, rather than fear of not receiving payments, predominated in Durrell communities (Figure 4-3b).

Community forest patrol members said they are hesitant to turn in fellow community members, but suggested they would confront the individuals personally. However, there is no evidence that this has occurred. Community monitoring has yet to catch local transgressors in the act. Nevertheless over the past two years, communities have actively addressed external threats through seizing canoes in Kiboy and impounding felled logs in Ampataka that were cut illegally by a regional logging business.

4.3.3 Stability of behavioural change

Despite the reported changes in behaviour, the majority expressed a desire to return to each activity and this was strongly related to the reasons individuals changed their behaviours. Individuals who changed due to social reasons were less likely to want to return to the behaviours than those who changed due to fear (Table 4-5). This is unsurprising considering that social reasons reflect reasons related to ageing, increased food security or an inability to engage in behaviours. Upon further examination of fear, individuals whose fear originated from local institutions were less likely to want to return to the behaviour. This suggests a potential long-term benefit of promoting local governance and wider community involvement in forest management. Community as a random effect was significant in each of these analyses, suggesting that the strength of the response varied among communities.

Table 4-5 GLMM on stability of behavioural change.
Summaries of logistic mixed effects models to describe how the desire to return to a behaviour or not is related to the general reasons people stopped forest-use behaviours (n=908) and the specific reasons related to fear that individuals stopped forest-use behaviours (n=382).

	Estimate	Standard Error	z
General reasons for changed behaviour			
Biogeography	0.25	0.206	1.21
Fear	0.719*** [a]	0.192	3.75
Social	-0.676***	0.194	3.48
Random effect: Community [b]	20%		
Specific reasons relating to fear for changed behaviour			
Fear of local community	-0.640*	0.291	2.19
Fear of government and Durrell	1.288***	0.224	5.75
Fear of the forest	-1.513***	0.379	3.75
Random effect: Community	22%		

[a] * Represents significant difference with $p<0.05$ and *** $p<0.001$.
[b] The percentage of the variance explained by the random effect is presented.

4.4 Discussion

4.4.1 The roles of payment size and distribution

Payments in this case study appear to influence individuals' behaviour only weakly. This limited impact may be partially attributed to the small size of payments and their community-based distribution. The current payment size when divided amongst the entire community is generally not significant at the individual level, with each community receiving between $200 and $2000 and the benefits being distributed among 40 to 500 individuals. In most cases, this would not offset the high opportunity costs experienced by many households and thus motivate behavioural change (Chapter 5). Nevertheless, the award amount appears sufficient to motivate the community associations and therefore ensures the existence of a local monitoring regime. Thus, the payments may be acting as a positive incentive for the association members to impose traditional command and control incentives or social pressure on the rest of the community, though some benefits of the payments undoubtedly trickle down to individuals and appear to result in positive attitudes towards monitoring. It is possible that if higher payments were instituted, and distributed effectively, these dynamics could change to the point at which individuals' actions were primarily motivated by the payments rather than by the actions of the community associations and Durrell's monitoring.

The role of the distribution of payments is reflected in the impact of the annual forest party sponsored by Durrell on behaviours and attitudes. This party, at which payments levels are announced, had the largest positive impact on local attitudes, despite the fact that the party costs were lower than the payments. While, payments are prone to being controlled by subgroups in the community, parties are non-rival, non-excludable goods that create local value and thus strongly influence attitudes (Chapter 5).

4.4.2 The roles of legality and regulation

The co-existence of illegal and regulated behaviours in PES schemes complicates the understanding of how payment amount and distribution influence behaviours. Illegality likely heightens the motivational power of fear and monitoring, thus reducing the necessary payment size. It also muddies relationships between government, NGOs and individuals with respect to laws, as payments may be seen as implicitly recognising a right to engage in an activity.

In the case of the Menabe, the PES was proposed by Durrell in part because of weak government capacity in the region to enforce resource management laws. If an organisation without legal enforcement authority spearheads a PES, the ability to withhold payments can create a third party authority to monitor and enforce rules. Nevertheless, these PES schemes still require some government support to adequately address external threats. For example, in two cases in the Menabe where community members encountered large-scale regional threats in their forest (commercial logging and canoe exploitation), the associations complained they lacked the jurisdiction and capacity to enforce the violation locally. Despite reporting the incidents to authorities, the cases remained unresolved more than a year later. This underscores the importance of ensuring that the service providers can enforce their property rights through local institutions or through relationships with regional or national bodies (Engel & Palmer, 2008). Thus, while one of the practical purposes of non-governmental PES systems in developing countries may be to attempt to circumvent poor governance at a regional or national level, there is still a need for engagement among administrative levels (Bowles & Gintis, 2002).

In contrast to NGO-led PES, it may be challenging for a central government to justify payments to service providers as an incentives not to break the law. Yet a potentially useful role for national PES schemes, rather than aiming to directly motivate behavioural change, may be to sway local opinions or to compensate a subsection of the population. This was noted in Costa Rica, where a change in national land-use laws restricting agricultural expansion was made palatable to rural people through the introduction of a PES system (Pagiola, 2008; Chapter 2). Thus, despite complications, both government and NGO-led PES systems can be viable in the case of regulated or illegal behaviours. However, it is important to understand the mechanisms under which PES interventions motivate individuals. While PES interventions may function to influence individuals' actions or to engender good will towards an existing regulatory regime, it is clear that in order to influence behaviour there is a need for monitoring.

4.4.3 The relationship between monitoring and payments

Our results highlight that fear is a strong motivator for changing behaviour and that this fear is brought on by monitoring. Studies have shown the probability of being caught motivates behaviour more than the size of the punishment (Stigler, 1970; Polinsky & Shavel, 1979). Thus, while the legal consequences of being caught by the government are much more severe than those posed by Durrell or the local communities, government monitoring is almost non-existent and so the chance of being caught is very low. Hence, the high reported impact of Durrell monitoring on forest-use behaviours is likely due to their relatively frequent monitoring. Although Durrell has no legal authority to inflict punishment, there is widespread belief in the communities that they have a private jail in the regional capital and this misinformation undoubtedly increases the capacity of Durrell to influence local behaviours.

While local monitoring has social advantages in terms of presence on the ground, its impact is limited to some extent by a lack of incentives for local enforcement. Community members bear a cost of monitoring their peers, which is reflected in monitors' hesitance to turn in fellow community members (Hechter, 1984).

Nevertheless, monitoring by the local association members still appears to impact local behaviours due to their constant presence in the community, irrespective of whether they are actively monitoring. Overall, these findings on the importance of monitoring in motivating behavioural change may explain the mixed results of some PES schemes with inadequate monitoring (Missrie & Nelson, 2005).

We also found that outreach and education play a role in motivating behavioural change. The importance of environmental outreach in influencing behaviours may be due not only to increased knowledge of the rules, but also to development of local support and pride and subsequently non-use value for the forest, as has been observed in other interventions (Borgerhoff-Mulder & Coppolillo, 2005). Indeed, outreach is a critical prerequisite for successful PES. For example, the ineffectiveness of a PES project in Mexico has been attributed to the lack of appropriate outreach and a focus on the technical aspects of the intervention (Corbera et al., 2007).

While payments themselves do not act as the primary drivers of individuals leaving behaviours in the Menabe PES system, they play an important role influencing local attitudes, particularly in terms of perceptions of monitoring and the illegality of behaviours. It is the presence of the payments that justifies the monitoring regime to local communities. Furthermore, as communities in the Menabe acquire legal control over their forests, they do not have to allow outsiders access to their forest. By placing a monitoring programme within the framework of positive incentives, Durrell’s work has been widely embraced in the region. Thus, payments create the positive attitudes and trust that lay the groundwork for voluntary local acceptance of monitoring, which ultimately acts as the motivational tool.

4.4.4 Correlates of behavioural change

The lack of difference between Durrell and non-Durrell communities was surprising; but it is possible that the regional presence of Durrell has had a corollary impact on the remaining communities in the Menabe. Though Durrell is not explicitly active in the five interviewed communities, there is undoubtedly some penetration of Durrell activities through the radio and sightings of the Durrell vehicle.

There is some evidence that Durrell may have successfully influenced lemur hunting in the region. While it is interesting that knowledge of interventions did not impact other behaviours, increasing awareness of the illegality of lemur hunting has been a specific focus of Durrell interventions over recent years (F. Rakotombololona, personal communication). Though the Durrell communities overall did not appear to have higher rates of reported reductions in behaviours than non-Durrell communities, anecdotal evidence from semi-structured interviews suggests an impact on lemur hunting. During the first year of the PES intervention, the Durrell monitoring team encountered lemur traps in the forests of Kiboy, Tsianaloky and Anketrevo. After revealing these results to the communities during the annual award ceremony, there have been no subsequent reports of lemur trapping (Durrell Wildlife Conservation Trust, 2004-2008). Indeed, over the 120 kilometers of transects we carried out in four community forests in 2007/2008, only one lemur trap was encountered (Chapter 6).

Individuals' reported reasons for engaging in or stopping behaviours were informative in understanding how they respond to incentives. For the two behaviours with the greatest consequences of being caught, hunting lemurs and agricultural expansion, fear was the dominant motivator of change. Fear played a lesser role for other behaviours suggesting the importance of the legal framework in influencing behaviour. The observation that individuals in Durrell communities tended to fear enforcement from the local associations and Durrell, rather than the national authority in Durrell communities, illustrates the importance of a frequent monitoring presence.

The decline of tenrec hunting behaviour was strongly related to a perceived reduction in tenrec populations. Previous studies have suggested that due to their high fecundity, overexploitation of tenrecs is not likely (Nicoll, 2003). These results suggest that harvest may in fact have a severe impact on tenrec populations. This may be partly due to observed illegal collection of females and offspring during the closed season (personal observation).

Social reasons for leaving behaviours, including aging, having others in the family able to do the work or simply not needing the resource anymore, were strong motivators of change. However, these drivers are not easily influenced by external interventions and therefore may be less important in the design of conservation programmes than consideration of the structure of incentives.

4.4.5 Stability of change

Despite a large percentage of individuals reporting moving away from behaviours, the reported changes do not necessarily reflect a lasting success. Though fear appears to be an effective tool at motivating individuals to stop behaviours (Witte & Allen, 2000), those motivated by fear were most likely to want to return to past behaviours. Fear of external bodies (national government and Durrell) promoted a less stable change in behavioural change than fear of local bodies, suggesting a potential benefit of engaging local communities through devolving monitoring responsibility and increasing local governance capacity.

4.5 Conclusion

Natural resource management projects are increasingly incorporating PES models into their interventions, making it particularly important that we try to understand how payments influence behaviour and attitudes of individuals. This chapter has highlighted the essential role that monitoring plays in PES compliance. Changes in behaviour were motivated more by fear of being caught and punished due to increased monitoring than by a desire to increase the payment the community received. However, this does not mean that payments are not important. Payments played an important role in improving attitudes towards the implementing NGO and provided justification for monitoring. While I found fear to be a strong motivator for changing behaviours, it also appeared to result in relatively unstable behavioural changes. Local empowerment of community associations through this community-based PES scheme demonstrated the potential to contribute to a more stable behavioural change. As PES theory and practice develop, conservation practitioners need to consider how payments interact with outreach, monitoring and other components of interventions to motivate service provision.

Chapter 5. The role of fairness and benefit distribution in community-based payments for environmental services interventions

5.1 Introduction

The previous chapter examined how a variety of incentives influence behaviours, however it is also important to understand the impact of intervention approaches on local attitudes. This chapter explores factors that influence individuals' perceptions of net-benefit and fairness of distribution of benefits, in order to better understand impacts on long-term social success of the PES intervention.

The international community has widely acknowledged the inequitable distribution of the costs and benefits of biodiversity conservation. For example, the costs of protected areas are born locally, frequently by poor rural communities, while the benefits accrue globally (Balmford and Whitten, 2003; Bawa et al., 2004). As a result, donors increasingly require international conservation programmes to consider poverty alleviation (Brockington and Schmidt-Soltau, 2004; World Bank, 2005), although the extent to which conservation and social objectives can be achieved concurrently is still debated (Newmark and Hough, 2000; Adams et al., 2004; Barrett et al., 2005). There are many examples of local people suffering as a consequence of conservation interventions, for example in cases of forced resettlement from protected areas (Colchester, 1997; Schmidt-Soltau, 2003) or inadequate compensation for limitations on natural resource access (Shyamsundar and Kramer, 1996; Peters, 1998). Yet even when local people's needs are integrated into project planning and implementation, there are challenges to ensuring the fair distribution of benefits. Benefit capture by the elite (Kellert et al., 2000; Thompson and Homewood, 2002; Balint and Mashinya, 2006; Fritzen, 2007), variable opportunity and transaction costs among individuals and communities resulting in the perception of unfair distribution (Kumar, 2002; Adhikari, 2005; Meshack et al., 2006) and the exacerbation of long-standing interpersonal conflicts (Koch, 1997; Agarwal and Gibson, 1999) all highlight the challenges of realising community-wide social benefits from conservation projects.

As conservation interventions move away from the regulatory-based fences and fines paradigm and towards approaches that focus on positive incentives, such as payments for environmental services (PES) (Ferraro, 2001; Wunder et al., 2008a), consideration of factors influencing individual choice becomes increasingly important (Adams and Hulme, 2001; Chapter 4). Most examples of PES in the literature represent transactions with individual providers or groups of coordinated landowners. However, an increasing number of PES interventions, particularly in the tropics, apply to community-managed land both legally and informally. For example there have been efforts towards community conservation concessions in Guyana and Indonesia (Niesten and Rice, 2004; Wunder et al., 2008b), and payments to communities for managing forest for biodiversity in Mexico (Missrie and Nelson, 2005). Furthermore, many nationally run programmes within an international agreement on payments from Reduced Emissions from Deforestation and Forest Degradation (REDD) will have to consider incentive distribution on community-managed land (Kaimowitz, 2008). These community-based

PES schemes offer a particular challenge, as incentives aim to influence individual behaviour, but they pass through community institutions.

Economic considerations with respect to costs and benefits certainly influence individuals' decisions to engage in behaviours (Persky, 1995). However, additional factors including procedural and distributive fairness impact individuals' motivation (Fehr and Falk, 2002, Vatn, 2009). Perceptions of unfairness can undermine the effectiveness even of incentives that provide apparent net benefits (Thibaut and Walker, 1975; Folger, 1977; Kanfer et al., 1987). As well as providing tangible benefits, conservation success is therefore contingent on developing positive local attitudes (Struhsaker et al., 2005). Consequently, the perception of fairness and net benefit at the individual scale can have a substantial impact on the participation of the wider community and thus the efficacy of an intervention.

In this study, I examine the role of the distribution of incentives in influencing how individuals within communities perceive benefits and fairness from a community-based PES intervention to better understand the particular opportunities and challenges facing community-based PES. I hypothesise that individuals in powerful positions in the community forest association will receive the greatest benefits and that individuals and communities with high opportunity costs will perceive the lowest levels of net benefit. I also hypothesise that governance of benefit distribution will influence perceptions of fairness. I use self-reports to examine how individual socio-economic characteristics, as well as community-level differences, relate to individuals' perceptions within 8 communities engaging in the Menabe PES. I discuss solutions for addressing issues of inequitable distribution, variable opportunity costs and governance failures in community-based PES.

5.2 Methods

5.2.1 Local context

In the Menabe PES, the overall size of the payment distributed does not change significantly from year to year, however the distribution of payments among communities does. For example, the community of Kiboy's payments have ranged in value from $370 to $2,230, whereas the value of the payments to Kirindy village, with a small community managed forest area and poor biodiversity, has not exceeded $250 (Chapter 3, Figure 3-4).

Though the monetary value of the payment to each community is announced annually, it is not distributed in cash, but instead used to purchase in-kind incentives. The board members of each community forest association, with the permission of at least 80% of the association, decide what they would like to purchase with the Durrell payment. Communities have purchased electric generators, building materials, cooking supplies, bicycles and cows. The subsequent distribution of these items differs between communities. In some communities, direct access to the items purchased is open to the entire community; while in others, they are only available to association members or for the facilitation of community-wide events. In the context of this chapter, it is important to consider the community's monetary award and the items purchased by the community separately. As a result, for the rest of this chapter the share of the annual award money distributed to each forest association is henceforth called "the payment," and the in-kind items and services purchased for community use is called "the incentive."

In addition to the payment, communities receive additional tangential benefits from the PES management system. Durrell holds an annual forest party in each community to discuss the score and announce the payment amount. This party includes music, dance and a community meal. Durrell performs environmental outreach programmes, forest association capacity building workshops and biodiversity monitoring programmes throughout the year. There are also occasional opportunities for individuals to receive paid work with Durrell as a forest guide or monitor.

The costs of engaging in the intervention vary by individual and by community. Individual costs vary to the extent that individuals had previously been involved in controlled activities. Each community association has an annual membership fee, ranging from \$0.30 to \$7, as well as a one-time joining fee between \$0.50 and \$2 (F. Rakotombololona, personal communication). Membership gives individuals the right to apply for permits to use resources in local multi-use managed forest and in some communities, membership is a prerequisite for taking advantage of incentives. Though Durrell did not directly introduce new rules or laws into the community forest management system, their independent species monitoring programmes throughout the region have created a perception of enforcement of existing laws that was not previously present, thus placing an increased perceived cost on non-compliance (Chapter 4).

5.2.2 Interviews

I carried out interviews from December 2007 to April 2008 in 8 of the 10 communities participating in the Durrell scheme. These included 645 structured interviews with individuals and 55 semi-structured interviews with small focus groups.

My team asked individuals to evaluate the impact of the incentive by weighing the benefits from the incentive against the costs from limitations on forest resource use at the family and community-levels (Interview questionnaire in Appendix 2). Individuals responded with their perception of whether their family and the community had benefited, stayed the same or lost out. We did not disassociate costs and benefits from each other in the questions, but rather asked for individuals' impressions of net benefit or loss at the family and community-level. We then questioned individuals regarding their perception of the fairness of the distribution of the incentive. We asked individuals about their knowledge of Durrell's work in the region. We also investigated whether individuals understood that the payment amount received by the community was based upon meeting forest management criteria.

To reflect the opportunity costs of engaging in the PES system, respondents were asked to self-report their forest-use behaviours, now and in the past (Chapter 4). Additional background socio-economic characteristics and whether individuals were members of the forest management association were also recorded. Relative wealth estimates were based on self-reports of the relative size of three field types owned by each household and the self-reported adequacy of harvest. Each of the field types, flood plain fields, rice fields and dry slash/burn fields, are important for food security and income (Chapter 4).

Semi-structured interviews (n=55) were performed with individual forest association board members from each community, and focus groups of 2-3 members of forest patrols within each forest association. The structure of the forest association, monitoring regimes, local politics and the size and distribution of the payment and the incentive were discussed.

5.2.3 Analysis

In order to understand how socio-economic characteristics (Table 5-1) relate to the perceived fairness and perceived net benefit from the PES system for each respondent's family and community, I used generalised linear models (GLM) with a logistic link function (Chapter 3).

For the purpose of logisitic analysis, perception of net benefit was simplified and grouped into a binary response variable (those who benefited vs. those who lost or were unchanged). Community was used as a fixed effect to test for an influence of community-level dynamics on perceptions of fairness and net benefit. Interactions between variables were not included in order to avoid over-parameterisation of the models. A backwards stepwise approach was used iteratively to eliminate non-significant explanatory variables (Chapter 3).

Table 5-1 Variables used in GLMs for family & community net-benefit and fairness. All of the variables were included in each maximal model. Percentages give the percentage of respondents in each category of the variable. For continuous variables means are given unless otherwise stated.

Variables	Summary
Community	8 level factor
Ethnicity	Sakalava- 43%, Tandroy- 29%, Korao- 20%, Other- 8%
Gender	male- 67%, female- 33%
Age	38 years
Household number	5 individuals
Years in community	22 years
Married	yes- 80%, no- 20%
Expanded land, hunted lemurs, collected honey/ tenrecs, built canoes prior to Durrell interventions	land- 78%, lemurs- 14%, honey- 60%, tenrecs- 53%, canoes- 22%
Wants to expand land, hunt lemurs, collect honey/tenrec, build canoes in future	land- 72% , lemurs- 15%, honey-54% , tenrec-65% , canoes- 21%
Desire to use a diversity of forest products (sum of behaviours)	2.33 products
Agricultural wealth (dry, wet & rice fields: perceived adequacy on scale of 1-9)	dry fields- 3.31, wet fields- 2.43, rice fields- 2.25
Has land next to forest	yes- 30%, no- 70%
Dry season forest use	never- 70%, seasonally- 3%, monthly- 7%, weekly- 14%, daily- 6%
Rainy season forest use	never- 48%, seasonally- 4%, monthly- 14%, weekly- 23%, daily- 11%
Forest association member	member- 30%, board- 10%, non-member- 60%
Knowledge of the work of Durrell	yes- 80%, no- 20%
Knowledge of the PES relationship between actions and incentives	yes- 79%, no- 21%
How has PES incentive impacted community?	benefit- 77%, loss- 7%, unchanged- 6%, don't know- 9%
How has PES incentive impacted family?	benefit- 47%, loss- 9%, unchanged- 40%, don't know- 3%
Is the distribution of incentives fair?	fair- 60%, unfair- 11%, don't know- 29%

5.3 Results

5.3.1 Do incentives benefit the community and individual families?

Perceptions of net benefits at the community and family levels

The majority of individuals reported that the community as a whole benefited positively from the intervention (Figure 5-1). Individuals perceived the community to have benefitted more often than they perceived their family to have benefitted ($x^2 = 210.9$, df = 2, $p<0.001$). Board members of the forest management association were less likely to perceive a community-level net benefit than the rest of the population (Table 5-2, column a). The communities of Ampataka and Kirindy had the lowest proportion of individuals reporting a community-level net benefit, though each still had a majority of individuals reporting a net community benefit.

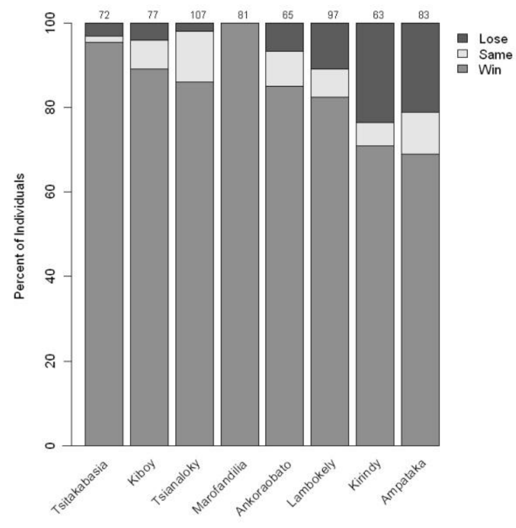


Figure 5-1 Percentage of respondents perceiving community benefit.
Respondents who perceived the community to have benefitted, stayed the same or lost out overall with regards to the intervention benefits and the forest access constraints. The total sample size for each community is presented above the figure. Communities are presented in order of the length of time that that Durrell has been active in them.

Relatively few individuals reported their family to have explicitly lost out from the conservation intervention (Figure 5-2). Individuals' perceptions of whether their family had benefitted or stayed the same/lost were related to their position in the forest management association, with board members more likely to report having received a net benefit than regular association members, and regular members more likely to report receiving a net benefit than non-members (Table 5-2, column b). This supports the hypothesis that association members and those in power receive the highest level of net-benefits.

Table 5-2 GLM summary on community and family net-benefit and fairness of distribution. Generalised linear models were used with a log-link function, and with a binary response variable, where 1 = fair or beneficial and 0 = unfair or no/negative benefit. Positive and negative signs represent the direction of the association between explanatory and response variables, where binary explanatory variables are coded such that 1 is a positive response. * significant at p< 0.05, ** significant at p< 0.01; and *** significant at p<0.001. Standard errors are given in parentheses. The full set of explanatory variables that were considered is presented in Table 5-1.

	a. Community Net-Benefit	b. Family Net-Benefit	c. Fair Distribution
Membership: Board member/ non members	-1.45*** (-0.39)		
Membership: Board member		1.05** (-0.38)	-1.20* (-0.56)
Membership: Not member		-0.65* (-0.26)	-1.57*** (-0.49)
Knowledge of the PES system			2.12*** (-0.62)
Knowledge of Durrell system		1.16** (-0.38)	
Want to expand dry fields		-1.08*** (-0.32)	-0.89 (-0.51)
Want to use forest		0.38*** (-0.09)	
Land next to forest		-0.77** (-0.25)	-1.81*** (-0.45)
Dry field wealth		0.17** (-0.06)	0.21* (-0.09)
Wet field wealth			-0.28** (-0.09)
Dry season forest use: never/seasonally			1.53* (-0.76)
Rainy season forest use: never/seasonally		-1.41* (-0.61)	-1.05* (-0.42)
Community: Ankoraobato/ Kiboy			-2.56*** (-0.43)
Community: Ampataka/ Ankoraobato/ Kirindy		-1.07*** (-0.32)	
Community: Ampataka/ Kirindy	-2.00*** (-0.36)		
AUC	0.78	0.79	0.89
Sample size	586	426	457

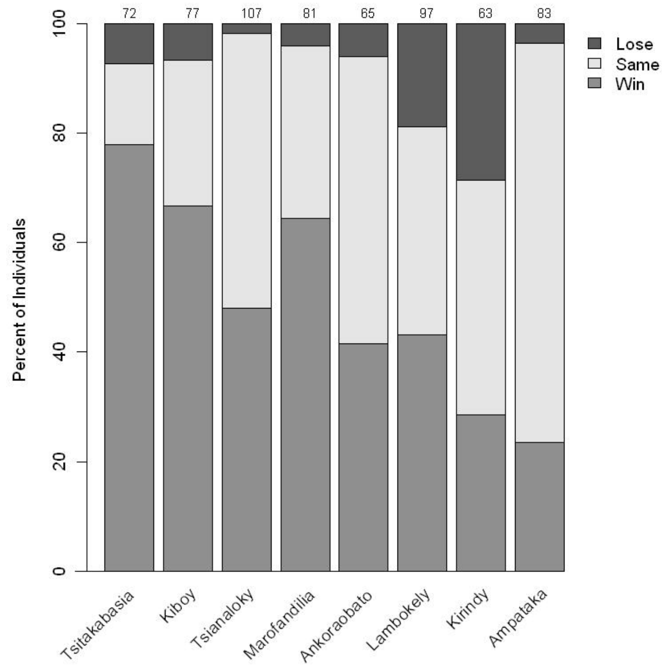


Figure 5-2 Percentage of respondents perceiving family benefit.
Respondents in each community who perceived their family to have benefitted, stayed the same or lost out overall with regards to the benefits from the intervention and the forest access constraints. The total sample size for each community is presented above the figure. Communities are presented in order of the length of time that Durrell has been active in them.

Individuals with land next to the forest, as well as individuals, who would like to expand their agricultural land, were less likely to perceive their family as benefitting overall from the intervention, supporting the hypothesis that those with high opportunity costs would experience lower levels of net-benefit. However, this was not supported universally in the case study. Though individuals who use the forest heavily for timber, honey and meat would be expected to have high opportunity costs from a PES intervention, the fact that they were actually more likely than others to report a net benefit suggests recognition of benefits from forest management.

This active interest in the results of forest management was also noted in the semi-structured interviews with some members of the association board. Those board members, who relied on forest products for their livelihoods, expressed a desire to control the access of outsiders to the strictly managed forest and were frequently the community forest guardians. Their interest in forest management contrasted with that of board members from the community economic elite who were more interested in managing incentives. Encouragingly, these forest guardians thus appear to be local advocates for forest conservation.

Reasons for differences in perceptions of net benefit among communities

A number of potential explanations for the relatively low levels of benefits experienced by Ampataka and Kirindy emerged in the interviews, including high joining fees, low payment sizes and high opportunity costs. In Ampataka, 39% (n=82) of respondents and 42% (n=60) in Kirindy suggested that individuals did not join the association because of insufficient time or money. Ampataka is the only community in the system in which households had marine fishing as a complementary primary livelihood alongside farming. In interviews, Ampataka's board members expressed ambivalence towards the conservation intervention, mirroring the high proportion of respondents who perceived there to be no change for their families due to the intervention (59%, n=80). The high association entrance fee ($7 per year) was a frequent complaint for members of the association in Kirindy. Finally, the structure of the PES system was criticised frequently in both Ampataka and Kirindy. Board members in the two communities frequently complained that their consistently low annual payments were due to the poor biogeography of their forests (west of the central forest block) and small area of locally managed forest.

Kirindy and Lambokely arguably experience the highest opportunity costs of not expanding their agricultural land into the forest, as they are limited to farming on dry fields and do not have forest to expand into. Thus, their relatively high level of individuals reporting a net loss supports the hypothesis of the impact of opportunity cost on perceptions of benefit at the community level. Each of these communities is composed of a clearing surrounded by primary forest. While only a portion of this forest is managed locally, and thus within the payment scheme, the forest management associations assert *de facto* management and enforcement of rules on the forest immediately surrounding the communities. Unlike the other six communities, Kirindy and Lambokely do not have a stream or river to grow crops next to outside of the wet season, and so their alternatives are limited.

5.3.2 Is the distribution of incentives considered fair?

Perceptions of fairness of distribution within communities

Most respondents appear to perceive the distribution of the incentive at the community-level to be fair, with over 85% of individuals reporting a fair distribution. Nevertheless, board members of the forest associations were more likely than regular members to perceive the distribution of the incentive as unfair (Table 5-2, column c). Non-members were also more likely than regular members to perceive the distribution as unfair. Those with agricultural land next to the forest and individuals who wanted to expand their agricultural land were each more likely than the rest of the population to perceive the distribution to be unfair. The respondent's community was also a significant determinant of perceived fairness, with individuals from Ankoraobato and Kiboy more likely to perceive the system to be unfair (Figure 5-3). New communities to the scheme, like Ampataka and Kirindy, had a relatively high percentage of respondents reporting that they did not know if the incentive distribution was fair or unfair.

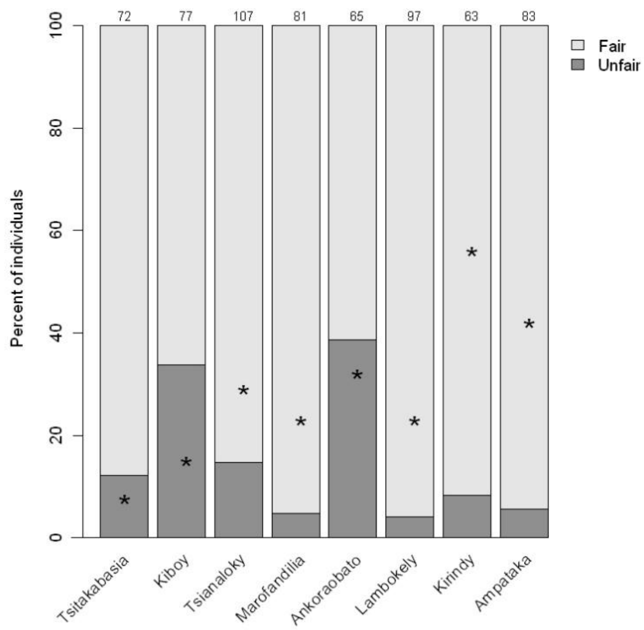


Figure 5-3 Percentage of respondents perceiving a fair distribution of benefits.
"*" indicates the percentage of "don't know" offered as a response, out of the total sample size for each community, which is presented above the figure. Communities are presented in order of the length of time that that Durrell has been active in them.

Reasons for differences in perceived fairness among communities

Supporting the hypothesis of the importance of governance on perceptions of fairness, the issue of political leadership emerged often in discussions in Ankoraobato and Kiboy.

Indeed, 28% (n=60) of respondents in Kiboy and 23% (n=53) in Ankoraobato suggested that a dislike of their association's politics or leadership was responsible for individuals not joining the forest association, far higher than the other communities. In these two communities, board members mentioned their dissatisfaction with the association presidents because the presidents reaped excessive personal benefit from the scheme, with accusations in both communities that the presidents had failed to distribute seeds from a conservation and development intervention, that they mismanaged funds from forest entry fees and multi-use forest permits, and that they used community goods such as bicycles and cooking materials for non-official business. In Ankoraobato, there were reports that the president arranged for a logging company to harvest in the forest. In Kiboy, the association was beginning to address this problem by establishing a committee of individuals outside the sphere of the president to control access to the payment and to incentive distribution. In Ankoraobato, board members suggested that they would like to have a vote for a new forest president. In both these communities, the associations have lost members over recent years. These two communities also had the highest absolute payment values in the year prior to the survey. Despite criticism regarding the fairness of distribution of incentives in Kiboy, the majority of individuals in Kiboy perceived that their family had benefitted overall from the intervention.

5.4 Discussion

5.4.1 Opportunities and challenges for community-based PES

Where communities have legal or *de facto* control over the quality or quantity of an ecosystem service of value to others, but lack an incentive to manage it; a community-based PES is a logical approach to ensure service provision. It has the potential to contribute to development objectives and build management capacity at the community level. These benefits were observed throughout the communities, particularly in the widespread belief that the PES intervention has an overall positive impact at the community level.

Nevertheless, significant challenges remain for community-based PES. Principle among these is that a community-based PES does not necessarily address individual opportunity costs, and may thus have difficulty in incentivising individual behaviours. Similarly, given the challenges of managing the distribution of incentives, there is danger of cooption of benefits by sub-groups within the community that leads to widespread disillusionment. Alternatively, those who benefit within a community may use command and control tactics to ensure compliance from the wider community, thus subverting the principles of PES, as a positive incentive, at the level of the individual (Chapter 2).

5.4.2 The impact of incentives within communities

While a net benefit was widely perceived at the community level, individuals, particularly those experiencing high opportunity costs and non-members of the forest association, were less likely to perceive a benefit at the family level, while board members of the association were more likely to perceive a family-level benefit. This suggests an inequitable benefit distribution and the potential presence of elite capture, as is common in many conservation and development projects (Kellert et al., 2000; Thompson and Homewood, 2002). This highlights the design question in community-based PES of whether those paying for the service provision wish to target a specific group within a community, such as the poor, or those experiencing opportunity costs of the community engagement, or whether they are content to let community associations govern incentive distribution.

Techniques such as offering in-kind, non-rival and non-excludable incentives may avert benefit capture by a small group and ensure access to the poor, whereas targeting incentives more precisely within communities may offer potential solutions to address variable opportunity costs. In many rural communities, it may not be possible to explicitly direct incentives to the relative poor without upsetting local social structures (Agrawal, 2001; Thompson and Homewood, 2002). This distributional issue is addressed in the Menabe through the use of in-kind incentives that are decided upon and shared by the community, such as bicycles, generators and public buildings. In-kind incentives accrue coarsely at the community-level while costs are experienced at the individual and family level. Thus, though this approach offers equal access to all members of the community in principle it does not easily address variable opportunity costs. Furthermore, in practice, the distribution of and access to these easily appropriated items have been criticised in some Menabe communities.

A focus on non-rival and non-excludable benefits, such as parties and community infrastructure would potentially ensure that the entire community has the opportunity to access the benefits. Indeed, community members expressed a preference for the annual community party offered by Durrell over the in-kind incentives (Chapter 4). However, many forms of community infrastructure that are commonly distributed as non-rival or non-excludable benefits from conservation interventions, such as educational and medical facilities, may be seen as human rights, which should not be contingent on local resource management (Chapter 2).

In cases when meeting individual opportunity costs is deemed important, targeting may be an approach to improve the efficiency of the intervention. However particularly in developing countries, targeting may be limited in practice due to the difficulty of extracting private information from individuals seeking to receive benefits (Akerlof, 1970). Techniques such as auctions and screening contracts can be used to induce individuals to share their true preferences and to complement publically available information on opportunity costs (Ferraro, 2008). Alternatively, spatial targeting has been explored for the distribution of incentives and may be appropriate for those with land directly adjacent to forest (Watzold and Drechsler, 2005; Wünscher et al., 2008). However, such forms of targeting typically involve relatively high transaction costs, particularly when dealing with large numbers of smallholders, and thus reduce gains in efficiency. These approaches also generally signify a shift away from community-based PES to a more individualised PES.

5.4.3 Differences among communities

Qualitative evidence from observations and semi-structured interviews suggests that differences in the perceptions of benefit between communities appeared to be influenced by economic issues, whereas failures in governance dominated differences in perceptions of fairness.

Given the high agricultural opportunity cost of communities surrounded by protected forest, it is not surprising that Lambokely and Kirindy reported the highest proportion of individuals expressing a net loss from the PES scheme, mirroring the primary costs of conservation in other interventions (Archabald and Naughton-Treves, 2002; Adams and Infield, 2003). In Kirindy I also demonstrated that high monetary cost of entering the association could also be a substantial barrier, as individuals are uncertain whether they will receive adequate benefits from participation (Gong et al., 2009). This raises concern regarding the absolute size of payments for each community. In the Menabe scheme, payments are based on the state of the system and management actions, relative to their forest size and the scores of neighbouring communities, rather than being based on opportunity costs. In Kirindy this lack of adequate compensation has led to discussion of dropping out of the management scheme entirely. There is thus a need for more research on the merits of basing payments on a competitive approach among individuals or communities. Nevertheless, it is clear that while the structure of the Menabe scheme may motivate communities to compete with their neighbours, strong monetary incentives are still necessary to encourage continued participation.

In contrast to the economic drivers of perception of net benefit, perceptions of unfairness appeared to be related to poor governance. Kiboy and Ankoraobato received the highest payment in 2006-2007 and some of the issues around fairness may result from the conflicts over the spending of the large payment and distribution of the incentive. However, the balance of evidence suggests that chronically poor governance on the part of community presidents has led to a collapse of trust. Indeed breakdowns in the perception of fairness and subsequent collapse of interventions are frequently caused by local-level leadership failures (Barrett et al., 2001; Thompson and Homewood, 2002; Smith and Walpole, 2005).

This demonstrates the importance of perceptions of fairness (Fehr and Falk, 2002) and governance (Antona et al., 2004) as key issues affecting the impact of payments and incentives in community-based management transfers, regardless of the tangible benefits received by individuals. Organisations that help to develop capacity in local institutions can address failed leadership by promoting clear democratic processes for electing board members at regular intervals and by trying to ensure that members have a voice in the association's actions (Thompson and Homewood, 2002). Such approaches may be considered paternalistic, but may give communities the perception that they are not constrained to a single local leadership paradigm. While this type of engagement offers promise, it also increases transaction costs, requires significant presence on the ground and is contingent on effective ground-level personalities.

5.4.4 Implications for the future

The creation of sustained positive social benefits has been acknowledged as a prerequisite for long-term intervention success within the natural resource management literature (Berkes, 2004). The observation that a higher proportion of individuals perceived a net benefit at the community-level (85%, n=545) than at the family level (58%, n=545) may present some cause for concern. Evidence from Tanzania suggests that communities initially engage in conservation programmes based on perceptions of future benefits, but will eventually drop out if realised family benefits are inadequate over time (Songorwa, 1999). Within the Menabe system it is not clear if this discord between the perception of community and family level benefits is a precursor to future weakness in the system, but the low proportion of individuals explicitly expressing loss due to the system (9%, n=545) is encouraging. Furthermore, there is some hope that a temporal component plays a role in community-wide perceptions of positive net benefit. Ampataka and Kirindy are the newest communities to participate in the competition and were the most likely to report community-level loses. Similarly, they had the highest percent of individuals reporting that they did not know if their incentives were fairly distributed. As their experience with the system grows, the annual payments and incentives may help to develop trust and demonstrate cumulative benefits so that positive perceptions of the PES system may grow (Salafsky et al., 2001). However, this is far from conclusive, as these two communities have also gained the lowest average annual payment of all the participating communities (Chapter 3; Figure 3-4).

5.4.5 Monitoring social impacts

In community-based conservation schemes, where local institutions control the distribution of incentives, the distribution structure and ultimate fairness may not be clear a priori (Adhikari, 2005). Monitoring social indicators is thus a critical, if rarely performed, component of conservation and development projects (Newmark and Hough, 2000; Pomeroy et al., 2004). Failure to consider distributional and fairness issues can undermine the impact of a PES or the long-term success of conservation interventions. Given the ethical and practical obligations to understand how interventions impact local people, studies of the distribution of positive and negative incentives should increasingly accompany and inform conservation interventions.

5.5 Conclusion

From a social perspective, the Menabe community-based PES appears to be an overall success due to high levels of perceived fairness of payment distribution and a low proportion of individuals expressing a sense of family-level and community-level loss. However, numerous pitfalls were observed in the scheme. There was a lack of net benefit accruing to those bearing high opportunity costs from not expanding their agricultural land. Poor governance of benefit distribution by local leaders also threatened to undermine the effectiveness of incentives in a few communities. As conservation interventions increasingly rely on positive incentives at the community level to motivate individual behaviours, these challenges, and others, need to be considered in the project planning stages and through monitoring social indicators throughout the intervention's lifetime. There are additional monitoring needs related to identifying whether the contingency criteria has been met within PES systems and for the ultimate distribution of payments. The next chapter will address the challenges relating to these monitoring requirements with respect to biodiversity conservation, in particular the selection of suitable indicators and approaches to monitoring differences.

Chapter 6. Paying for biodiversity: Can service provision be monitored?

6.1 Introduction

A defining feature of PES is that payments are conditional on provision of a service, which implies that those paying can monitor each service directly or via proxy indicators. The concept of statistical power, or the ability of a monitoring scheme to correctly detect differences among samples (Gerrodette, 1987; Legg and Nagy, 2006), is a potentially useful approach for assessing whether a scheme effectively monitors service provision. Indeed, if service provision is monitored without a consideration of a monitoring scheme's ability to detect differences, then payments are unlikely to create an incentive to influence behaviour, as service providers will not perceive a relationship between their actions, or the state of the system and the payment they receive. Those designing PES interventions must consider how to structure schemes in terms of: 1) which indicators to monitor; 2) how they will be monitored; and 3) how the monitoring information will be used to define payments (Figure 6-1). Each of these decisions has implications for the total amount of effort required for monitoring in PES interventions. Indeed, the challenges and costs of monitoring biodiversity in the context of PES schemes have been downplayed in the literature (Ferraro, 2001), but may be partially responsible for the relative rarity of biodiversity-based PES programmes in comparison to carbon or watershed PES programmes, where common indicators of service provision are more straightforward to monitor.

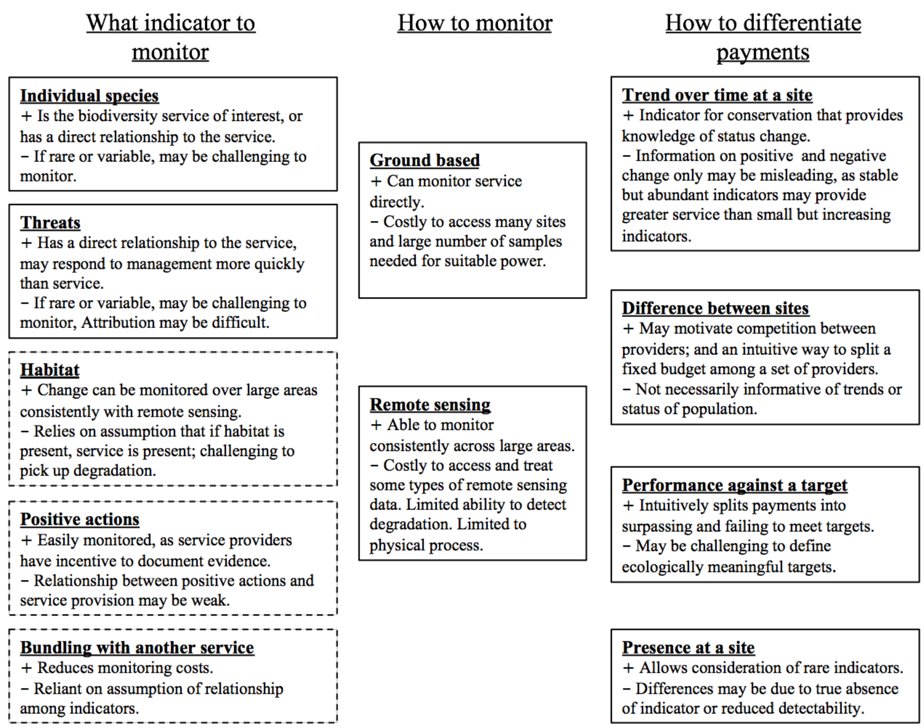


Figure 6-1 Framework for deciding on indicators for biodiversity conservation PES. Potential indicators for biodiversity conservation PES schemes, how indicators may be measured, and how payments may be determined. **"+" represents** advantages **of option and "-" represents** disadvantages of option. Boxes with solid outlines represent approaches that are addressed in this paper.

The concept of biodiversity has a variety of potential meanings (Purvis and Hector, 2000) and there is no universally accepted definition in the context of environmental services. As a result, choosing indicators for a biodiversity PES may be challenging. Some indicators of biodiversity are inherently more or less difficult to monitor than others and the choice of an indicator will therefore have a large impact on the cost and thus the feasibility of monitoring for payment distribution. The strength of the relationship of each indicator to the biodiversity goal (Lindenmayer, 1999; Le Tellier et al., 2009), as well as the precision with which each indicator can be monitored for a given level of effort (Taylor and Gerrodette, 1993), are critical to the decision of the most appropriate indicators.

Where the desired goal is the maintenance of populations of rare species in a landscape, five broad indicator types can be monitored. Firstly, individual species that are the target of conservation concern, or that are proxies for the ecological community of concern could be monitored directly (Durrell Wildlife Conservation Trust, 2004). For example, landowners may be paid for the presence of breeding birds or carnivores on their property (Musters et al., 2001; Zabel and Holm-Muller, 2008). Secondly, some authors have suggested that monitoring threats, and particularly changes in threats, may be more cost effective than monitoring biodiversity directly, because evidence of threats is often more easily detected than the biodiversity indicators themselves (Salafsky and Margoluis, 1999). Thirdly, the extent of habitat could be used as a surrogate for the presence of particular species. In most of these cases for PES, the relationship between biodiversity and habitat is assumed (Pagiola et al., 2004), rather than empirically demonstrated. While this assumption potentially reduces monitoring costs, it has the serious danger of giving a false indication of biodiversity conservation effectiveness (Chan et al., 2006; Nelson et al., 2008). Next, monitoring specific actions with positive impacts on biodiversity (rather than the presence of negative actions) has been proposed as a cost-effective way of monitoring biodiversity service provision under certain circumstances, as service providers have an incentive to prove their actions. For example, payments to individuals for guarding the known nesting sites of sea turtle or bird nests have been widely applied (Clements et al., 2009; Ferraro and Gjertsen, 2009). Finally, given the challenges in monitoring in PES for biodiversity conservation, a number of payments schemes have effectively ignored the need to monitor biodiversity indicators, and instead have "bundled" assumed biodiversity benefits with other services such as carbon or watershed protection (Asquith et al., 2008; Wunder and Wertz-Kanounnikoff, 2009). In bundled approaches, biodiversity service provision is assumed to accompany another service and a biodiversity premium may or may not accompany the primary service payment.

The choice of indicator is intimately linked to the decision on monitoring methodology (Figure 6-1). Monitoring for PES can be based on estimates from ground-based sampling or a remotely sensed estimate of the state of a system as a whole. Remote sensing, while ideally suited for measuring physical processes, like fires (Andrianandrasana et al., 2005) or habitat change, has a limited, but improving, capacity to detect the process of changing community structure within habitats (Turner et al., 2003; Goetz et al., 2007; Gillespie et al., 2008). In all cases, the ability to detect a true difference between sampling periods or sites is a function of sampling design and intensity (Taylor and Gerodette, 1993; Legg and Nagy, 2006). The power to detect a difference of a given magnitude is positively related to sample size and negatively related to the variability of the system. Increasing the number of sites monitored or the number of visits per site increases power (Pollock, 2006), but also increases the cost of monitoring (Field et al., 2005; Joseph et al., 2006; Pollock, 2006).

Finally, the amount of monitoring required is dependent on how monitoring results are used to define payments. At the simplest level, payments may be based on the presence or absence of an indicator following a particular monitoring effort. Alternatively, payments could be based on change through time of an indicator (Kremen et al., 1994, Maxwell and Jennings, 2005), on differences among sites, or on a comparison of each site against a particular target (Huggett, 2005; Baldwin and Bender, 2008).

In this paper I address the question of how much effort is required to monitor biodiversity as a basis for making conditional payments in a PES. Using the Durrell community-based biodiversity PES case study, I examine the effort needed to monitor individual species and threat indicators with a variety of approaches to monitoring, including trends within sites, differences between sites, performance against targets. I also examine multi-species presence and the presence of remotely sensed threats.

6.2 Methods

6.2.1 Context

In the Menabe, annual payments to the communities are based on a combination of data on the presence of species and threats in their area, and forest association governance indicators. The presence data are based on an annual 5-10 km transect walk inside the strictly protected forest, which is carried out by Durrell staff and local guides. The location of the transect differs each year. The staff records sightings of target species and threats on the transect. Governance indicators include increasing the participation of women in the community forest association and record keeping. Although these governance indicators have an indirect and uncertain relationship with the biodiversity indicators of interest, they offer some advantages to communities with poor baseline levels of biodiversity, but with high levels of motivation to participate. Data from the transects and governance indicators are then turned into scores, with high weightings for sightings of rare endemics. The scores are then added together for each community and the annual payment for the region is divided among the community forest associations based on their relative scores (Chapter 3). This chapter focuses on the results from four of ten communities that participate in the scheme, Kiboy and Tsitakabasia in the north and Ampataka and Marofandilia in the south. Payments to these four communities in 2007 totalled $3,505, and these past awards have been used to purchase electric generators, cooking equipment, bicycles, and building materials for community forest association offices (Chapter 3).

6.2.2 Data collection

Field transects

Forty non-repeated transects were carried out within each of the strictly protected forests of the four communities between November and February 2007-2008 (160 transects in total). I generated random points on the existing trails that mark the boundaries of each community's strictly protected forest. A biologist and local field assistant entered the forest at these points and followed at a bearing towards the forest interior for 1500m. This is similar to the approach Durrell biologists take each year, though the Durrell transects vary in length, depending on forest area. Morning transects began between 6:00 – 8:00 in the morning depending on the distance of the starting point from the base camp and each transect took between 1.5 and 4.5 hours to complete, depending on density of vegetation (mean = 2.8 hours). We recorded all sightings of indicators used by Durrell. These included 12 types of threats, 22 animal species and four plant species (Table 6-1). The position along each transect where the indicator was sighted was recorded with a GPS. Animals travelling in groups were documented as single sightings (with number in group noted). Cut trees were recorded as well as the diameter of each cut individual. Multiple records for felled trees, thought to have been part of the same cutting event, were noted as a single threat if they were visible from one another. Local assistants estimated the age of each threat. I validated these estimates of age by comparing independent estimates when two separate assistants encountered the same threat. 40% of the threats estimated to be under two years by one assistant were estimated to be between two and five years old by the other assistant, though neither assistant showed a consistent bias. This demonstrates some subjectivity in aging threats. Only recent threats were included in the analysis, as these are of interest for monitoring for annual payments, but I included in this category any observation that either assistant had identified as being of less than 2 years old, in order to maximise the sample size and ensure that no new threat was missed.

6.2.3 Indicators

The indicators that were selected by Durrell, and monitored in this study (Table 6-1) represent a broad range of mammals, bird, reptile, amphibian and plant species, as well as

Table 6-1 List of indicators used in the Menabe community forest transect monitoring. The total number of individual sightings and the percentage of transects (n=40 for each community forest) with sightings of the indicators in each community forest. IUCN Red List status is abbreviated (DD = Data Deficient, LC = Least Concern, V= Vulnerable, NT = Near Threatened, T = Threatened, E = Endangered, CE = Critically Endangered).

	Animals	Latin name	Red List	Total sightings	Percentage of transects with sightings			
					Kiboy	Tsitakabaksia	Ampataka	Marofandilia
Mammal	Common tenrec	*Tenrec eucaudatus*	LC	5	0	3	0	0
	Mouse lemur	*Microcebus sp.*	LC/E	7	3	0	3	15
	Striped mongoose	*Mungotictis decemlineata*	V	16	13	23	0	0
	Red-fronted lemur	*Eulemur fulvus rufus*	NT	34	20	18	8	13
	Red-tailed sportive lemur burrow	*Lepilemur ruficaudatus*	DD	53	40	23	20	25
	Sifaka	*Propithecus verreauxi*	V	54	38	23	8	15
	Jumping rat burrows	*Hypogeomys antimena*	E	178	68	78	13	23
	Pale fork-marked lemur	*Phaner pallescens*	LC	0				
	Coquerel's mouse lemur	*Mirza cocquereli*	NT	0				
	Fat-tailed dwarf lemur	*Cheirogaleus medius*	LC	0				
	Fosa	*Cryptoprocta ferox*	V	0				
Birds	Madagascar crested ibis	*Lophotibis cristata*	NT	5	5	5	0	3
	White-breasted mesite	*Mesitornis variegata*	V	62	23	28	5	38
	Giant coua	*Coua gigas*	LC	78	13	10	28	60
	Coquerel's coua	*Coua coquereli*	LC	342	50	65	78	95
	Crested coua	*Coua cristata*	LC	436	83	85	83	85
Reptiles	Mantellid frogs	*Aglyptodactylus sp.*	LC/E	2	5	0	0	0
	Chameleon	*Brookesia sp.*	??	5	5	5	0	0
	Madagascar tree boa	*Sanzinia madagascariensis*	V	7	0	0	5	5
	Chameleon	*Furcifer sp.*	V	9	8	0	5	8
	Flat-tailed tortoise	*Pyxis planicauda*	CE	16	3	8	3	0
	Madagascar snake	*Heteroliodon sp.*	?	17	15	8	8	8
Plants	Homonymous tree (Masonjoany)	*Tarenna madagascariensis*	?	50	40	25	0	5
	Hazomalany	*Hazomalania voyroni*	?	82	28	10	33	0
	Ebony	*Diospyros sp.*	?	251	35	25	38	3
Threats	Lemur trap			1	3	0	0	0
	Hunting of birds (evidence of feathers)			0				
	New Camps			2	3	3	0	0
	Large fire (>1 hec)			5	3	3	0	3
	Tree cut for hunting lemurs			11	8	13	3	0
	Small fire (<1hec)			4	0	0	18	0
	Cut tree for canoes			22	18	20	0	0
	Cut tree for honey			21	13	15	5	10
	New cart paths			8	13	50	33	3
	New walking paths			67	0	8	3	0
	Small cut trees			41	8	25	15	3
	Large cut trees			101	8	28	48	5

the most common threats in the region. Durrell's scientific research and conservation has principally focused on the flat-tailed tortoise, giant jumping mouse and local lemur species. The flat-tailed tortoise, giant jumping mouse, and a small mouse lemur, *Microcebus berthae*, are endemic to the forests of the central Menabe. The jumping mouse and mouse lemur are listed as endangered on the IUCN Red List, while the flat-tailed tortoise is critically endangered (IUCN, 2009). The birds represent a selection of easily detected species due to their calls, like the crested coua, to relatively cryptic species like the white breasted mesite. The plants included in the survey each have valuable uses for timber in the case of ebony and hazomalany, and for makeup and medicine for the masonjoany. Each of these species has a different importance in terms of local use value, endangerment, and international non-use value. Durrell tried to reflect international importance and endangerment within their monitoring scheme by awarding a greater weight to observations of some species such as the giant jumping rats, lemurs and the flat tailed tortoise.

The threats that were monitored have varying impact on the landscape and on specific species. Some threat indicators such as evidence of lemur traps or bird feathers represent direct impacts on the species of interest, some represent vectors for new threats, such as new roads or trails, while others are more related to the quality and structure of habitat, like evidence of fires or harvested trees. In Durrell's awards, these indicators are also weighted based on their severity, with destructive activities, such as large fires, having the greatest impact.

6.2.4 Analysis

I divided each 1500m transect into five 300m sections resulting in 200 sections per community forest. I re-sampled with replacement the sections for a given community forest to reassemble simulated transects of 1500m that included a beginning, three middle, and an end sections, to account for any potential biological edge effects or observer effects (e.g. tiredness at the end, training the eye at the beginning). To simulate a range of survey datasets representing different investments of monitoring effort, I then compiled the reconstructed transects into datasets of 1 to 500 transects at intervals of three transects (1,4,7,etc.). I repeated this process to create 500 iterations at each level of monitoring effort. I used the proportion of transects with sightings of each indicator as my metric for the state of the indicator within a given community's protected forest. I present my results on individual indicators through the power to detect changes as a function of monitoring effort.

Detectability

There are a number of factors influencing the detectability of species and threats, which therefore influence how well encounter rates reflect the true status of the indicator. Habitat variability can result in detectability differences (MacKenzie et al., 2006), but since the transects were all in ecologically similar dry deciduous forests, I assumed that there were no differences in detectability due to habitat differences between community forests though there was variability within the forest. Similarly, I controlled for observer effects as much as possible by carrying out transects consistently at the same time of day, with the same team and in a short time period. As populations decrease, group sizes may decrease, making the groups less detectable (McConville, et al., 2009). Although this may apply to the two lemur species that travel in groups or to clumped threats, such as forest clearings, it will not affect the other indicators and is likely to be a relatively small effect. Some species may be more wary and likely to flee from disturbance in areas where they are hunted than in areas where they are not. Thus, if the different community forests were exposed to very different levels of hunting there may be a bias introduced for some of the larger birds and lemurs.

Detecting differences in indicators

To avoid over-extrapolation from the data, I only considered indicators with more than thirty observations in the dataset as a whole and of the 38 indicators, only 22 fulfilled this criterion (Appendix 3). From the re-sampled datasets for each community forest I calculated the percentage of transects with sightings of each indicator for each transect number and iteration. I then simulated a change in the underlying encounter rate over time by decreasing the percentage of transects with sightings, using small (10%), medium (25%) and large (50%) effect sizes. For example, in a resampled dataset made up of 50 transects per community forest, 17 transects in Kiboy's community forest had sifaka (*Propithecus verreauxi)* sightings. Then, I decreased the 34% of transects with sighting by 10%, 25% and 50% to represent samples with 31%, 26% and 17% of transects with sightings at a 50 transect monitoring effort, (reflecting 15, 13 and 8 transects with sightings). I then used these proportions to assess the power to detect this change using a proportion test for each community forest at each iteration for each monitoring effort with

$\alpha = 0.05$ (Cohen, 1977). I averaged these power values from each iteration over the 500 iterations for each monitoring effort in each forest. I used these average power values from each monitoring effort to determine the minimum sample size needed to, on average, reach an 80% power to detect a difference between the proportion of transects with sightings for a given indicator. The tests were two-sided; as in a real scenario, the direction of differences between the proportion of transects with sightings would not be anticipated in advance. Raising the α value, for example from 0.05 to 0.1 or 0.2, would increase the power to detect change at a given sample size (Taylor and Gerodette, 1993; Di Stefano, 2003a; Di Stefano, 2003b). However I chose to perform all analyses with $\alpha =$ 0.05 to follow standard convention.

We also assessed the sample size required to detect a difference in the proportion of transects with sightings between each community's forest for each indicator with an 80% power. To complement this understanding, I modelled the minimum percentage difference that can be detected between two sites or time periods based on the proportion of transects with sightings at one site or in the initial time period for four levels of sampling effort (15, 30, 60 and 120 transects per site), using the formula for a proportion power analysis (Cohen, 1977).

Finally, I compared the proportion of transects with sightings for each indicator in each community's forest against specific targets that were designed to represent high biodiversity and low biodiversity situations. These targets were set at, respectively, 125% and 75% of the mean percentage of transects with sightings from all forests for each species (with an upper limit of 100% sightings). In practice, such targets would be based on values that have particular ecological importance for each species or threat, such as minimum viable populations or sustainable harvest levels (Shaffer, 1981; Robinson and Redford, 1991).

We also examined a framework for examining difference between forests that aggregates records of individual species or threats observed. I used the bootstrapped transects to track the cumulative number of species and threat indicators observed per community forest as monitoring effort increased from 0 to 40 transects. I compared the presence of all 38 indicators in this accumulation analysis between forests, rather than a single forest against a target or through time.

Finally, I reported observations of the presence of threat based on remote sensing through the number of fires detected within each community forest in 2007. Fires were identified by the University of Maryland's Fire Information for Resource Managers (FIRMS) project (Davies et al., 2009). The presence of fires within 100ha grid cells is estimated using a MODIS sensor on NASA's Terra and Aqua satellites daily. Fires greater than $50m^2$ are routinely detected, but cloud cover, non-homogenous surfaces and glint from the sun may obscure these estimates.

6.3 Results

For detecting differences in individual indicators, I present results for five indicators, representing a range of common to relatively rare indicators across the spectrum of animals, plants and threats. These are two lemurs, sifaka and red-fronted lemur (*Eulemur fulvus rufus*); one bird, crested coua (*Coua cristata*); one relatively rare tree, hazomalany (*Hazomalania voyroni*); and one threat, timber harvesting of large trees. I present the results on the remaining indicators graphically through power graphs in Appendix 4. I include all 38 indicators in the results on indicator accumulation curves.

6.3.1 Detecting differences in individual indicators

Trends through time

The effort required to detect a reduction in an indicator varies dramatically based on its initial rarity as well as the degree of reduction involved. To detect a 10% reduction in the proportion of sightings observed required well over 200 transects per community forest even for the most common species, the crested coua (Table 6-2a). By contrast, a 50% reduction was detectable for the coua with only 20 transects per site. For rarer indicators, change of 50% or 25% could only be discerned in a subset of community forests, and no changes of 10% were distinguishable. A reduction of 10% could not be discerned with a reasonable effort unless the initial presence approached 100% of transects, while detection of a 25% reduction required only slightly less effort (Figure 6-2).

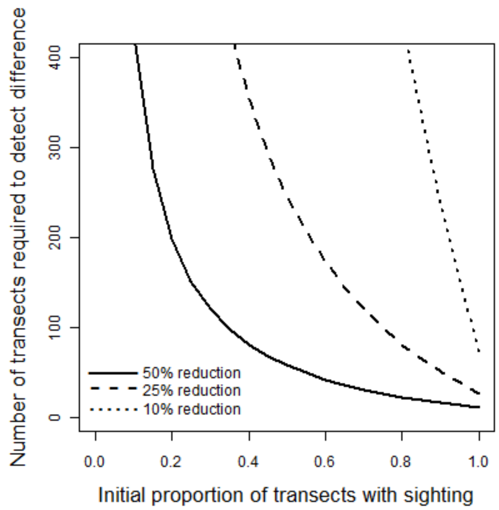


Figure 6-2 Number of transects required to detect reduction in sightings of indicators. Represents 10%, 25% and 50% reductions with 80% power from an initial proportion of transects with sightings of an indicator, at $\alpha = 0.05$.

Table 6-2 Summary of monitoring effort required for a variety of indicators. The number of transects required for an 80% power to detect change based on differences within a community forest, between community forests, and against a baseline for five indicators using presence/absence transects for two community forests (a high biodiversity community: Kiboy, and a low biodiversity community: Ampataka). The percentage of transects with sightings for each species can be found in Appendix 3. + represents an effort greater than 400 transects per community forest per year.

a.		Community	Sifaka (lemur)	Brown lemur	Crested coua (bird)	Hazomalany (tree)	Timber (threat)
Change over time	50% change	Kiboy	98	164	16	132	+
		Ampataka	+	+	20	80	38
	25% change	Kiboy	+	+	56	+	+
		Ampataka	+	+	56	340	152
	10% change	Kiboy	+	+	266	+	+
		Ampataka	+	+	272	+	+
b.							
Difference between community forests	Kiboy & Tsitakabasia (neighbouring)		+	+	80	+	44
	Kiboy & Ampataka (distant)		40	98	320	+	10
	Ampataka & Marofandilia (neighbouring)		220	+	10	+	10
c.							
Difference against baseline	125% of average	Kiboy	+	+	290	+	46
		Ampataka	62	136	160	140	52
	75% of average	Kiboy	104	202	60	136	142
		Ampataka	220	+	60	52	22

Differences between community forests

For plant and animal species, differences between community forests were most easily detected between distant communities (Kiboy and Ampataka), likely reflecting the ecological differences between habitats, which led to relatively large effect sizes (Table 6-2b). In contrast, neighbouring communities' forests tended to require much greater effort to discern a difference in an indicator; for example the communities of Kiboy and Tsitakabasia. In general, indicators that had a wide range of probabilities of sightings between communities, such as timber, required fewer transects to detect a difference. Thus effect size was again the critical factor determining whether a difference could be observed with 80% power. Indeed, effect sizes of less than 25% could never be detected with less than 120 transects per community forest (Figure 6-3).

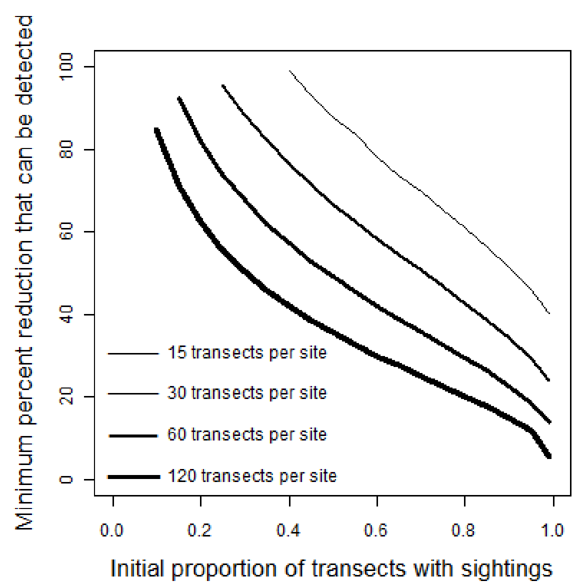


Figure 6-3 The minimum effect size that can be detected based on initial sightings.
The minimum percent difference between sites (a proxy for effect size) that can be detected with 80% power for a variety of initial proportions of transects with sightings at three sample sizes (15, 30, 60 and 120 transects per site). α= 0.05.

Performance against a target

Unsurprisingly, highly biodiverse forests like Kiboy were generally able to demonstrate that they had exceeded the 75% target for most indicators with a comparatively low effort (Table 6-2c). It was generally possible to discern whether a community forest had exceeded or significantly underperformed against at least one of the targets with adequate power, in fewer transects than it took to be assured of adequate power to measure differences between community forests or trends within forests. Nevertheless, this approach was subject to the same challenges regarding small effect sizes, in that it was difficult to measure performance with adequate power for communities and indicators, which were close to the targets.

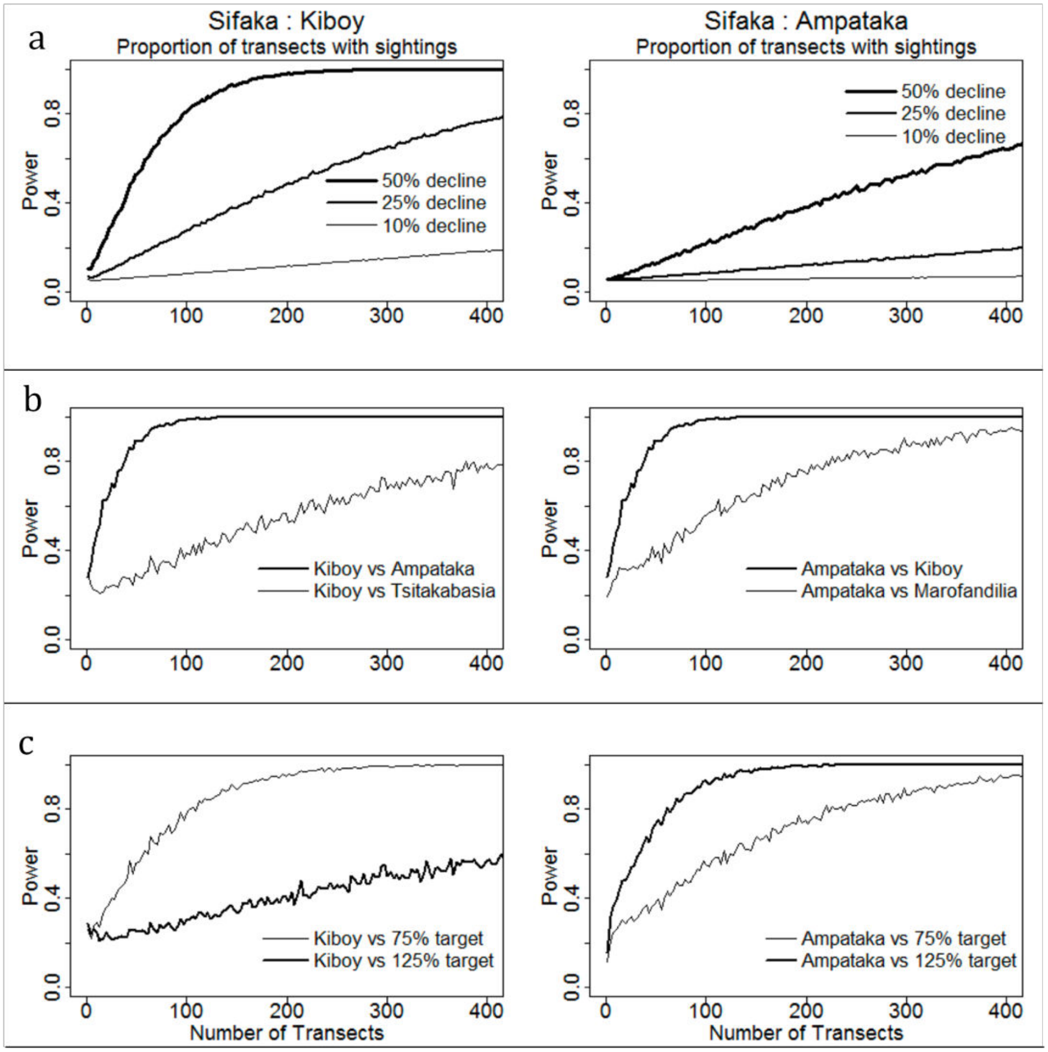


Figure 6-4 Example of power graphs for sifaka as transect numbers increase.
The relationship between the number of transects and power to detect change for a case study species and two communities (sifaka, in Ampataka and Kiboy). Similar results are obtained for all indicators and villages. α=0.05. a) Changes between two time periods, b) Differences between the **two communities' forests, and c) Differences compared to targets of 75% and 125% of average** indicator value

Comparison of results for individual indicators

No single approach was universally better for a given indicator. Figure 6-4 gives an example of this for one species, sifaka, in the two most geographically separated communities. Power to detect large reductions in sifaka numbers over time within a community forest was much higher in Kiboy than Ampataka. This reflects the fact that sifakas were observed on 38% of transects in Kiboy in comparison to 8% of the transects in Ampataka (Figure 6-4a). However, this difference in encounter rates meant that there was a high power to detect a difference between these two community forests, while detecting differences between neighbouring community forests took a great deal more effort (Figure 6-4b). Sifaka numbers were closer to the 75% target in Ampataka and to the 125% target in Kiboy, and this is reflected in the large number of samples needed to detect difference from these targets in each village.

6.3.2 Multi-indicator monitoring using accumulation curves

By the 10th transect, the biogeographic differences between the southern communities (Ampataka and Marofandilia) and the northern communities (Tsitakabasia and Kiboy) became apparent in the species accumulation curves, with large differences apparent between the two pairs of villages (Figure 6-5). After 40 transects, there was still a great deal of overlap within the regions, however (Figure 6-6a). The shape of the curves, with a rapid accumulation and a subsequent gradual increase, highlights the presence of ubiquitous species, such as the crested coua, and ebony (*Dyospyrus sp.)* together with rare or difficult to detect species, including flat-tailed tortoise and chameleons (*Fucifer sp.*).

The picture for threats is rather different, with no clear difference between northern and southern villages and Ampataka continuing to accumulate threats as transects increased.

In contrast, the number of threats observed in Tsitakabasia and Kiboy stabilised quickly (Figure 6-6c).

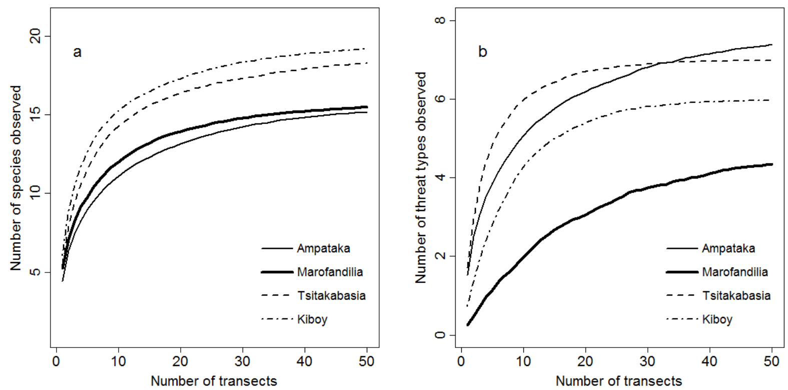


Figure 6-5 Species and threat accumulation curves.
Bootstrapped average over 500 transects of accumulation curves of a) species of interest (total possible number of species for each community forest = 27) and b) indicators of threats for four forests as effort increased.

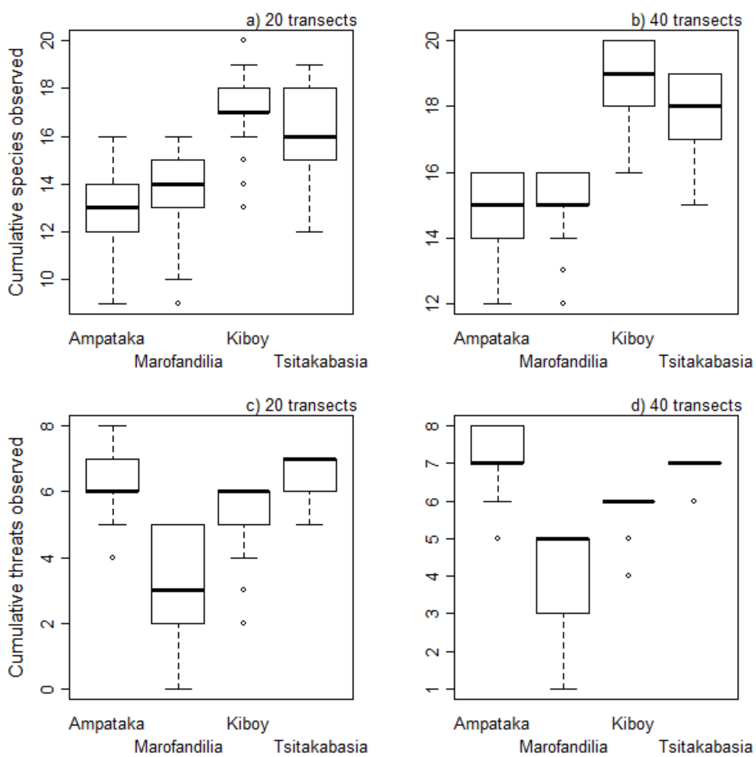


Figure 6-6 Box and whisker plots of species and threat accumulation curves.
At 20 and 40 transects for the accumulation curves of a) and b) species indicators and c) and d) cumulative threat types of observed from each of the four community forests (n=500 for each forest).

6.3.3 Remote sensing of threats

While hundreds of fires were detected in the Menabe region during 2007, mostly in already cleared agricultural fields, only five fires were noted in the community forests considered in this study. Two of these were within the forest of Kiboy and three were in Marofandilia. All were in areas that had also burned extensively in 2005. These new fire events were not detected on the transects we performed (Figure 6-7).

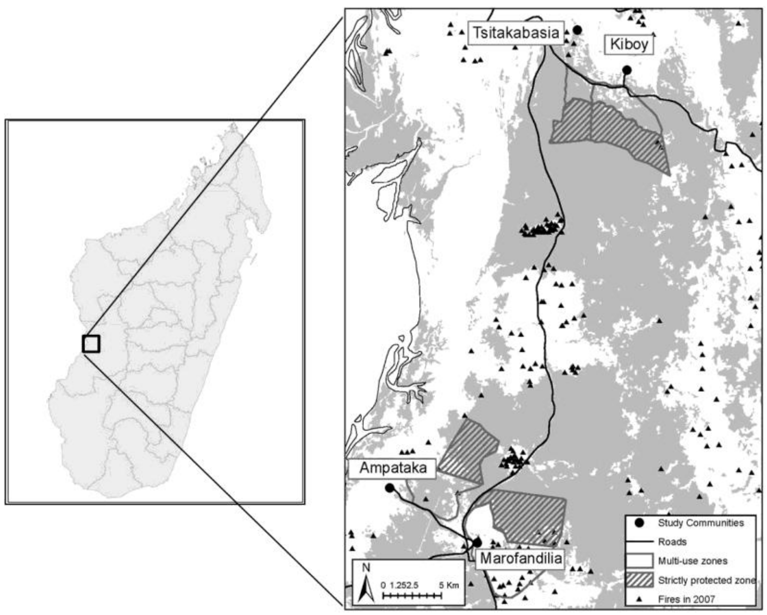


Figure 6-7 Map of fires detected by the FIRMS project in 2007.
The map of Menabe, Madagascar, includes the four strictly protected and multi-use community forests considered in this study. Gray areas on the Menabe map represent forest as of 2000 (Harper et al., 2008).

6.4 Discussion

6.4.1 Challenges of monitoring biodiversity services

The results of this study highlight the difficulties of monitoring biodiversity service provision for PES. Even with 40 transects per site (Durrell currently carry out only 1), the power to detect changes over time, differences between forests, or performance against a target, was very low for most species and threat indicators that were investigated. While species and threat accumulation curves tended to flatten out by 40 transects, they did not provide a statistical framework to assess the power of the accumulation curve approach at variety of monitoring efforts, as they provide a single estimate without confidence intervals.

Based on our knowledge of daily payments, subsistence costs and travel for each 10 day monitoring trip, I calculated that each transect costs approximately $30 and that two independent transects of 1500m could be performed by each biologist in a single day. As a result, even 30 transects per community would mean that as much was being spent on monitoring service provision as is currently being spent on payments to incentivise service provision in the Menabe. I do not believe that monitoring in the Durrell system is particularly costly compared with other PES programmes in developing countries (Asquith et al., 2008), though of course monitoring costs vary dramatically with accessibility (Danielsen et al., 2005).

6.4.2 Monitoring rare indicators

Although conservation managers may be interested in population trends of threatened species within the intervention area, and want to create incentives for protecting these particular species, it is unlikely to be possible to monitor rare species annually with adequate power to use the results as a basis for payments without a very large monitoring budget (Maxwell and Jennings, 2005). For example in the Menabe, three of the species of highest conservation interest in the area (flat-tailed tortoise, Madame Berthe's mouse lemur, and the striped mongoose (*Mungotictis decemlineata*)) were not encountered frequently enough to allow us to compare trends over time, and I could only detect differences between community forests where there were observations in one forest and none in the other. Previous work has emphasized that due to intrinsically low power to detect changes for rare species, managers may want to implement conservation interventions regardless of evidence of decline of rare species (Taylor and Gerodette, 1993). However, this poses a challenge in the case of PES where the intervention itself is intrinsically tied to monitoring. Thus, although a PES scheme may have conservation of rare species as a primary objective, in many cases, basing payments on estimates from relatively common proxy species that respond positively to management actions and negatively to disturbances may be more feasible.

6.4.3 Monitoring species or threats

This study suggests that small-scale threats are not necessarily easier to monitor than species presence. This is partially due to the fact that presence of threats was encountered relatively rarely in this study, and thus estimating differences in threats faced the same limitations as monitoring rare species. The spatial distribution of threats will also impact both detection and estimates. The detection of active hunting threats may pose a challenge due to the rapid decomposition of evidence such as animal remains, though traps may be readily detected (Rao et al., 2005; Coad, 2007). And while the clustering of threats, such as timber exploitation, may increase detectability, clusters also increase the variability of sighting across transects (Kenkel et al., 1989). Habitat variability also influences detectability (Bailey et al., 2004), and while some transects had areas of extremely dense vegetation, such pockets are found across the entire Menabe (Sorg et al., 1996). Monitoring threats posed additional challenges, as it may be difficult to monitor age and provenance of some activities, such as small-scale timber exploitation. Furthermore, if resource use is subject to seasonal closure, such as the collection of honey or tenrecs in the Menabe, illegal use in the closed season may leave a lasting impact on the status of the species that can be picked up by monitoring, but illegality cannot be ascertained. The accumulation curves for threat types did not level off across all forests sooner than those for species. As a result, in this case study, measures of small-scale threats based on forest transects did not necessarily provide a simplified measure of short-term conservation success, contrary to hopes expressed in the literature (Salafsky and Margoluis, 1999). These observations highlight the need to consider the implications of detectability and relative abundances of threats and species across the landscape in advance of deciding on species or threat indicators. Threat monitoring may be more applicable early in the implementation of a conservation project, or for before-after comparisons of intervention success, when threat observations are higher or more variable between sites.

6.4.4 Differentiating payments

Small and medium reductions over time within sites were only detected for relatively common indictors and in the Menabe this applied primarily to bird species due to ease of detection. In terms of difference among sites, a higher power to detect differences was achieved most frequently when comparing distant sites. In these circumstances it is possible that differences simply reflect relatively stable biogeographic differences and such a scheme would likely lead to similar relative levels of payment across years. The detection of differences from targets demonstrated potential for offering estimates of whether indicator sightings were greater than, less than or the same as targets, though establishing targets that have ecological significance, such as ecological thresholds, remains challenging (Groffman et al., 2006).

Despite the limitations of the above approaches to considering species individually, alternatives, such as aggregate species richness or presence/absence of a suite of indicators for a comparison among sites may offer advantages. In our case study, the accumulation curve approach allowed for consideration of all indicators under a single metric and demonstrated differences between community forests at a lower monitoring effort than using presence/absence of individual species. The fact that the accumulation curves levelled off earlier than other approaches examined in the study suggests that basing payments on this approach may be more cost-effective than individual measures of species or threat indicators. Policy makers can use weightings to account for the higher value of particular species. Indeed, the current Menabe monitoring scheme does this to demonstrate the relative conservation importance of a variety of local species.

However, basing a large percentage of a payment on rare indicator sightings, which are essentially stochastic, may lead to payments that do not reflect the true state of the system. Furthermore, in relatively homogenous habitats or those with only a few species of interest, an accumulation curve may not provide adequate information for calculating payments. In addition, the lack of statistical support for the observed differences in any given year is a drawback of the accumulation curve approach.

6.4.5 Ground-based sampling and remote sensing

In ground-based monitoring it is costly to detect enough sightings to estimate differences between sites or through time. However remote sensing also poses challenges in terms of matching freely or cheaply available datasets with local scale management needs. For example, the fire results from this study are not particularly enlightening for the management goal of differentiating among communities for payment. Furthermore, since the FIRMS dataset is best able to detect large-scale fires, the lack of fires in 2007 does not necessarily rule out the presence of small fires in the community forests. Indeed some were encountered in the transects, but not detected remotely, and the remotely detected fires were not encountered on the transects. Also the ecological damage caused by fires varies significantly with the fires detected in 2007 by FIRMS in areas of Kiboy's forest that had previously burned probably having a lower impact on the forest than similar fires in intact forest. While there has been increased availability of global remote sensing data, their applicability to PES monitoring will vary depending on monitoring questions and the spatial and temporal scales of interest.

6.4.6 Implications for Durrell intervention

Another important consideration when choosing indicators is the extent to which the indicators respond to the intervention. Only species that are directly targeted, such as the larger lemur species, birds and timber species, could see direct improvement in their status based on changes in behaviour or reduced threats through the Durrell intervention. The remaining species of interest, particularly the flat-tailed tortoise, giant jumping rat and Berthe's mouse lemur would likely benefit from lower anthropogenic pressure in general, however, the drivers of their declines are poorly understood, so with the exception of limiting habitat conversion, it is not clear whether payments for changed forest-use behaviours would directly impact the wider species of interest. As a result, if many of these indicators are not expected to respond to changes in actions by the individuals who are being paid, they may not be particularly useful.

6.4.7 Other roles for monitoring in PES

Monitoring for assigning payments does not need to occur in isolation. In addition to determining payment amounts, monitoring can provide benefits in the form of knowledge that may be useful for management purposes (Nichols and Williams, 2006), such as the location of access points for timber exploitation. In some cases monitoring can act as a tool influencing motivation to comply with rules (Chapter 4), or for collaborative engagement (Danielsen et al., 2005). In the best case, annual monitoring for payments would feed into longer term ecological monitoring, and the ecological monitoring would inform management decisions over a number of years. These decisions could include a review of the structure and magnitude of the payment mechanism, or whether specific reactive actions should be taken. Such processes reflect the tenets of adaptive monitoring; where long-term monitoring to answer well defined initial questions, underpinned by rigorous statistical design, can lead to new questions that address management needs (Lindenmayer and Likens, 2009). Indeed, in the Menabe system, locations of burrows of giant jumping rats were initially identified on payment transects and subsequently have been used to investigate specific questions regarding burrow occupancy over time (F. Rakotombolona, personal communication). Similarly, the sighting of lemur traps in one community's forest in 2004 led to investment in education programmes on the legality of lemur hunting by Durrell and many fewer traps have since been observed in that community's forest (R. Lewis, personal communication). The impact of monitoring is not limited to the knowledge gained for ecological or management purposes. Rather, engagement with participating communities can foster good will and collaboration, as well as encourage compliance with rules by increasing the probability of detecting violations (Danielsen et al., 2005).

While this chapter examined particular approaches to monitoring biodiversity services, there is inherent flexibility in PES schemes. Basing payments on a mixture of specific actions and the status of the conservation target, as the Durrell case study does, is a promising avenue for increasing the motivation of PES participants (DeCaro and Stokes, 2008). Yet while a broad suite of indicators may be monitored, the bulk of payments should be based on indicators or criteria that can be monitored with power. It is clear that monitoring the status of most biodiversity conservation indicators requires a substantial amount of effort and that this needs to be built into the planning and budgets of PES schemes. This lack of power at a reasonable cost for monitoring indicators in biodiversity PES interventions is likely to be faced by many biodiversity PES interventions in developing countries. This study does not suggest that PES for biodiversity conservation is not feasible, but rather urges careful consideration in decisions regarding the indicators that will be monitored and how they will be used to award payments if PES for biodiversity conservation are to live up to their potential.

Chapter 7. Discussion and conclusions

7.1 Context

Much of the world's biodiversity is concentrated in developing countries in the tropics (Baillie et al, 2004) and hundreds of millions of dollars have been spent in these nations over recent decades on biodiversity conservation (James et al., 2001; Balmford and Whitten, 2003). However, the success of these biodiversity interventions has been mixed (Struhsaker et al., 2005; Garnett et al., 2007). Commonly implemented conservation paradigms, such as integrated conservation and development projects and ecotourism, have received criticism for being unable to deliver adequate levels of conservation or adequate economic and social benefits to communities (Barrett and Arcese, 1995; Kiss, 2004). Furthermore, most of the drivers of biodiversity loss, including habitat conversion, eco-toxification, climate change, and direct exploitation by individuals, show few signs of slowing in the near future (Ehrlich and Pringle, 2008).

Over the past decade, payments for environmental services (PES) have been heralded in the natural resource management literature as a novel approach to create incentives for mangers to improve the provision of services including carbon sequestration, landscape benefits, watershed protection and biodiversity conservation (Ferraro, 2001; Landell-Mills, 2002; Pagiola et al., 2004; Engel et al., 2008). Given the recent increase of interest in managing habitats for environmental service production, there is a need for analyses of both the theory and practice behind PES, particularly with regards to the challenges posed by their implementation in developing countries. This dissertation makes important contributions to the literature by evaluating the implementation of Durrell's community-based PES in the Central Menabe, Madagascar. This project is one of the longer running PES schemes strictly for biodiversity conservation and therefore provided a useful case study to examine the opportunities and challenges of community-based PES interventions for biodiversity conservation. The next section (7.2) describes the framework for PES that was introduced in Chapter 2 and examines the contributions of this dissertation to elements of the framework. This is followed by a discussion of limitations of this study and topics for future research (7.3), as well as brief conclusions (7.4).

7.2 Contributions

7.2.1 A definition

The term PES has been widely used since the mid-1990s, yet its definition has been ambiguous for much of this time. A definition by Wunder (2005, 2007), which states that PES represents a voluntary transaction between at least one buyer and at least one seller that is conditional on service provision, has received wide support in the recent literature. However, the definition does not fully incorporate the range of interventions that many would consider PES (Swallow et al., 2007; Wunder et al., 2008). In Chapter 2, I presented a revised framework for PES that identifies two defining principles, net-positive incentives and conditionality, and highlights the importance of additionality and institutional contexts as key concerns for successful PES interventions. This simplifies

Wunder's definition and broadens its applicability, particularly to cases for biodiversity in developing countries. The remaining chapters in the dissertation presented novel insights into how these central characteristics to PES implementation function in practice.

7.2.2 Positive incentives and the local institutional contexts

In theory, net-positive incentives, typically in the form of monetary awards, act as the motivational tool within PES interventions to impact participants' behaviours. However, PES interventions rest within local institutional contexts that mediate the extent to which incentives impact behaviour. My research demonstrated the importance of considering the wider set of positive and negative incentives that accompany the implementation of community-based PES schemes, in addition to payments (Chapter 4), as well as the role of the distribution of incentives (Chapter 5).

The observation in my case study that payments had a relatively limited impact on the decision to change behaviours, and a stronger relative impact on individual attitudes, challenges the standard narrative of PES. In the Menabe, monitoring was the strongest driver of behavioural change (Chapter 4). Durrell's payments created a locally accepted justification for outside monitoring of environmental service provision. This impact of monitoring is likely repeated in other case studies with behaviours of varying legality, such as in Costa Rica's Pagos por Servicios Ambientales where a PES followed a new national law against deforestation on private lands (Pagiola, 2008). In these circumstances, the positive incentives act as a tool to make individuals amenable to a regulation. This may be a particularly useful approach for national governments or

NGOs in developing countries when national enforcement capacity is limited and/or where regulation of customary rights is concerned.

In addition to creating a justification for the acceptance of monitoring at the community level, the Durrell PES appeared to have empowered local forest management associations, as fear of local associations was a motivator of behavioural change in Durrell communities (Chapter 4). This study observed that changes in behaviour due to pressure from local communities were reported to be more stable than behavioural change influenced by Durrell or the national government. This evidence demonstrates the benefit of implementing interventions through existing local community structures and highlights the potential lasting impacts of interventions that use local institutions.

However, there are certainly challenges in realising net-positive incentives at both the individual and community level in community-based PES interventions. There are a variety of choices on how to distribute incentives in PES schemes related to the local institutional context and decisions on the part of service buyers (Chapter 2). In particular, poor governance and elite capture have been blamed for past failures of community-based interventions (Barnett et al., 2001; Fritzen, 2007). In-kind incentives have been used in many conservation and development programmes to increase transparency and avoid corruption (Balmford and Whitten, 2003; Wunder, 2005), however within the Menabe, there were frequent complaints of unfair distribution with respect to individual families taking advantage of the in-kind incentives (Chapter 5). Nevertheless, the preference for distributing in-kind, rather than direct incentives continues (Wunder, 2005), with implications for how PES interventions influence behaviour.

Non-excludable benefits, in the form of community parties, appeared to avoid cooption of benefits by individuals. The larger relative impact of parties on individual attitudes than in-kind incentives distributed by Durrell, as well as discussions with members of the local community demonstrated this (Chapter 4). However, community-wide benefits have a significant downside in that they are generally not able to address variable opportunity costs experienced within the community, leaving those with high costs of participation under-compensated (Chapter 5). Addressing this trade-off between distributing payments based on opportunity costs or equitable distribution of benefits is not straightforward and likely depends on local contexts. This topic requires further theoretical and field based research.

In contrast to the simplistic description of monetary payments directly influencing individual behaviours, it is clear that the impact of a community-based PES on individual motivation is based on a wider framework of positive and negative incentives and the local norms of community benefit distribution.

7.2.3 Conditionality

In all conservation interventions, managers need to know whether their conservation actions are having the desired impacts. As a result there is a need for monitoring. In PES this need is particularly acute as monitoring acts as the basis for making payments conditional. The challenges associated with developing meaningful biodiversity monitoring programmes are well documented and include funding limitations, unclear objectives and poor sampling design (Yoccoz et al., 2001; Legg and Nagy, 2005; Field et al., 2007). PES interventions address the issue of objectives through the indicators on which they base their payments. But despite these well-defined objectives, it is not clear that many programmes have adequate statistical power to detect the changes of interest.

This thesis presents the first study to look specifically at the issue of power in the design of monitoring programmes in PES for biodiversity conservation interventions (Chapter 6). No single approach to selecting indicators and tracking their change was universally advantageous. However, the results underscored the importance of considering monitoring costs in the assessments of whether PES interventions are feasible (Meijerink, 2008). This is particularly an issue for monitoring rare indicators, which are often of interest in biodiversity conservation PES schemes, but generally require large sample sizes to detect changes or difference on which to base payments. Indicators must reflect the service of interest and should be able to be monitored with power over appropriate time scales. Irrespective of the indicator that is chosen, minor changes or differences in indicator abundances are likely to require substantial monitoring effort and these costs need to be considered in implementing PES interventions (Chapter 6). My observation that monitoring influences behaviours (Chapter 4) provides added justification for the value of monitoring within PES, and for all conservation interventions.

Though there are challenges to biodiversity monitoring in PES, the need for monitoring (for payments to be conditional) offers opportunities for evaluating project effectiveness and motivating compliance by PES participants. The collection of ecological and social data for annual payment can feed into a framework for a wider evaluation of trends within or between habitats or communities (Chapter 4). This could act as an early warning system of ecological or social change or it could feed into an adaptive management framework (Cowling et al., 2008).

7.2.4 Additionality

While conditionality creates a mechanism for measuring whether an intervention meets its self-defined goals (service provision), the proof of additionality ensures that meeting these goals provides a benefit that would not have occurred otherwise (Ferraro and Pattanayak, 2006). This should be an ambition of all conservation interventions. By making additionality a common component of conservation planning, managers would be better able to demonstrate impact of interventions and this may create more empirical data on projects to help researchers move towards statistically rigorous evaluations (Agrawal, 2001; Garnett et al., 2007).

I evaluated whether the project created additional environmental benefits over what would have occurred in the absence of the payment scheme by examining self-reports of individuals in both participating and non-participating communities regarding behaviours before and after the intervention (Chapter 4). This time-series design with a control group and the use of mixed-effects models allowed me to isolate impacts from the intervention. Despite a reported decline in all resource-use behaviours in the region, there was limited evidence that this difference was attributable to Durrell's PES intervention, except with respect to lemur hunting. Even for changes in lemur hunting behaviour, it is not clear that the payments from the Durrell's work were the motivating force behind behavioural change. Instead regional education efforts appeared to drive the reported change (Chapter 4). As a result, the programme appeared to create limited additionality in terms of changes in local forest use behaviours. However the difference in the drivers of change between Durrell and non-Durrell communities was significant, and demonstrated a shift in responsibilities from fear of government in non-Durrell communities, to fear of local forest associations and Durrell's monitoring in the Durrell communities. This demonstrates a shift of responsibility towards those managing a PES and local institutions.

As a result, the Durrell intervention shows evidence of additionality in terms of social impacts through the empowerment of local community forest organisations in participating communities. While the literature on PES has addressed social impacts in terms of poverty alleviation (Pagiola et al. 2005), this is the first study to demonstrate improved resource governance at the community level through community-based PES. This represents a significant measure of success for the intervention, as increased engagement by communities in forest management was one of the primary goals of the intervention (Chapter 3).

7.3 Policy recommendations for Durrell

This dissertation used the Durrell Case Study from the Menabe to examine the social economic and ecological dynamics of PES for biodiversity conservation interventions and sought to generalise these lessons for the wider conservation and environmental service community. However, from the interviews and informal discussions I had with individuals in the Menabe, a number of general and location specific lessons were learned that improve the Menabe payment scheme. These recommendations are related to:

- the size of payments at the community and individual levels;
- the competitive framework for awarding payments among communities;
- the need for flexibility in addressing the differing issues facing individual communities;
- the ability to address threats posed to the Menabe forests by communities not engaged in the intervention;
- compensation of local community members who act as monitors;
- the role of communication and feedback to local communities;
- the relative importance given to effort vs. performance in awarding the communities and;
- the role of the implementing organization in helping local governance institutions take responsibility for management.

In terms of whether the size of the payments are sufficient to motivate behaviour change, it is clear that the payments themselves are not acting as the primary motivational force behind the decisions of most individuals in the community. However, payments certainly impact the decisions of members of the forest association board. If payments are to influence the decisions of the average family in the Menabe, they would certainly have to be greater, though the required value differs among families. Nevertheless, because behaviours are regulated (to varying degrees) and there is some monitoring on the part of both Durrell and the community, simply relying on more monitoring may be a more cost effective approach to encourage compliance than increasing the value of awards. As a result, before practitioners consider the size of payments, the motivational role of payments and how they may fit into a wider framework of traditional and government rules need to be considered.

Durrell initially established a competitive approach to payment distribution in order to promote motivation based on the goal of winning and to ensure that all funds are spent. This approach is unique to Durrell, but has gained support within the conservation community in Madagascar. Based on semi-structured interviews, it was evident that the competition produced a positive feeling for those communities that "won." In contrast, communities with consistently smaller payouts, such as Kirindy and Ampataka, had a negative feeling towards the competition that was beginning to poison their interest in continuing with the intervention. This was due in part to the perception that they cannot compete on a level playing field with communities that have larger forest areas or "better" and more biodiverse forest. The use of a competition created an initial assumption that there was a level playing field, when in fact communities soon discern that they are either advantaged or disadvantaged. Based on the negative responses of the losing communities and the danger of alienating these communities, I believe that it would be more beneficial to use a more consistent and less comparative scoring system.

While the Menabe PES provides a framework to address motivation for managing community forests across the Menabe, there is still a need for community specific interventions to accompany the PES. Even within the eight participating communities where I worked, there were community specific challenges. For example, in Lambokely agriculture expansion adjacent to the community was rampant, and though it would not be detected in the PES monitoring, it posed a long-term danger to the stability of forest cover in the Menabe. Similarly, in Ampataka the identification of logged areas within the protected forest has little efficacy if the local community does not have the capacity to stop the commercial loggers. This highlights the danger of focusing strictly on behaviours and species within monitored locations to the exclusion of site-specific threats, and underscores the need to maintain adaptive management at a landscape level.

This call for addressing issues at a landscape level is a challenge even within the relatively small area of the Menabe. While the intervention targets 10 communities, there are at least 10 additional established and emerging communities, using the forest for agriculture and products. The existence of forest management contracts for each of the participating communities is useful for identifying appropriate target communities, however it does not necessarily lead to a successful intervention across the landscape. This thus becomes a challenge of both the logistics of managing PES across numerous small communities and securing adequate funding to create incentives for each community in the landscape. Even if implementation is initially on a small-scale, there should be a plan for expanding to cover all communities or threats to the landscape over the long-term.

Within the scheme there has been some uncertainty as to whether local monitors should be paid for their work. Some forest association leaders were unsure if they were "permitted" to pay the monitors from the Durrell cash incentives. Similarly, one of the most frequent complaints from the forest monitors was that they received little benefit from spending at least one entire day each month uncompensated. An increase of Durrell's annual payment could ensure that each community receives a large enough award to cover fixed operating costs including monitoring, some of which may be used to advance Durrell's own scientific research, and an additional portion of the award could be based on performance. As a result, it may be useful to include fixed costs of local monitoring within the system. Novel cost-effective tracking systems could be used to prove that monitoring rounds have been undertaken, such as time-stamped digital photographs of the monitoring team at locations within the forest.

As demonstrated in Chapter 4, communication and education have important motivational roles to play. Communities cannot take action in line with the aims of the incentives from organizations like Durrell, unless they have feedback highlighting the success of their management actions and advice regarding future actions they can take. In addition, these sessions of human interactions where Durrell representatives present their perspectives on the efficacy of the intervention, and, importantly, listen to the perspectives of community members build capacity, trust and mutual understanding. Frequently community members, particularly in the communities off of the beaten track, expressed frustration at a lack of direction and uncertainty regarding the course of action they should take. While the efforts by Durrell to communicate and offer frequent feedback, both positive and negative, to communities is commendable, it could be improved. Such work is costly in terms of man-hours and travel, however these costs may be reduced by combining the work with Durrell's standard scientific surveys. Additionally, local radio has been used widely in the region for health, safety and conservation information and could be better exploited to provide region-wide information at a low cost.

Given the challenges in measuring the differences among community forests for individual indicators and the likely impact of variables such as the start time of transects, weather and the trajectory of the transects, decreased weight should be put on the results of transects for the forest indicators in the annual awards. Instead, indicators that can be monitored with less bias should be used. These may include governance indicators or remotely sensed indicators, like forest fires. This approach may reduce the randomness in the annual results and provide a greater incentive for action in establishing robust local institutions at the community level. Such indicators may be most appropriate not only within the community managed forest, but also within the wider landscape that is under the control of each community.

Presently there is not enough demand on the part of tourists or researchers in each of the community forests to make entrance fees an integral part of the incentives for community associations. In the course of the research, I observed a reluctance by the forest associations to charge Durrell for use of the community forests for research and conservation purposes. As a part of this push to increase the incentives to govern local forests, Durrell should make it a point to pay admission to perform monitoring or interventions within the community forests. While there may be some hesitancy, because the implementing organization is responsible for pushing/facilitating the agenda of the intervention, there is a need to demonstrate to local communities that the forest resource is valuable for a number of different non-use reasons. In most communities, the individuals in the forest association were not sure whether they should or could charge Durrell for research and monitoring activities in the community forests. Durrell should be upfront about insisting on paying these fees that help the local forest associations realize value from management.

Despite the challenges posed by benefit sharing within communities, monitoring, and the small size of the award for some communities, it seems that on-the-whole the Durrell intervention is unlikely to collapse. However, efforts should be made to address the concerns of the communities that are feeling marginalised. The Durrell field staff has built a large amount of goodwill between local communities and the NGO, and this will serve to ensure that the intervention will continue to be acceptable to local communities. Despite this social acceptance of the intervention and lack of direct confrontation, it seems that the scheme could be altered to provide more effective and sustainable behaviour change.

The recommendations presented here are by no means cure-alls. Nevertheless, they highlights some of the key issues on the ground that may influence the long-term sustainability of the incentive system. Thus, addressing them with the understanding that they require further monitoring and consideration will allow Durrell to take an adaptive management approach to implementing their payment intervention.

While further examination of other case studies in conservation may be needed to ensure that the results of this dissertation can be generalised beyond the Menabe, many of the conclusions of this work have been observed within experimental studies in other disciplines. However, it is possible that some of the conclusions may not hold in schemes with different institutional and payment structures, for example where payments are significantly larger, or in private property systems.

7.4 Limitations and future research

While this dissertation examined critical issues related to the implementation of payments for environmental services in developing countries, questions remain for both the theory and practice of PES. In particular, futher work is needed to expand on and challenge the conclusions of this dissertation on: the impact of local contexts; decisions on structuring incentives; the balance between social and ecological additionality; and the design and impact of monitoring within PES.

There is a need for more research on *how* the local context influences the feasibility and design of potential PES interventions. This dissertation demonstrated that it is possible to implement PES interventions in situations where behaviours are regulated and where individuals do not have full property rights (Chapter 4). However, the universality of these observations remains untested. Similarly, the dissertation describes the potential advantages and disadvantages of decisions by those implementing PES, such as whether incentives should be in cash or in-kind (Chapter 5); non-excludable or targeted; and how ease of monitoring impacts what makes an appropriate indicator (Chapter 6). However, the dissertation does not outline a rubric for deciding when each approach is most appropriate. This underscores the need for a wider range of cases studies to complement evidence from the Menabe and to develop a better understanding of institutional preconditions for success in community-based PES for biodiversity conservation.

Considering the interest that many have in using PES as a tool for poverty alleviation (Pagiola et al., 2005; Bulte et al., 2008), more research is required on the trade-offs and synergies between addressing social and ecological goals (Engel et al., 2008). This may be particularly important in community-based PES systems where it is challenging to identify how costs and benefits are distributed within a community (Chapter 5). The use of the imprecise metric of perceptions of “net-benefit”, “net-loss” and “unchanged” in individual responses was a limitation in this study, and could be improved by a more comprehensive dissection of costs and benefits at the individual and community level. In addition, the potential in community PES for the elite to capture benefits and use command and control methods to influence behaviours of the rest of the community could subvert the goal of PES to distribute net-positive benefits, and thus deserves more attention (Chapter 4).

In terms of the study design for determining additionality, the use of mixed effects models, a control group and before-after questionnaire provided a strong framework for analysis, but the sole use of reported behaviours was a limitation. While my key informant validation approach was useful when there was unanimity in responses, often the key informants contradicted one another and this therefore provided limited information on behaviours of individuals throughout the communities. There are well-documented challenges in eliciting honest responses from interviewees regarding regulated behaviours (Bernard, 2002). One approach for gathering information on illegal or embarrassing behaviours that has been used in the public health and criminal justice fields is the random response technique (Warner, 1965; Bowling, 2005). This method makes respondents' answers anonymous though provides data at the population level and this does not allow for modelling individual choices. Direct observation of individual resource-use would provide a method for verification, but would require greater effort, and thus likely limit the number behaviours that could be tracked. Furthermore, direct observations may be subject to strategic behaviour on the part of individuals avoiding detection. In order to address some of these challenges during the early stages of my research, I trialled methods using local individuals to collect data on behaviours of other members of the community, but faced challenges, such as illiteracy and data falsification by local assistants, that are common in such research. The lack of a baseline from each of the communities prior to the intervention also provided a challenge to measuring impacts that are common to conservation interventions (Kremen et al., 1994; Edgar et al., 2005), and there is a need for future evaluations of PES interventions to incorporate baseline data.

While the study highlighted the multiple roles of monitoring in PES for influencing individual changes in behaviours (Chapter 4), as the basis for payments (Chapter 6) and as a metric for long-term evaluation of effectiveness (Chapters 4 & 6), more research is needed in each of these areas. For example, it is not clear to what extent the strength of monitoring at influencing individual behaviours was due to the illegality of a number of forest-use behaviours in this study. Choosing appropriate indicators that reflect the biodiversity service of interest and can be monitored with statistical power will remain a challenge, but one that PES managers need to consider during intervention design. As a result, more empirical research is needed on the relationship between specific actions and the improvement of ecosystem service provision (Ehrlich and Pringle, 2008). Finally, future PES interventions should seek to develop frameworks to ensure that monitoring for service provision feeds into long-term monitoring of project effectiveness.

As a way forward on many of the issues raised in this section, a wider body of empirical data on case studies is needed. Furthermore, PES practitioners need to explore the theoretical and practical work performed in other applied fields, such as management, criminal justice and health to learn how incentives influence individual behaviours (Laffont and Martimort, 2001).

7.5 Conclusions

This dissertation represents one of the first detailed evaluations of a case study on community-based PES for biodiversity conservation in a developing country. It contributes to the understanding of how incentives interact in PES interventions to influence the behaviours and attitudes of individuals. It demonstrates the need for a contextual understanding of the communities where interventions are implemented in order to work toward a locally accepted distribution of incentives. Furthermore, it highlights the very real concern of identifying biodiversity indicators and developing monitoring schemes that have the statistical power to base payments off of and influence behaviours. If community-based PES is to succeed as an alternative and preferable method to traditional biodiversity conservation schemes, it is critical that the full range of incentives that accompany an intervention are considered and that the intervention is implemented contextually and monitored rigorously.

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Appendices

Appendix 1 List of indicators in Durrell monitoring

	Indicator		Measurement of indicator on transects
BIODIVERSITY			
Mammals	Giant jumping rat	*Hypogeomys antimena*	Number
	Striped mongoose	*Mungotictis decemlineata*	Number
	Sifaka	*Propithecus verreauxi*	Number
	Red-fronted lemur	*Eulemur fulvus*	Number
	Red-tailed sportive lemur	*Lepilemur ruficaudatus*	Presence
	Fossa	*Cryptoprocta ferox*	Presence
Birds	White-breasted mesite	*Mesitornis variegata*	Number
	Madagascar crested ibis	*Lophotibis cristata*	Presence
	Giant coua	*Coua gigas*	Presence
	Coquerel's coua	*Coua coquereli*	Presence
	Crested coua	*Coua cristata*	Presence
Reptiles	Flat-tailed tortoise	*Pyxis planicauda*	Number
	Labords chameleon	*Furcifer labordi*	Presence
	Nicosia chameleon	*Furcifer nicosia*	Presence
	Madagascar tree boa	*Boa sanzinia*	Presence
	Mantellid frog sp.	*Aglyptodactylus sp.*	Presence
	Colubrid snake sp.	*Heteroliodon sp.*	Presence
Plants	Hazomalany	*Hazomalania voyroini*	Number
	Baobab sp.	*Adansonia sp.*	Number
	Ebony	*Diospyrus sp.*	Presence
	Masonjoany	*Santalina sp.*	Presence
THREAT			
	New foot paths		Number
	New cart paths/ roads		Number
	New camps		Number
	New small tree stumps		Number
	New large tree stumps		Number
	Cut logs		Number
	Logs for canoes		Number
	Lemur traps		Number
	Evidence of lemur hunting		Presence
	Evidence of bird hunting		Presence
	Forest fires detected		Presence
	Deforestation		Presence
GOVERNANCE			
	Size of protected forest		3 categories
	Financial governance		3 categories
	Proportion of community members in association		6 categories
	Growth of association		3 categories
	Proportion of protected forest that is strictly protected		3 categories
	Application of rules		yes/no

Initiative to monitor rules	3 categories
Quality of reports	3 categories
Number of people at association meetings	4 categories
Participation of women	2 categories

Appendix 2
Social Questionnaire

BACKGROUND INFORMATION

1. Date:
2. Village name:
3. Location of interview: Village / Field
4. Interviewer:
5. Others present:
6. Household number (from community map):
7. Name:
8. Sex:
9. Age:
10. Ethnicity:
11. Married:
12. Children: Number:
13. Household size:
14. Born in village:
    a. If not, where:
    b. How many years here:

ASSOCIATION BACKGROUND

1. Are you/ have you been a member of the community forest management association?
    a. If so, are you a member of the board?
2. Reasons for membership:
    a. If member, why are you a member of the association?
    b. If not member, why do you think others have decided to join the association?
3. Reasons against membership:
    a. If member, why do you think others choose not to join the association?
    b. If not member, why have you chosen not to join the association?

KNOWLEDGE OF INCENTIVES

1. Do you know of:
    a. Durrell?
    b. The staff of Durrell?
    c. The vehicle used by Durrell?
    d. The competition?
    e. The award from the competition? (give example)
2. (if relevant) What was the award last year ____?

   a. What place were you last year ____?
3. What actions could the village take to get a higher award next year?
4. Has the distribution of the award within the community been fair?
   a. Why?
5. In relation to what your family has had to give up through the village's engagement with the Durrell management competition, have you (benefited, lost out, stayed the same) from the Durrell competition in the area?
   a. What specific benefits have your family received?
   b. What has your family had to give up?
6. In relation to what the village has had to give up through the village's engagement with the Durrell management competition, has the village (benefited, lost out, stayed the same) from the Durrell competition in the area?
   a. What specific benefits has the village received?
   b. What has the village had to give up?

PERCEPTIONS OF ORGANIZATIONS
1. What is good about **Durrell's** work in the area?
2. What is bad about **Durrell's** work in the area?
3. What is your opinion overall of **Durrell's** work in the area (very good, good, normal, bad, very bad)

Same as above with Forest Service; then another local NGO, Fanamby.

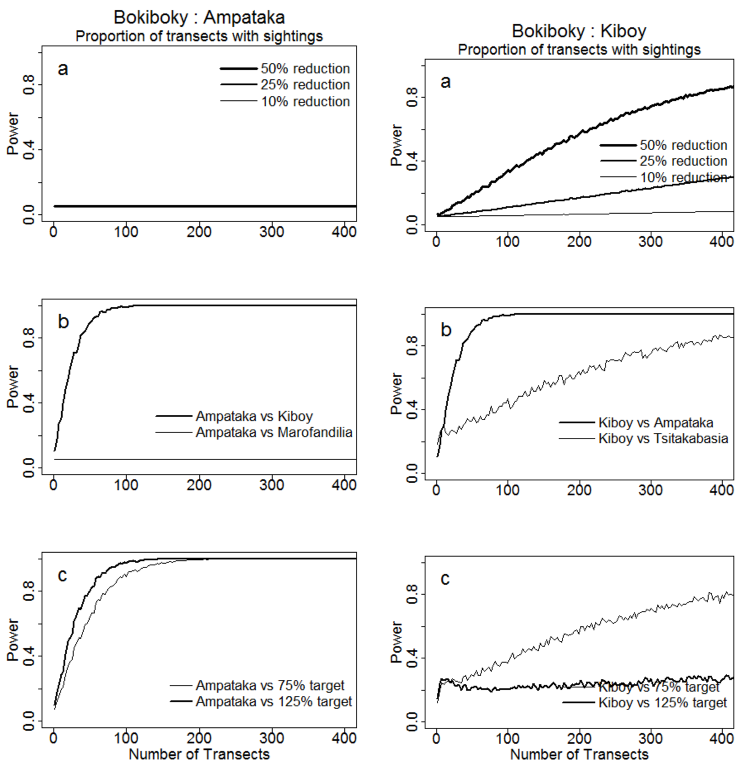
KNOWLEDGE OF SPATIAL LOCATION OF MANAGED FOREST BOUNDARIES
1. Where is the strictly protected forest?
2. Where is the multi-use forest?

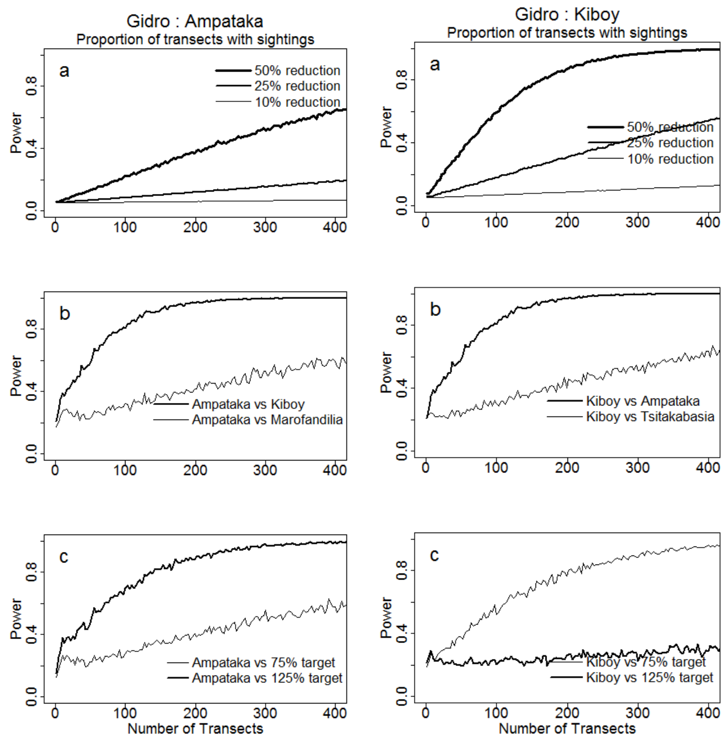
FOREST BEHAVIOUR QUESTIONS
1. How often do you use the forest in the rainy season?
2. How often do you use the forest in the dry season?
3. Did you harvest honey in the past?
4. Do you collect honey now?
5. Would you like to (continue to) collect in the future?
6. Do you sell honey? Do you consume it at home? Do you give it away? (proportions)
7. (if relevant) Did you collect (a description of area of the strict forest) during or before the year of the eclipse (reference point)?
   a. If no, why not?
8. (if relevant) Do you collect (a description of area of the strict forest) now?
   a. If no, why not?
9. (if relevant) Why did you change?
10. (if relevant) Why haven't you collected?
11. (if relevant) If there were no forest management rules, would you collect?
12. (if relevant) If there were no forest management rules, would you collect in the (a description of area of the strict forest)?

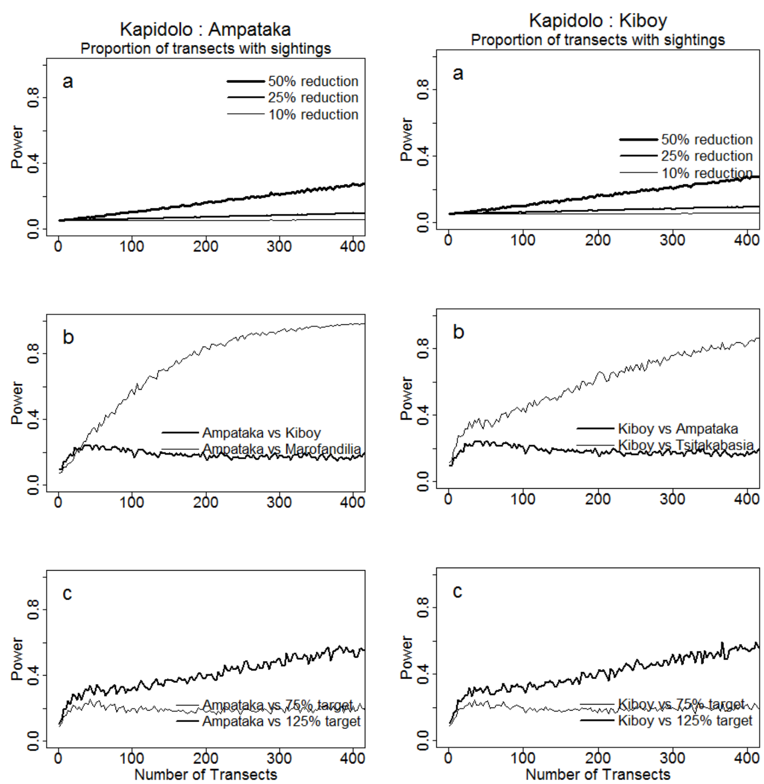
AGRICULTURAL WEALTH

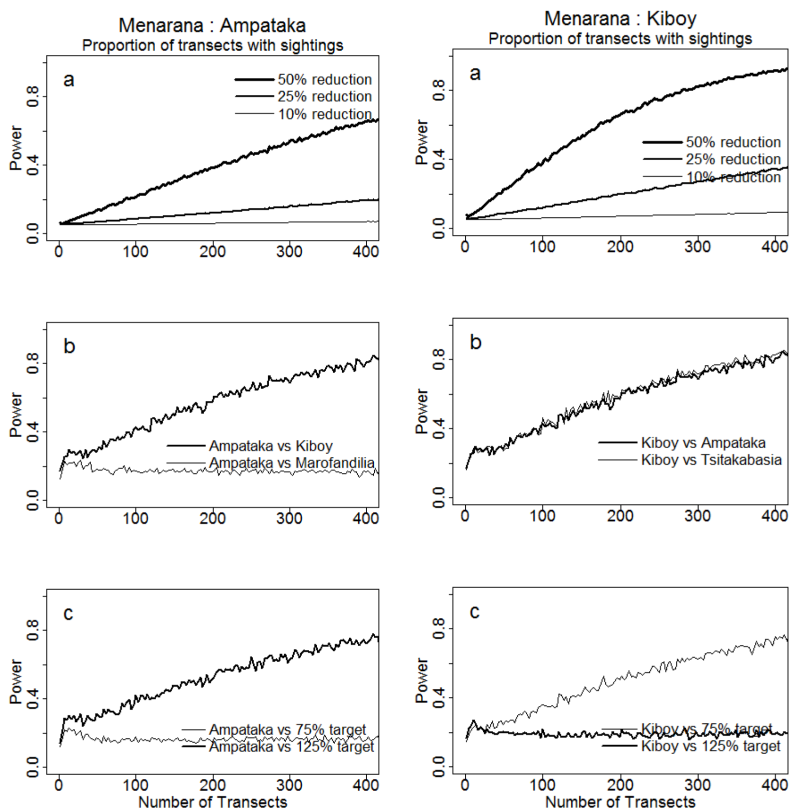
1. Do you have any dry fields?
    a. Relative to others in the village, do you have (less, more or same amount) of dry fields?
    b. Does it produce (more than enough, adequate, or not enough) for your family's needs?
2. Do you have any fields next to the forest?
3. Do you have any rice fields?
    a. Relative to others in the village, do you have (less, more or same amount) of rice fields?
    b. Does it produce (more than enough, adequate, or not enough) for your family's needs?
4. Do you have any wet fields?
    a. Relative to others in the village, do you have (less, more or same amount) of wet fields?
    b. Does it produce (more than enough, adequate, or not enough) for your family's needs?

Appendix 3
Additional power analyses for species with more than 30 sightings on transects.
Animal Observations

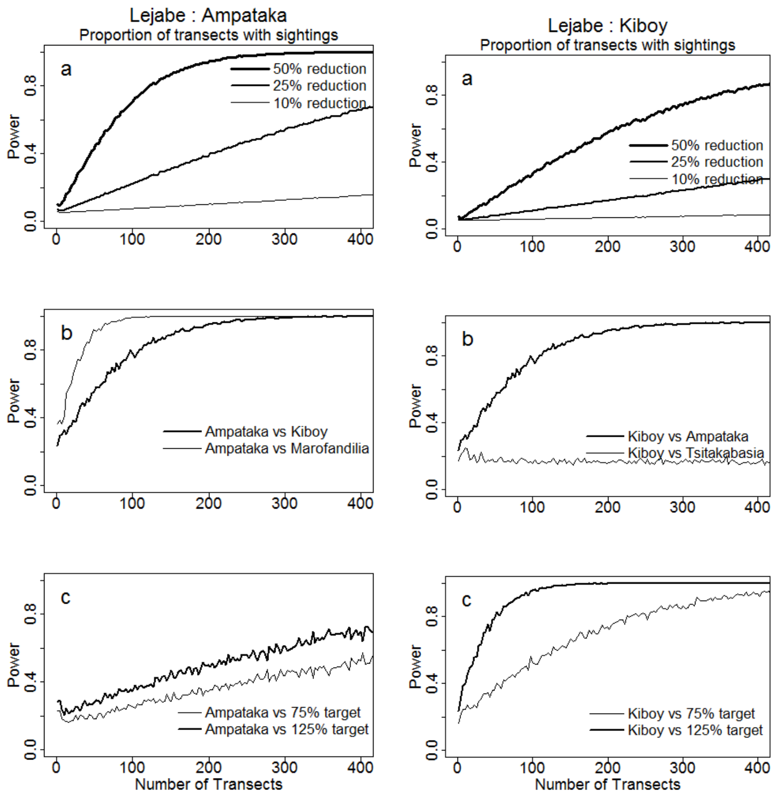


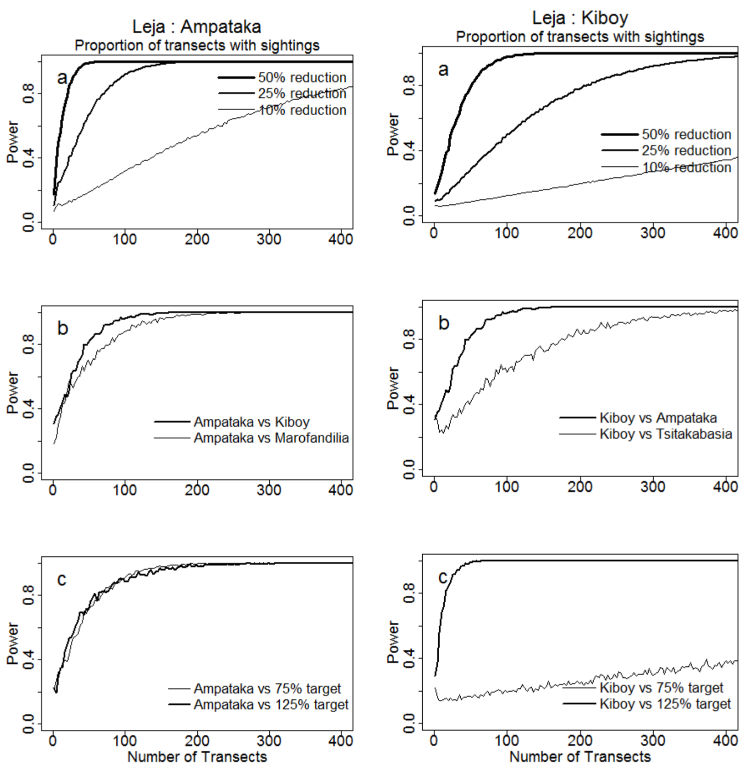

Gidro : Ampataka
Proportion of transects with sightings
50% reduction
25% reduction
10% reduction
Ampataka vs Kiboy
Ampataka vs Marofandilia
Ampataka vs 75% target
Ampataka vs 125% target
Gidro : Kiboy
Proportion of transects with sightings
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects

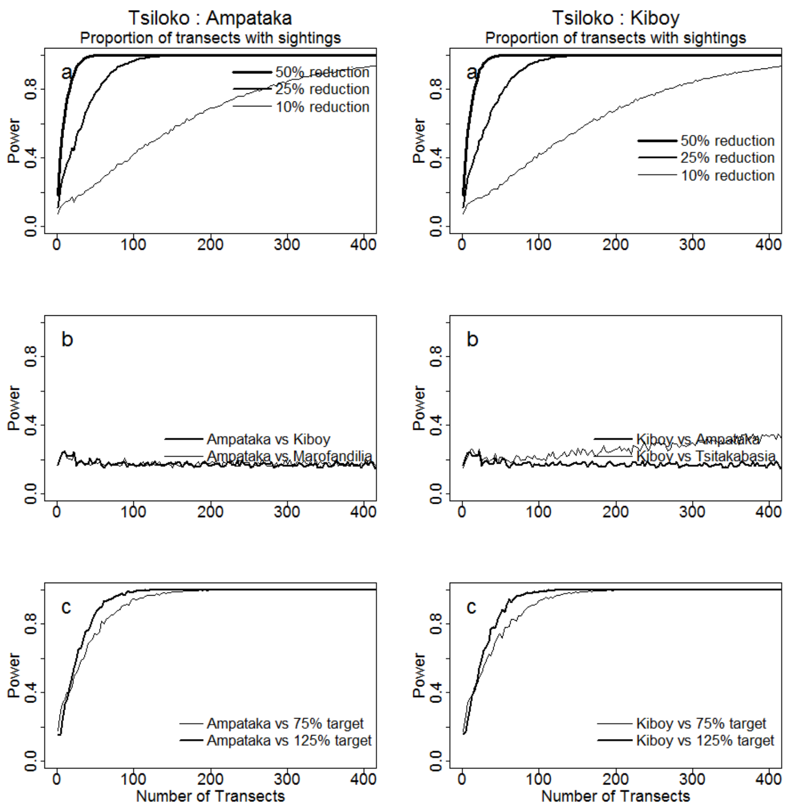

Kapidolo : Ampataka
Proportion of transects with sightings
50% reduction
25% reduction
10% reduction
Ampataka vs Kiboy
Ampataka vs Marofandilia
Ampataka vs 75% target
Ampataka vs 125% target
Kapidolo : Kiboy
Proportion of transects with sightings
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects



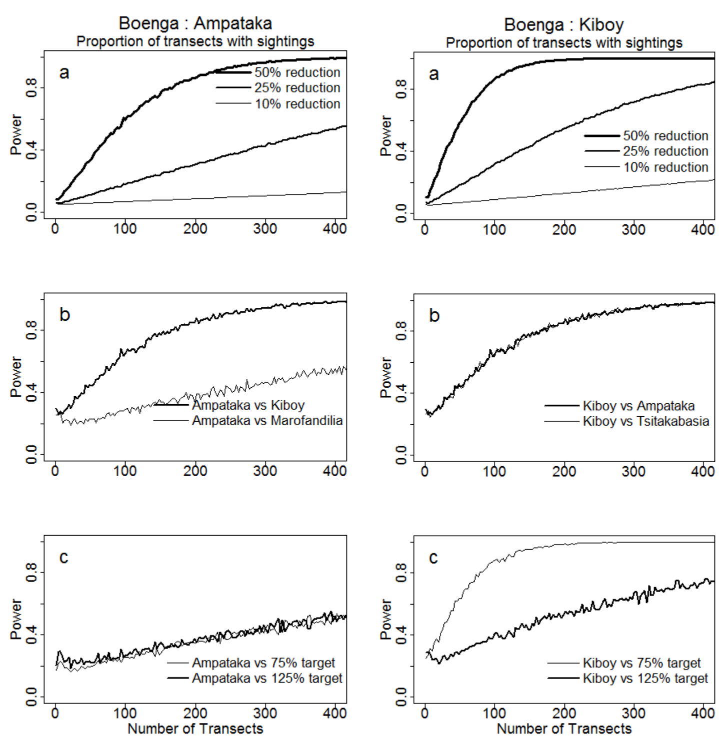
Bird Observations/ Calls

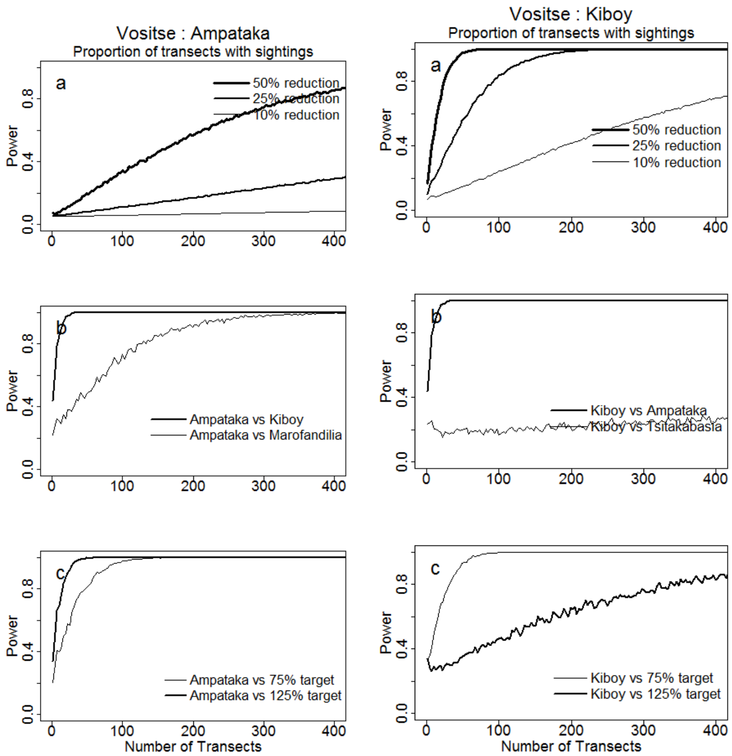

Lejabe : Ampataka
Proportion of transects with sightings
Lejabe : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
c
Ampataka vs 75% target
Ampataka vs 125% target
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects


Leja : Ampataka
Proportion of transects with sightings
Leja : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
c
Ampataka vs 75% target
Ampataka vs 125% target
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects

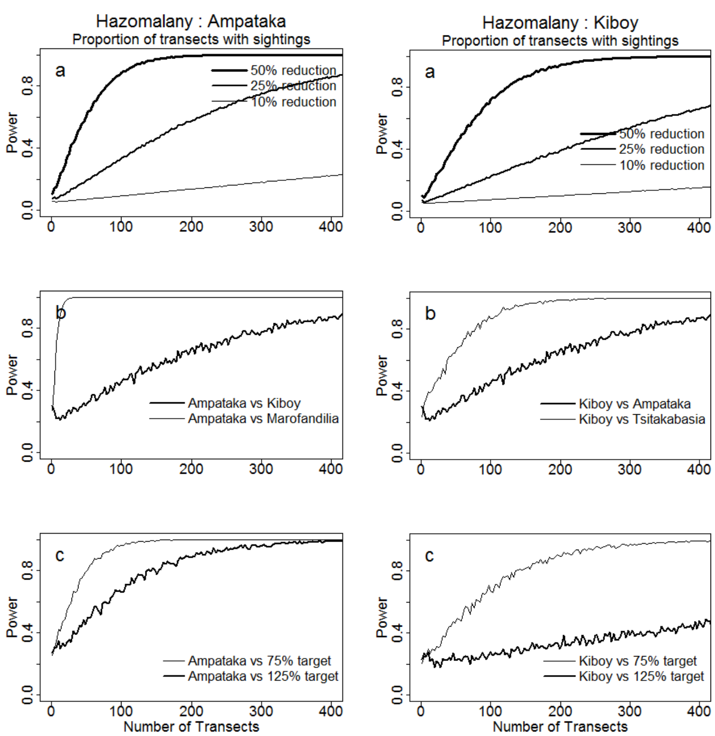


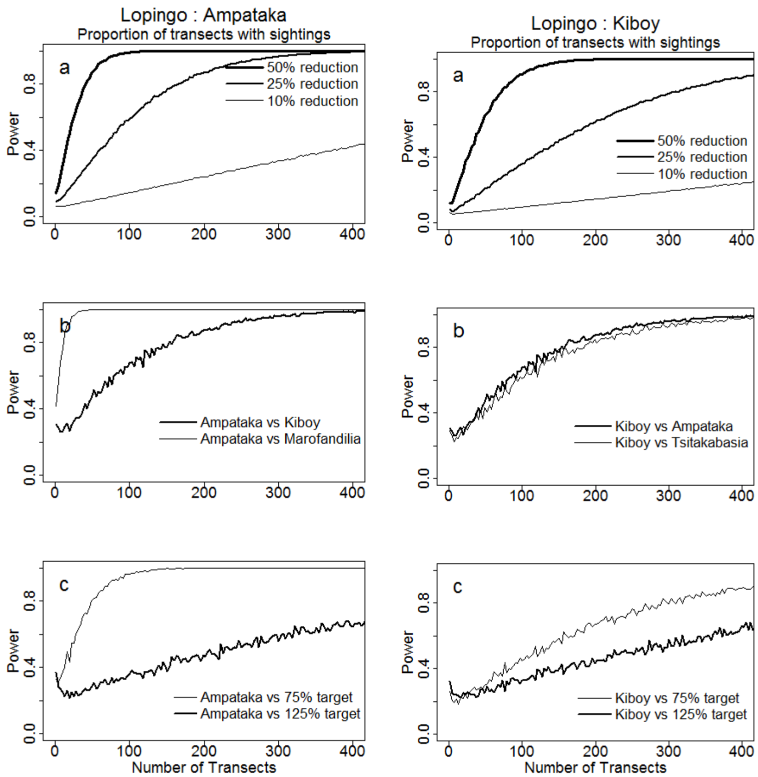
Animal Signs/Evidence

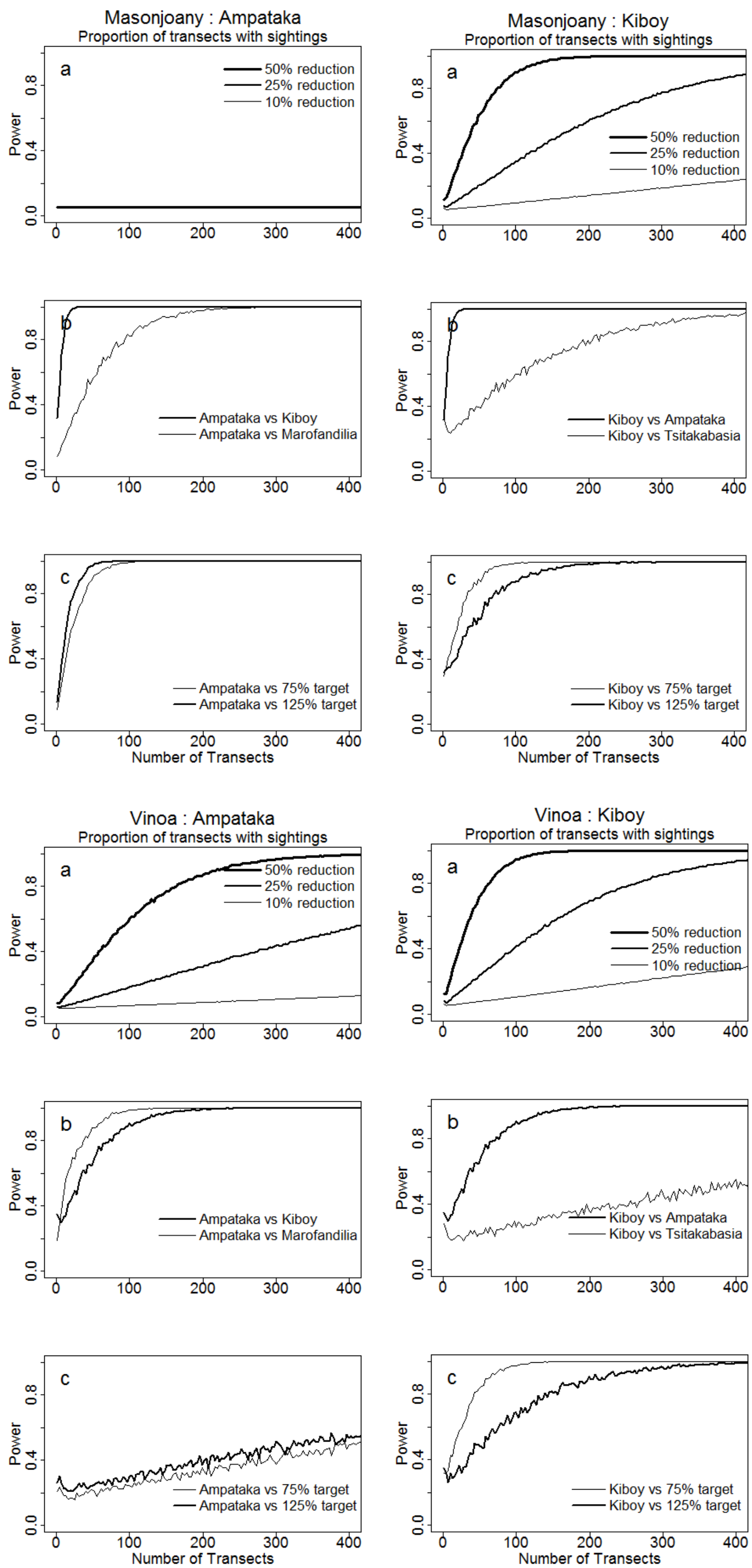



Vositse : Ampataka
Proportion of transects with sightings
Vositse : Kiboy
Proportion of transects with sightings
50% reduction
25% reduction
10% reduction
Ampataka vs Kiboy
Ampataka vs Marofandilia
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Ampataka vs 75% target
Ampataka vs 125% target
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects

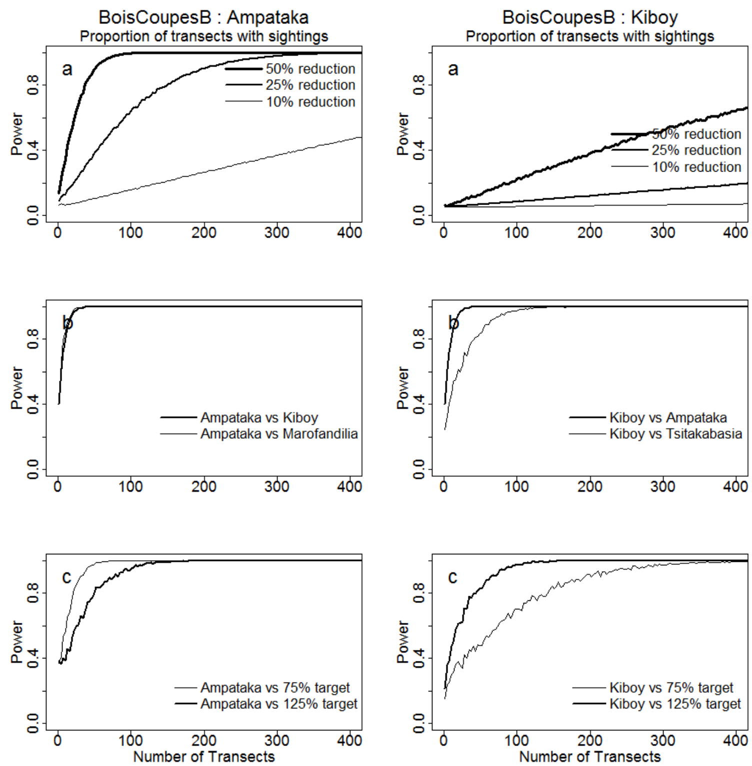
Trees

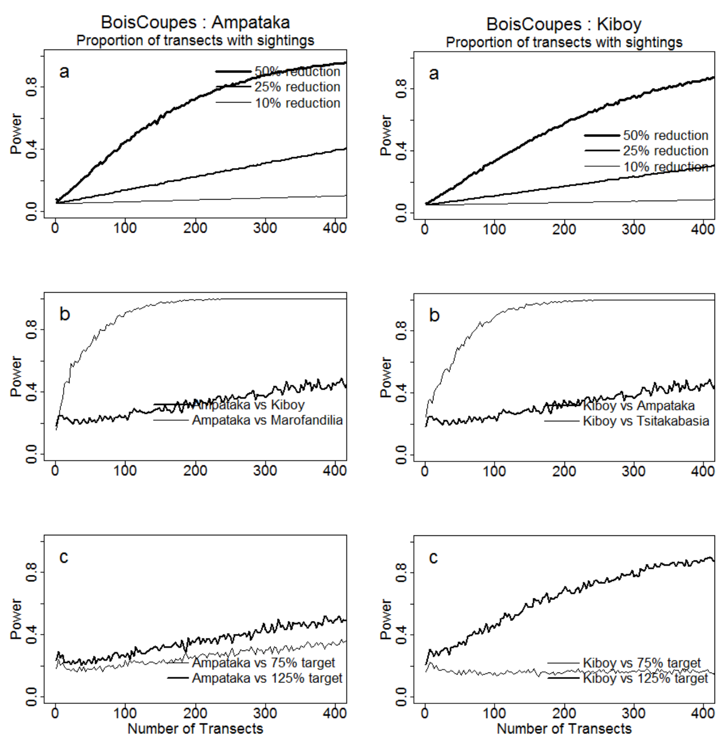

Hazomalany : Ampataka
Proportion of transects with sightings
Hazomalany : Kiboy
Proportion of transects with sightings
50% reduction
25% reduction
10% reduction
Ampataka vs Kiboy
Ampataka vs Marofandilia
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Ampataka vs 75% target
Ampataka vs 125% target
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects

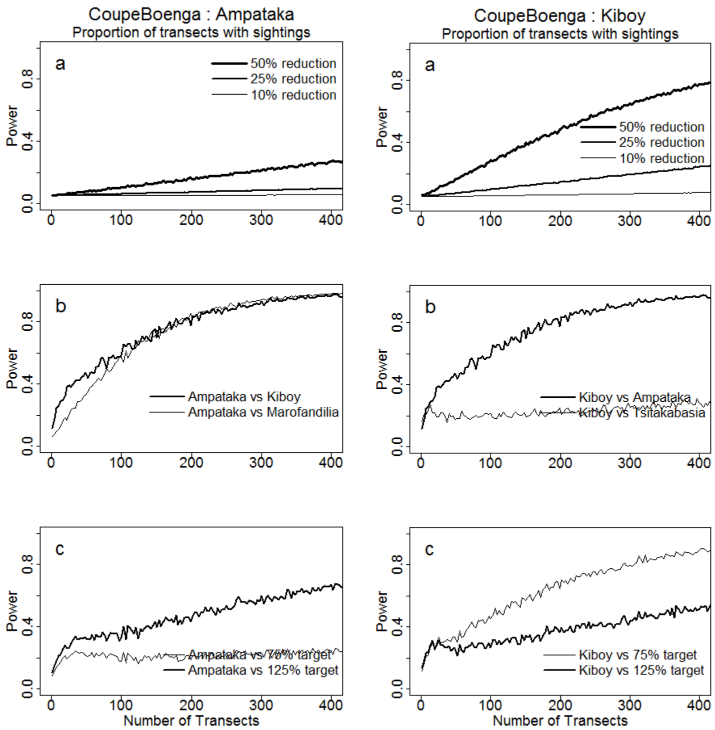

Lopingo : Ampataka
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
c
Ampataka vs 75% target
Ampataka vs 125% target
Lopingo : Kiboy
Proportion of transects with sightings
50% reduction
25% reduction
10% reduction
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects

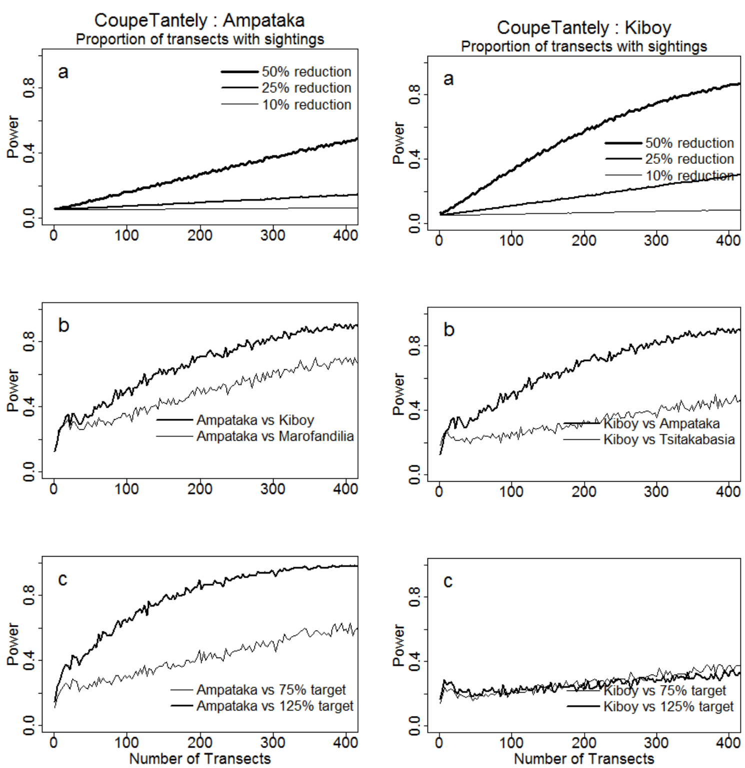

Masonjoany : Ampataka
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
Power
Masonjoany : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
Power
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
Power
b
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Power
c
Ampataka vs 75% target
Ampataka vs 125% target
Number of Transects
Power
c
Kiboy vs 75% target
Kiboy vs 125% target
Number of Transects
Power
Vinoa : Ampataka
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
Power
Vinoa : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
Power
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
Power
b
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Power
c
Ampataka vs 75% target
Ampataka vs 125% target
Number of Transects
Power
c
Kiboy vs 75% target
Kiboy vs 125% target
Number of Transects
Power

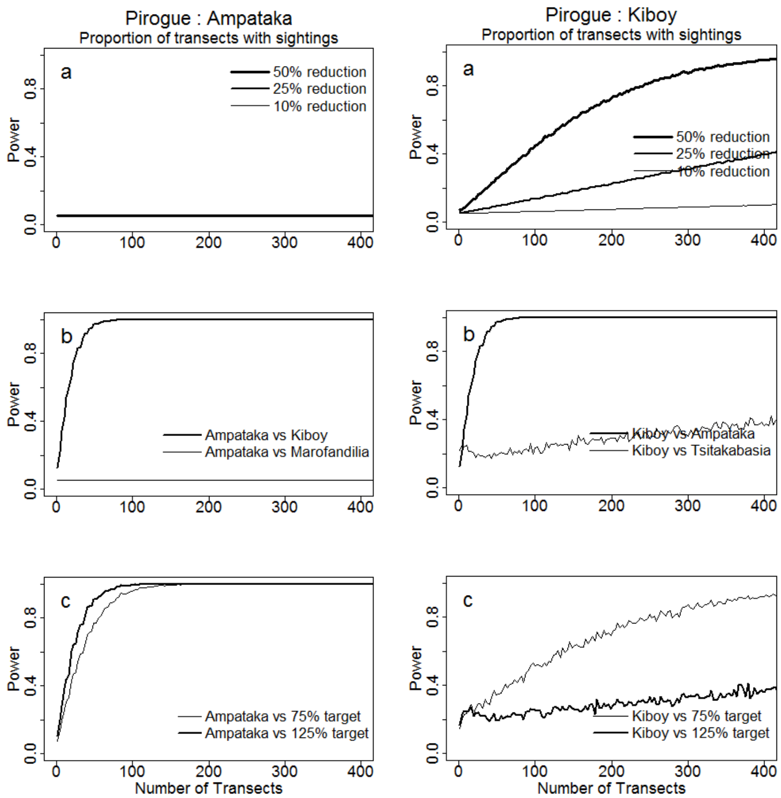
Threats


BoisCoupesB : Ampataka
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
c
Ampataka vs 75% target
Ampataka vs 125% target
Number of Transects
BoisCoupesB : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
c
Kiboy vs 75% target
Kiboy vs 125% target
Number of Transects
Power

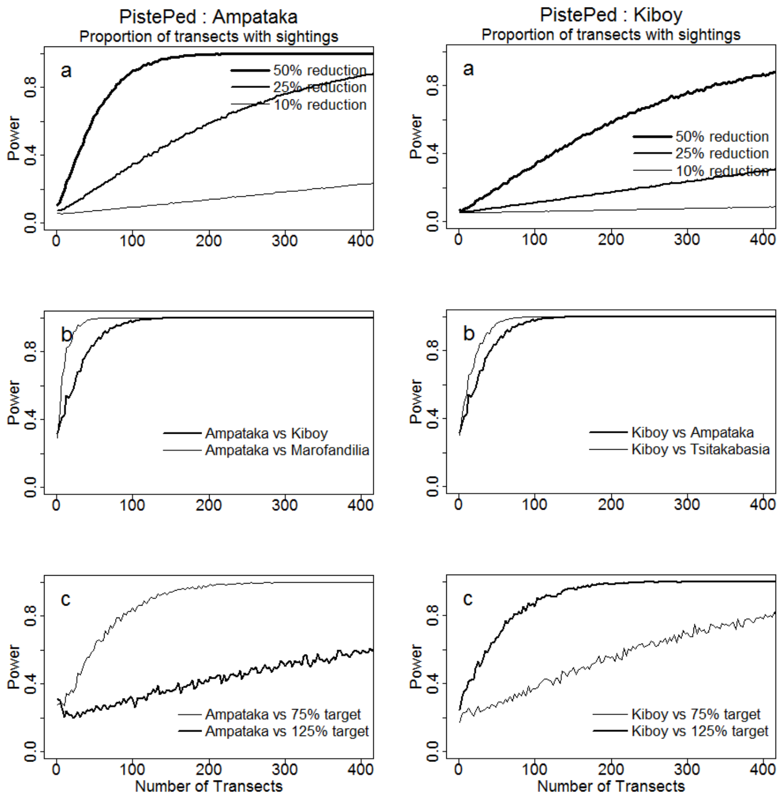

BoisCoupes : Ampataka
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
c
Ampataka vs 75% target
Ampataka vs 125% target
Number of Transects
BoisCoupes : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
c
Kiboy vs 75% target
Kiboy vs 125% target
Number of Transects
Power


CoupeBoenga : Ampataka
Proportion of transects with sightings
CoupeBoenga : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
c
Ampataka vs 75% target
Ampataka vs 125% target
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects


CoupeTantely : Ampataka
Proportion of transects with sightings
CoupeTantely : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
c
Ampataka vs 75% target
Ampataka vs 125% target
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects



Pirogue : Ampataka
Proportion of transects with sightings
Pirogue : Kiboy
Proportion of transects with sightings
a
50% reduction
25% reduction
10% reduction
b
Ampataka vs Kiboy
Ampataka vs Marofandilia
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
c
Ampataka vs 75% target
Ampataka vs 125% target
Kiboy vs 75% target
Kiboy vs 125% target
Power
0.0
0.4
0.8
0
100
200
300
400
Number of Transects


PistePed : Ampataka
Proportion of transects with sightings
50% reduction
25% reduction
10% reduction
Ampataka vs Kiboy
Ampataka vs Marofandilia
Ampataka vs 75% target
Ampataka vs 125% target
PistePed : Kiboy
Proportion of transects with sightings
Kiboy vs Ampataka
Kiboy vs Tsitakabasia
Kiboy vs 75% target
Kiboy vs 125% target
Power
Number of Transects