Money for Something? Investigating the effectiveness of biodiversity conservation interventions in the Northern Plains of Cambodia

Tom Clements

Darwin College

University of Cambridge

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Declaration: This dissertation is the result of my own work and includes nothing which is the outcome of work done in collaboration except where specifically indicated in the text.

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Summary

Despite substantial investments in biodiversity conservation interventions over the past two decades there is relatively little evidence about whether interventions work, and how they work. Whether an intervention is deemed to "work" depends upon how goals are defined and then measured, which is complex given that different stakeholders have very different expectations for any intervention (including species conservation, habitat protection, human wellbeing or participation goals), and because the process of measuring impacts can involve a simplification of more sophisticated ideals. These questions were investigated for a suite of biodiversity conservation interventions, implemented during 2005-2012 in the Northern Plains landscape of Cambodia. The interventions included the establishment of Protected Areas (PAs), village-level land-use planning, and three different types of Payments for Environmental Services (PES) instituted within the PAs. The PES programmes were (*1*) direct payments for species protection; (*2*) community-managed ecotourism linked to wildlife and habitat protection; and (*3*) payments to keep within land-use plans.

The impact evaluation compared the results of each of the interventions with appropriate matched controls, considering both environmental and social impacts between 2005-2011. Both PAs and PES delivered additional environmental outcomes: reducing deforestation rates significantly in comparison with controls and protecting species for those cases where appropriate data was available. PAs increased security of access to land and forest resources for local households, benefiting forest resource users, but restricting households' ability to expand and diversify their agriculture. PES impacts on household wellbeing were related to the magnitude of the payments provided: the two higher-paying PES programmes had significant positive impacts for participants, whereas a lower-paying programme that targeted biodiversity protection had no detectable effect on livelihoods, despite its positive environmental outcomes. Households that signed up to the higher-paying PES programmes, however, typically needed more capital assets and hence they were less poor and more food secure than other villagers. Therefore, whereas the impacts of PAs on household wellbeing were limited overall and varied between livelihood strategies, the PES programmes had significant positive impacts on livelihoods for those that could afford to participate. This is one of the first evaluations of the social impacts of PES that has been completed globally.

Summary

The PA authorities were primarily effective at deterring external drivers of biodiversity loss, especially large-scale developments, land grabbing and in-migration, and had much more limited impact on local residents as the impact evaluation results demonstrated. The PES programmes had little or no effect on the external drivers, and instead explicitly targeted the behaviour of local residents. The three PES programmes differed in the extent to which they rewarded changes in individual or collective behaviour, and whether or not they were managed locally or externally. Household-level, conditional, payments were more effective at changing individual behaviour than collective payments; although there was evidence that both types of payments did lead to protection of forests at the village scale. Village-managed PES programmes empowered a subset of households that were then effective at enforcing regulations within the village. Externally managed PES programmes were more popular and viewed as fairer, but did not change collective behaviour. The general conclusion is that the design and institutional arrangements of PES programmes determines how participants perceive the programmes, and then the extent to which they bring about changes in behaviour.

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Tom Clements

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Abbreviations

3ie	International Initiative for Impact Evaluation
AICc	Akaike's Information Criterion corrected
BNS	Basic Necessities Survey
BPS	Buddhism for a Progressive Society
CALM	Establishing Conservation Areas through Landscape Management (CALM) in the Northern Plains of Cambodia
CBD	Convention on Biological Diversity
CCTs	Conditional Cash Transfers
CENTDOR	Center for Development Oriented Research in Agriculture and Livelihood Systems
Danida	Danish International Cooperation Agency
DFID	UK Department for International Development
DFW	Department of Forests and Wildlife of MAFF, now the Forestry Administration
eCDF	empirical cumulative distribution function
eQQ	empirical quantile-quantile
GEF	Global Environment Facility
GPS	Global Positioning System
ICDPs	Integrated Conservation and Development Project
IEG	Independent Evaluation Group of the World Bank
IUCN	International Union for the Conservation of Nature
JICA	Japanese International Cooperation Agency
KPWS	Kulen Promtep Wildlife Sanctuary
KS	Kolmogorov-Smirnov test
MAFF	Ministry of Agriculture, Forestry and Fisheries, Royal Government of Cambodia
MEA	Millennium Ecosystem Assessment
MLMUPC	Ministry of Land Management, Urban Planning and Construction, Royal Government of Cambodia
MoE	Ministry of Environment, Royal Government of Cambodia
MPWT	Ministry of Public Works and Transportation, Royal Government of Cambodia
NCCD	National Committee for Sub-National Democratic Development
NGO	Non-Government Organisation
NONIE	Network of Networks Impact Evaluation Initiative
NTFP	Non-Timber Forest Product
NZAID	New Zealand's International Aid and Development Agency
OECD	Organisation for Economic Cooperation and Development
PA	Protected Area
PES	Payments for Environmental Services
PVPF	Preah Vihear Protected Forest
REDD+	Reducing emissions from deforestation and forest degradation in developing countries; and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries.
TEEB	The Economics of Ecosystems and Biodiversity
UN OHCHR	United Nations Cambodia Office of the High Commissioner for Human Rights
UNDP	United Nations Development Program
UNEP-WCMC	United Nations Environment Program – World Conservation Monitoring Centre
UNFCCC	United Nations Framework Convention on Climate Change
WCS	Wildlife Conservation Society
WWF	World Wildlife Fund for Nature

Chapter 1. The Design and Evaluation of Biodiversity Conservation Interventions to achieve multiple goals in the context of weak Institutions

> *"What is the most effective way to slow deforestation? How can we reduce poaching of protected species in low income nations? Do conservation [interventions] lead to changes in behaviours that affect biodiversity?*
>
> *For far too long, conservation scientists and practitioners have depended on intuition and anecdote to guide the design of conservation investments. If we want to ensure that our limited resources make a difference, we must accept that testing hypotheses about what policies protect biological diversity requires the same scientific rigor and state-of-the-art methods that we invest in testing ecological hypotheses."*
>
> — Money for Nothing? A call for empirical evaluation of biodiversity conservation investments (Ferraro & Pattanayak, 2006)

1.1 The Policy Dialogue

Human dominance of the earth is leading to unprecedented changes in the world's natural ecosystems (Vitousek et al., 1997; MEA, 2005a) and climate (Stern et al., 2006; IPCC, 2007), causing increasing use and scarcity of environmental resources, such as land (Lambin & Meyfroidt, 2011), tropical forests (Hansen et al., 2008) and biodiversity (Pimm et al., 1995; Butchart et al., 2010; Laurence et al., 2012). The loss of environmental resources and ecosystem services has global consequences, and is potentially of huge significance to the livelihoods of hundreds of millions of some of the most vulnerable people worldwide (TEEB, 2010; Vira & Kontoleon, 2010; Turner et al., 2012). The urgency of the problem has led to repeated, high-profile, calls for action (Oates, 1999; Wilson, 2002; Terborgh, 2004; Butchart et al., 2010; Laurence et al., 2012), and has stimulated a plethora of policy forums under the umbrella of the Rio conventions adopted at the 1992 Earth Summit (on Biodiversity, Climate Change and Desertification[1]). New global funding facilities have been established to disperse billions of dollars to finance action[2]. Average spending on biodiversity conservation alone by bilateral and multilateral donors is now estimated at more than $1.5 billion per year (Miller et al., 2012). Nevertheless these amounts fall considerably below what is required

[1] Convention on Biological Diversity (CBD; www.cbd.int), the United Nations Convention to Combat Desertification (www.unccd.int), and United Nations Framework Convention on Climate Change (UNFCCC; www.unfccc.int).
[2] Such as the Global Environment Facility (GEF), which has provided >$10.5 billion in grants for over 2,700 projects in over 165 countries since 1991, and the German Government International Climate Initiative, which has provided >$850 million to climate change projects since 2008.

to protect biodiversity (James et al., 2001; Balmford & Whitten, 2003) or to reduce greenhouse gas emissions from deforestation of tropical forests (Eliasch, 2008), leading to calls for increases in future funding by an order of magnitude or more (Eliasch, 2008).

Designing policies to conserve and sustainably mange environmental resources and biodiversity is however extremely challenging, given the complex and dynamic nature of social and ecological systems, which are often poorly understood (Ostrom, 2007). Policy responses in developing countries are often based on narratives that provide an argument for a particular action or intervention based on a simplification of this complexity (Hirschmann, 1968; Roe, 1991; Mosse, 2004). Strong narratives can become convincing arguments for change, particularly when supported by powerful actors, such as national Governments, donor aid agencies and international organisations, and can lead to the mobilization of large amounts of funding for standardised blueprints for action that may or may not be effective (Pritchett & Woolcock, 2005). Biodiversity conservation in particular has been characterized as a 'crisis discipline' (Pullin, 2002; Wilson, 2002); its focus on rapidly altering ecosystems, biological extinctions, and the ultimate loss of biodiversity defining a discipline that lacks the luxury of time (Chapin III et al., 2000) and is chronically under-funded (Balmford & Whitten, 2003). In this context, strong narratives for action have a particular appeal. However, history tells us that great ideas designed to improve human societies can sometimes fail with quite startling results (Scott, 1998) and standardised policy blueprints have been condemned as ineffective or destructive (Korten, 1980; Rondinelli, 1983 & 1993). Institutional failure in developing countries means that imported western ideals of how bureaucracies and institutions 'should' function may not work (Barrett et al., 2001); although institutional failure is also common in natural resource management in developed countries (e.g. fisheries; Acheson, 2006). This is not to say that great ideas are doomed to failure, but rather to emphasise that, amidst the rush to act, the process of simplifying the social, economic, political and institutional complexity of the problem when designing solutions carries inherent risks.

Strong narratives are also often associated with wider political trends, in order to achieve greater legitimacy, mobilise additional sources of funding, and encourage action by different stakeholders (Mosse, 2004). Recent examples have included the uneasy integration of conservation and poverty reduction goals (Adams et al., 2004; Roe, 2008), or conservation and climate change, adoption of ecosystem services language (e.g. Naidoo et al., 2008), and the rise of neoliberalism in conservation (Büscher, 2008). The integration of biodiversity conservation goals into wider political trends leads to proposed approaches that are expected to achieve multiple goals (so-called 'win-win'). These include biodiversity conservation and poverty reduction (e.g. through integrated conservation and development projects, ICDPs; Hughes & Flintan, 2001), and the adaptation to (e.g. CATIE, 2011) or mitigation of global climate change (e.g. through REDD+[3]; Clements, 2010). Political trends can also dictate mechanisms by which outcomes should be achieved, inherent in the neoliberal market-based approaches such as Payments for Environmental Services (PES) and REDD+. David Brown and Neil Bird (2008) write with respect to REDD+: "policy development is problematic in arenas that are excessively 'mechanism driven'. The approach needs to be turned on its head, and the mechanism subordinated to the problems it is trying to address". Given the plethora of ambitious policy narratives with unrealistic expectations for multiple 'wins', some authors have reached the conclusion that good development policy is unimplementable (Mosse, 2004).

An unanswered question is whether or not any of these policy instruments 'work', with perceptions of success or failure being formed often in the absence of any significant evaluations of programme impacts (Ferraro & Pattanayak, 2006). This has led to repeated calls for the adoption of evaluation techniques in the environment and development sector in the expectation that this would lead to better information regarding the extent to which policies are effective at achieving their stated goals (e.g. Kleiman et al., 2000; Pullin & Knight, 2001, Saterson et al., 2004; Sutherland et al., 2004; Stem et al., 2005; Ferraro & Pattanayak 2006; Frondel & Schmidt 2005; Pullin & Knight, 2009; Angelsen et al., 2011). Evaluating 'success' or 'failure' of interventions is far from straightforward, given the multiple, synergistic, competing or over-burdened goals of policy (Adams et al., 2003; McShane et al., 2010), and the lack of consensus over how outcomes should be measured or assessed (e.g. Agrawal & Redford, 2006). The important research question is therefore not whether a policy 'works' or not, but how to achieve a more nuanced understanding of impacts: **what are the conservation outcomes attributable to a particular implementation of a policy tool, in a specific environmental and social context? And what are the positive and negative impacts of the intervention, both socially and environmentally, and from whose perspective?**

Measuring the impacts of a policy intervention is, however, of superficial value without an analysis of how observed changes were brought about. As standardized blueprints for conservation are proposed for new countries or areas based upon global priorities (e.g. Myers et al., 2000) and used to mobilise funding, the top-down planning and resource mobilization process may or may not then be reconciled with the local social, political, economic and ecological realities that the programme

[3] REDD+ is formally defined by the UNFCCC as: Reducing emissions from deforestation and forest degradation in developing countries; and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries.

ultimately must address if it is to achieved its stated goals (Mosse, 2004). And in turn the policy process may or may not be co-opted by local agents for their own ends (Büscher, 2010). As these processes play out, no two implementations of the same policy prescription are likely to be very similar, and making generalisations about whether a policy prescription 'works' or not is potentially highly risky. Coleman (2009), writing about different forest management interventions, says that the point is to move beyond blueprint thinking such as that one particular tenure intervention is always best – whether that be community managed, protected area or otherwise – and focus on the particular management needs of specific forests.

Effective policy implementation therefore needs to be grounded in a strong understanding of what causes the loss of biodiversity and ecosystem services, including an analysis of the agents driving these changes, the proximate and underlying causes of those agents' actions, and finally how the interventions alter this complex socio-ecological system in order to bring about observed results (Salafsky et al., 2002; Margoluis et al., 2009). Understanding these processes is necessary if conservation science is to move beyond pattern-based descriptive analyses (or priorities, impacts, global trends, case study descriptions) and towards a more fundamental understanding of the ecological, economic, political and social processes driving the loss of biodiversity and ecosystems, and how policy interventions can alter these complex underlying processes at different spatio-temporal scales to deliver the desired outcomes. Given this, a second priority for research is: **how does policy implementation influence the dynamics of complex socio-ecological systems in order to bring about observed changes which can be attributed to that policy?**

This thesis considers these two themes – understanding impacts and the underlying processes – with respect to the implementation of policy prescriptions for biodiversity conservation in the Northern Plains of Cambodia. In this chapter I initially briefly review some of the common blueprints for conservation action, focusing on the management of tropical forests. I then assess the impact evaluation literature to understand how to apply evaluation methodologies to complex socio-ecological problems. Finally, I address the underlying causes of biodiversity and ecosystem loss from first principles, drawing upon the theoretical literature to consider the design of environmental programmes, focusing both on human behaviour and the interaction between individual behaviour, institutions and external drivers. The chapter concludes by outlining the main research questions, based upon the two overarching themes, and describing the outline of the thesis.

1.2 Common Blueprint Approaches

Command and Control: Protected Areas

Protected areas (PAs) are the traditional cornerstone of conservation (Bruner et al., 2001), currently covering >12% of the terrestrial land surface (UNEP-WCMC, 2012). They have, however, proved highly controversial, particularly in developing countries (West et al., 2006; Adams & Hutton, 2007; Roe, 2008). Establishment of the first modern protected area networks in Africa, Australia and North America was driven by colonialists' ideals of the preservation of supposed 'wilderness' areas free from human influence, which was achieved in some cases through resettlement of local human populations (Adams & Hulme, 2001; Hutton et al., 2005; Adams & Hutton, 2007). The extent to which this 'fortress conservation' ideal has been achieved is unclear: well-funded strictly enforced protected areas from which people have been resettled or excluded certainly exist, but in many countries protected areas lack funding and political legitimacy ('paper parks'; Wilkie et al., 2001; Balmford et al., 2003; Chape et al., 2005; Joppa et al., 2008). The current paucity of funding for protected area management is such that global expenditure on protected areas in the developing world has been reckoned, roughly, to be less than one twentieth of that needed (James et al., 1999; Balmford et al., 2003). Protected areas may also contain substantial human populations. A recent review concluded that the majority of evictions from protected areas occurred in the past and in particular regions of the world (e.g. East and Southern Africa; Brockington & Igoe, 2006). Seventy per cent of a non-random sample of protected areas contained people (Bruner et al., 2001), whilst a global review indicated that 56-85% of protected areas in developing countries have resident human populations (Brockington & Igoe, 2006). IUCN (the International Union for Conservation of Nature) consequently formally identifies a range of protected area types, from strictly protected areas and wilderness areas, to multiple use areas that combine protection with sustainable use of natural resources, such as by local people (Dudley, 2008).

The decline in support for 'fortress conservation' has been associated with mainstream agreement that, if conservation is to be sustained, protected areas must achieve social and political legitimacy, including recognition of the rights of local people and meeting their development aspirations. Debates have focused on whether the environmental goals of protected areas are compatible with poverty alleviation goals, especially in developing countries (Adams et al., 2004). There is now widespread acceptance that conservation policies such as protected areas should, at the very least, do no harm, and where possible should contribute to poverty alleviation (CBD, 2008). Protected areas can impose costs on local people, such as restrictions on agriculture or access to natural resources (Brockington & Igoe, 2006; Cernea & Schmidt-Soltau, 2006; Coad et al., 2008), displacing and resettling people (Brockington & Igoe, 2006), or through human-wildlife conflicts (Woodroffe et al., 2005), impacting both their livelihoods and their human rights. Of Bruner et al.'s (2001) non-random sample of protected areas, 54% had residents who contested the ownership of some percentage of the area. Protected areas may also benefit local people, by encouraging and promoting local forest resources use, for example through improved marketing or safeguarding access rights, exclusion of outsiders to create local monopolies, and providing alternative pathways out of poverty through employment and business opportunities (Wunder, 2001; Scherl et al., 2004; Coad et al., 2008). How the costs and benefits of protected areas trade off to determine overall net impacts (positive or negative) to local people is unclear and under-researched. Only three studies have evaluated the social impacts of protected areas in developing countries, and in these cases protected areas had no net impact or slightly positive impacts for local people (Andam et al., 2010; Sims, 2010; Naughton-Treves et al., 2011). The impacts of protected areas may also be marginal in comparison with broader economic trends as villages become integrated into national, regional and global markets (e.g. Brashares et al. 2004).

The evidence base regarding the effectiveness of protected areas as a conservation intervention is considerable and broadly more positive. Protected areas can be very effective at protecting habitat if sufficiently well managed (for reviews see Albers & Ferraro 2006; Joppa & Pfaff 2011; Nelson & Chomitz, 2011; Ferraro et al., 2012; Geldmann et al., 2012). A systematic review of 57 studies documenting habitat change found that in 53 of these the rate of habitat loss was lower inside protected areas when compared with a counterfactual scenario (Geldmann et al., 2012). Several authors have reviewed the literature regarding the effectiveness of different types of forest tenure, comparing protected areas with community forest management (Hayes, 2006; Robinson et al., 2011; Porter-Bolland et al., 2012). In general, these reviews conclude that the pattern is mixed – in some cases deforestation rates are higher in protected areas (Porter-Bolland et al., 2012), and in some cases they are higher in community managed forests (Robinson et al., 2011). Regardless of whether the forest is government-owned or community-owned, rule enforcement has been shown to be a significant predictor of forest condition (Gibson et al. 2005; Chhatre & Agrawal, 2008; Coleman 2009).

The effectiveness of protected areas at reducing rates of decline in wildlife populations has been more mixed (Craigie et al. 2010); a recent review by Geldmann et al. (2012) concluded that 29 of 40 studies reported that protected areas were effective in protecting target species populations, when compared to the counterfactual scenario. The large number of pattern-based studies, documenting trends and impacts over time, has prompted the authors of at least one global review to call for more research on why protected areas are effective in addition to measuring how effective they are (Geldmann et al., 2012).

Changing Incentives: Payments for Environmental Services (PES)

Although the global benefits of biodiversity conservation and ecosystem services are well recognised (Daily, 1997; Stern, 2006), these benefits are often valued differently at the local level (Kremen et al., 2000). In particular the costs associated with conservation may be disproportionately borne locally, and the benefits felt globally (Balmford, et al., 2002). Payments for environmental services (PES) and direct payments for conservation have been proposed as mechanisms to translate external, non-market values of the environment into real financial incentives for local actors to provide such services (Ferraro, 2001; Ferraro & Kiss, 2002; Wunder, 2007; Engel et al., 2008). PES thus seeks to internalize what would otherwise be an externality (Pagiola & Platais, 2007). The enthusiasm for PES and direct payments builds on the documented failings of indirect approaches that aimed to reduce human livelihood dependence on wildlife and natural resources, such as integrated conservation and development projects (ICDPs; Wells et al., 1998). Advocates argue that PES may be both more effective and cost-efficient as a mechanism for encouraging local actors to deliver conservation outcomes in a way that also provides local benefits, in comparison with alternative interventions such as ICDPs (Ferraro, 2001; Ferraro & Kiss, 2002). The popularity of PES is to some extent linked to the increasing use of conditionality and performance measures to distribute aid and subsidies (e.g. conditional cash transfers; Fiszbein & Schady, 2009).

PES have been defined as voluntary transactions where a well-defined environmental service is bought by a buyer (i.e. someone who is willing to pay for it), if and only if the provider secures the provision of such service (the conditionality criterion; Wunder, 2007). In effect, PES programmes attempt to put into practice the Coase theorem, which stipulates that the problems of externalities, under certain conditions, can be overcome through private negotiation between affected parties, assuming that transaction costs are low and property rights clearly defined (Coase, 1960; Engel et al., 2008). The largest global PES programmes are run by governments in developed countries, such as conservation easements in the United States of America or the Common Agricultural Policy in Europe (Ferraro & Kiss, 2002). These programmes conform to the Coasean view: land ownership or resource tenure is clearly defined, these rights are protected by law, enforcement agencies are well funded, and there are credible external monitoring systems. Within the past 10-15 years a number of government-financed PES programmes have been established in developing countries with similarly well-defined institutional frameworks (Engel, et al., 2008), including the Costa Rican payments for environmental services programme (Zbinden & Lee, 2004; Pagiola, 2008) and Mexico's payments for hydrological environmental services programme (Muñoz-Piña et al., 2008). In addition, there are a growing number of user-financed programmes, such as payments for watershed services between downstream users and upstream forest owners in Ecuador (Wunder & Albán, 2008) and Bolivia (Asquith et al., 2008), and contracts brokered between organisations and private landowners, communities or governments for biodiversity conservation (Ferraro & Gjertsen, 2009; Milne & Niesten, 2009). In the vast majority of cases, but not all, these PES programmes have been established in situations where property rights are usually (though not always) clear, although other aspects of the institutional framework may be weaker. Many of these PES programmes do not conform exactly to Wunder's original definition. In order to encompass the wide range of PES and PES-like interventions, Sommerville et al. (2009) have consequently proposed a revised definition for PES as approaches that aim to (*1*) transfer positive incentives to environmental service providers that are (*2*) conditional on the provision of the service, where successful implementation is based on a consideration of (*1*) additionality and (*2*) varying institutional contexts. The Sommerville et al. definition is used throughout this thesis.

Wunder (2007) suggested that effective implementation of PES may be considerably more difficult where institutions are weak. Institutional failure is problematic for implementation of a PES programme to protect biodiversity for a number of reasons: poorly defined property rights makes it challenging to determine who to pay, contracts cannot be legally enforced, elite capture is common, and enforcement of laws (e.g. prohibiting land clearance) may be weak. However, institutional failure makes it challenging for any conservation intervention to succeed (Barrett et al., 2001), hence a critical area for research is to understand which approach is most effective given these circumstances.

In contrast to the protected area literature, there have been very few studies of the effectiveness of PES as a conservation intervention. A recent PES review concluded "we do not yet fully understand either the conditions under which PES has positive environmental and socioeconomic impacts or its cost effectiveness" (Pattanayak et al., 2010). Ferraro and Kiss (2002) have hypothesised that the effectiveness of payments will depend upon the extent to which incentives are conditional upon performance. Wunder et al. (2008) suggested that PES may not be cost-effective in government-financed cases because side objectives, such as poverty alleviation, will affect programme design, weakening the conditionality criterion; by contrast they suggest that user-financed PES programmes may be more effective. Similar claims have been made regarding the effectiveness of government-managed conditional cash transfer (CCTs) programmes, with political economists suggesting that CCTs are used to make social security (i.e. unconditional payments) politically acceptable, rather than designed to achieve their stated targets (such as boosting school attendance rates; Pritchett, 2012). Perhaps unsurprisingly, therefore, the few evaluation studies that have been completed for government-financed PES have suggested that these programmes have had limited additional environmental impact (Pattanayak et al., 2010). Ferraro et al. (2012) state that no evaluations of user-financed PES have yet been completed. No studies have investigated whether or not PES contributes to local poverty alleviation.

1.3 Evaluation of Programme Impacts

Measuring the impact of environmental policies is increasingly viewed as critical in order to assess their contribution to achieving their stated goals. In order to determine impacts, researchers have drawn upon rigorous impact evaluation methodologies from health and development economics, which are widely credited with having transformed development policy (Banerjee & Duflo, 2011). The World Bank's Independent Evaluation Group defines impact evaluation as "*the systematic identification of the effects, positive or negative, intended or not, on individual households, institutions, and the environment caused by a given development activity such as a programme or project*" (IEG, 2011). Rigorous impact evaluation uses experimental and quasi-experimental techniques to compare the outcomes of the intervention with the counterfactual: what would have happened in the absence of the intervention. Applied in this way, impact evaluation methodologies have a huge advantage over other evaluation techniques (such as theories of change; Richards & Panfil, 2010); they assess the degree to which changes in outcomes can be attributed to the intervention as opposed to other factors and they estimate the magnitude of impact attributable to the intervention (Ferraro, 2009; IEG, 2011). Rigorous impact evaluation methods have been applied to fields as diverse as health and nutrition (Habicht et al., 2009; Gaarder et al., 2010), education (Glewwe et al., 2006), agriculture (Duflo et al., 2008), microfinance (Banerjee et al., 2010), water and infrastructure (van der Walle, 2009; Waddington et al., 2009) and several manuals are now available (Kusek & Rist, 2004; Ravallion, 2006; Khandker et al., 2010). The enthusiasm for impact evaluation in conservation is linked into the rapid expansion of neoliberal policies to conserve environmental resources and ecosystem services, such as market-based mechanisms (PES, REDD+; Wunder, 2007) and payment-for-results (Ferraro & Kiss, 2002). These new policies and interventions *require* quantitative measurement of results in order to assess values and determine the conditional incentives given to service providers.

Measuring the impacts of environment policy is also important because although new policy initiatives abound, in practice the suite of on-the-ground intervention types is relatively limited. Within this context, there is a huge need to re-evaluate the evidence base to understand how different interventions contribute, separately and synergistically, to achieving poverty alleviation, environmental, climate change or governance goals. The evidence base concerning the impacts of different interventions is very weak (MEA, 2005b; Pattanayak et al., 2010). Examples include community forestry (Somanathan et al., 2009) and the impacts of protected areas on environmental goals (Albers & Ferraro 2006; Andam et al., 2008; Joppa & Pfaff, 2011; Nelson & Chomitz, 2011; Ferraro et al., 2012) and poverty (Andam et al., 2010; Naughton-Treves et al., 2011; Sims, 2010). Expanding the application of impact evaluation tools is necessary to provide critical information to guide policymakers.

A common finding from completed studies is that the environmental impacts of interventions, when compared to an appropriate counterfactual, are often weaker than might have been expected or would be inferred from a simple comparison (e.g. with adjacent areas or a nearby non-treatment group). This effect is due to nonrandom factors that influence where environmental interventions are implemented, or biases in which groups sign up to participate in programmes. Protected areas, for example, may be located in more remote areas away from roads and population centres, or at higher elevations, or on soil types that are marginal for agriculture (Andam et al., 2008; Pfaff et al., 2009; Joppa & Pfaff, 2010), all of which would be expected to have lower deforestation rates regardless of the protected area intervention itself.

Application of impact evaluation methods to complex problems, such as interventions to address drivers of biodiversity and ecosystem loss, is limited by several factors, including:

- *Interventions have multiple goals*. Focusing on single outcome measures may distort decision-making by policymakers, which is particularly problematic for complex environment and development policies that seek to achieve multiple goals from the perspectives of different stakeholders (Adams et al., 2003), such as improvements in human wellbeing, governance and institutions, in addition to environmental sustainability. Very few published examples of impact evaluations of development programmes attempt to measure more than one outcome (Barrett et al., 2011).

- *Environmental goals have multiple dimensions and can be measured using a variety of metrics*. Protected Areas may be effective at securing wildlife habitat (Joppa & Pfaff 2011; Ferraro et al., 2012), but may fail to arrest continued decline in wildlife populations in some cases (Craigie et al., 2010), leading to concerns about the spread of 'empty forests' (Redford, 1992; Corlett, 2007; Wilkie et al., 2011).
- *Human wellbeing has multiple dimensions*. Most evaluation studies use relatively narrow definitions of economic poverty; mainly income and consumption (Ravallion, 2003; Vira & Kontoleon, 2010). Decades of research has shown that poverty is a multi-faceted concept incorporating social, political, cultural, institutional and environmental dimensions (Scoones, 1998; Sen, 1999; McGregor, 2007; Coulthard et al., 2011), collectively here called wellbeing, which can be measured in several aspects: incidence, intensity, inequality, temporality and spatiality (Agrawal & Redford, 2006). For environmental interventions, measuring how the multiple dimensions of wellbeing are experienced at the level of individual is particularly appropriate: (i) because of the need to capture non-economic values, and (ii) because each individual makes their own decisions regarding how to trade-off economic gain versus other dimensions of wellbeing. The success of an intervention therefore depends on impacts on individual wellbeing and how local people perceive impacts on their wellbeing, which may, or may not, be similar to results measured using externally conceived metrics.
- *Large-scale studies may miss important local heterogeneities.* Impact evaluation studies have become relatively popular in environmental science, since the publication of Ferraro and Pattanyak's (2006) original paper. The majority of studies completed to date are desk-based analyses, generally using already available large-*n* low-resolution datasets, such as large-scale analyses of changes in deforestation, to evaluate programme impacts (e.g. see Andam et al., 2008; Andam et al., 2010). These low-resolution national-scale evaluations may miss important local heterogeneities: programmes may be more effective in some places, or when outcomes are measured in more detail, than suggested by low-resolution national-scale evaluations. For example, several studies have evaluated the environmental impact of the Costa Rican PES programme using large-scale remote sensing datasets, concluding that it had minimal impact on deforestation rates (Sánchez-Azofeifa et al., 2007; Pfaff et al., 2008; Robalino et al., 2008). By contrast, Arriagada et al. (2012) combined fieldwork and analysis of remote-sensing images to re-evaluate the Costa Rican PES programme at

the scale of individual farms in one region of the country. They found that unlike the national-scale analyses, the Costa Rican PES programme had delivered significant environmental outcomes.

- *Moving beyond averages*. There is a need to disaggregate outcomes in order to understand the impacts on different subsets of society, as even policies that have overall positive impacts may still incur disproportionate costs for some people (Daw et al., 2011). Impacts must also be understood in the context of a dynamic system – livelihoods are complex and changing. Even well designed conservation programmes that have been specifically designed to have minimal impacts on current livelihoods may constrain future livelihood options in an increasingly market-linked rural economy.
- *There is a need for mixed research techniques*. Qualitative methods may be just as important as quantitative methods in impact evaluation. Some outcomes may be difficult or inappropriate to quantify, lending themselves to qualitative analyses – such as changes in governance or power relationships (Chambers, 2009; Garbarino & Holland, 2009; Drury et al. 2011). Impact evaluations are undertaken in the context of complex systems, with many confounding factors and co-occurring processes of change that may be unrelated to the intervention being assessed. In these cases an in-depth understanding of the system is required before valid, testable, hypotheses that are amenable to quantitative techniques can be developed, or in order to determine sampling regimes for counterfactual analyses (Drury et al., 2011). The tradeoff in accuracy between qualitative (internal) validity and quantitative (external, statistically representative) validity necessitates a careful consideration of methods before evaluations take place. The need for a more nuanced approach to impact evaluation embracing qualitative and quantitative methods is recognized by international guidance (DFID, 2009; NONIE, 2009), but is rarely applied in practice.

Finally, impact evaluation experts (e.g. Ravallion, 2006) emphasise that methodologies will only give robust results if potential sources of bias are correctly accounted for in the survey design. This requires a strong prior understanding of the system (Ravallion, 2006). Poor survey designs might identify an effect when in fact none exists or may mask effects. This highlights the importance of understanding the dynamics of the socio-ecological system when designing an impact evaluation, and suggests that impact evaluations conducted remotely or post-hoc based on opportunistic data are much less likely to deliver accurate results.

1.4 Understanding underlying processes

An ‘effective’ biodiversity conservation intervention would be based upon a strong conceptual understanding of the underlying drivers of biodiversity and ecosystem loss, in order to design an appropriate response (Salafsky et al., 2002; Margoluis et al., 2009). Deforestation and exploitation of natural resources, including biodiversity, are driven by a complex set of processes (Robinson & Bennett, 2000; Chomitz et al., 2007). Very broadly, these can be summarised at three different scales:

- the behaviour of individual actors (e.g. local people living within a Protected Areas), which is influenced by the prevailing economic incentives to clear land and harvest natural resources, the costs and benefits of changing livelihoods, and the ability of individuals to invest.
- the governance and institutional framework, including the types of property rights that exist and ability to enforce property rights, and the processes by which rules and regulations are determined and then enforced.
- external pressures, from migrants, or more powerful government actors or companies that seek to appropriate forest land and natural resources.

A good example of this type of analysis is the extensive research that has been undertaken over the past two decades concerning the agents and drivers of forest loss (Angelsen & Kaimotwiz, 1999; Geist & Lambin, 2002), which has led to the development of predictive models (Angelsen & Kaimotwiz, 1999) and more recently application of this knowledge to develop predications about how new policies such as REDD+ should be implemented (Angelsen, 2010).

Individual-level Behaviour

Rational choice theory assumes that individuals seek to maximise their utility, by maximising benefits and minimising costs (Coleman, 1973). The theory also assumes that individuals have perfect information about their choices and are not constrained in their ability to make decisions. Models of complete rationality have been highly successful in predicting marginal behaviour in competitive situations (Ostrom, 1998). In environmental science, one of the most influential applications of rational choice theory was Hardin's (1968) 'Tragedy of the Commons'. Hardin argued that the 'rational' user of a commons (or, more properly, an open-access resource) makes demands on the resource until the expected private benefits of his or her actions equal the expected private costs. In his model, because each user ignores the externalities their behaviour imposes on others, individual decisions culminate in the potential destruction of the open-access commons. The precise outcome in a particular situation depends upon the payoffs. In evolutionary theory, the one-play prisoner's dilemma game, which rewards both players if they cooperate, but rewards defection substantially more, is the equivalent to a tragedy of the commons, for which defection is the best strategy. The problem with this interpretation of human behaviour is that open-access commons are not always over-harvested and individuals do not always defect (Ostrom, 1990; Ostrom, 1999).

It is now well understood that cooperation is a common human behaviour, particularly if individuals repeatedly interact so reciprocation can be expected. Much of the original evidence stems from game theory, such as the success with which the simple ‘tit-for-tat ‘strategy (cooperate first, then do what the other player did in the last round) negotiates an iterated prisoners' dilemma (Axelrod & Hamilton, 1981). In real-life experiments, reciprocity – a pre-disposition to cooperate and to punish defectors – is a basic element of human behaviour. A well-known example is Berg’s trust game (Berg et al., 1995) in which players are usually pre-disposed to trust other players, even if they do not know who the other players are. In public goods experiments participants are also pre-disposed to make initial positive contributions (Ostrom, 2000). Even when repeated interactions are impossible, strong reciprocity has been observed, whereby individuals are pre-disposed to cooperate or punish defectors where these actions lead to no direct personal gain and reputation-building is not possible (Fehr & Fischbacher 2002; Fehr & Gächter 2000 & 2002; Gintis 2000). Evidence from neurobiology suggests that individuals receive subjective rewards from altruistically choosing to cooperate or punish defectors (Rilling et al., 2002; de Quervain et al., 2004). Results are robust even when experiments are undertaken in widely different cultures, although with much more variance in responses consistent with cultural norms (Henrich et al., 2001; Henrich et al., 2005; Henrich et al., 2006; Travers et al., 2011). Altruistic rewards and punishment imply that individuals have proximate motives beyond their economic self-interest – their subjective evaluations of economic payoffs differ from the immediate economic payoffs on offer; humans are not *Homo economicus* (Persky, 1995). Such social preferences arise when individuals are concerned about externalities which affect the welfare of others (Coleman, 1990), and are highly dependent on the social or institutional conditions in which people operate (Biel & Thørgersen, 2007). The most well-known and researched of these is the use of common-pool resources (Wade, 1988; Ostrom, 1990; Baland & Platteau, 1996).

Psychologists and sociologists have constructed theories of human decision-making that incorporate both economic and social preferences and that have been validated by experiments in real-world situations (St John et al., 2010). The Theory of Planned Behaviour (Ajzen, 1985; Ajzen, 1991) examines an individual’s behavioural intention, which then determines actual behaviour. Intention depends upon a combination of attitude, subjective norm and perceived behavioural control. Attitude is assessed by determining the consequences of the behaviour and the extent to which these consequences matter to the individual (including the economic payoffs). There is a considerable literature on what values may influence attitudes or beliefs (which then affect behaviours) and how they can be measured (Dietz et al., 2005), however there is no particular consensus and methods can be difficult to translate into different settings (Browne-Nuñez & Jonker, 2008). The subjective norm is the extent to which the behaviour fits with the prevailing social norms combined with the extent to which the individual wishes to meet these expectations (Fishbein & Ajzen, 1980). 'Social norms' is a general term for the shared understanding about what actions are obligatory, acceptable or forbidden (Ostrom, 2000). Perceived behavioural control is a product of factors that help or hinder a decision and the importance each has for the individual. Thus, according to the Theory of Planned Behaviour, whether you do something or not depends upon how good an idea you think it is, how much you care about what others think of the consequences and the extent to which you believe you are able to perform the behaviour.

Conservation interventions can seek to influence human behaviour by changing individual payoffs; for example by increasing the costs of the behaviour (e.g. through developing and enforcing rules such as protected areas), reducing its benefits (e.g. by targeting markets; Damania et al., 2005), or making alternative behaviours more attractive (e.g. by reforming property rights or implementing PES). Other interventions may seek to influence social norms (e.g. through attempting to change institutions) or perceived behavioural control (e.g. through gun confiscations). The basic model of rule enforcement states that compliance is a decreasing function of two factors: the probability of an act of non-compliance being detected and punished and the severity of punishment that results, which together influence the economic motivation to offend (Becker, 1968). Social preferences are important also, however, because individuals may perceive social costs to sanctions in addition to the monetary ones (Kaplow, 1990), or refuse to accept rules that are seen as illegitimate (Hønneland, 1999; Sutinen & Kuperan, 1999), or unfair (Fehr & Falk, 2002). The importance individuals attach to fairness is also relevant to other aspects of conservation interventions, such as the distribution of benefits from PES (unfair payments may be less effective at motivating a change in behaviour), or participation in decision-making during the development and implementation of local natural resource management rules (Falk et al., 2001; Fehr & Falk, 2002; Fehr & Rockenbach 2003; Fehr & Gintis, 2007). Brown and Corbera (2003) have developed a framework for considering equity in PES programmes which is to some extent based upon these underlying social preferences by evaluating equity in access, equity in decision-making and equity in benefits.

Another well-established effect that is relevant for the design of conservation interventions is the importance of intrinsic motivation in decision-making (DeCaro & Stokes, 2008; Deci et al., 1999; Deci

& Ryan, 1985 & 2004). External rewards, such as payments from a PES programme, can either complement or counteract intrinsic motivation, depending on how the rewards are structured and perceived (Deci et al., 1999). Intrinsic motivation might include a desire to work, or achieve social status, or improve your community. This might suggest that rules that are designed in close consultation with local people, and which are managed by local people, might be more effective at changing behaviour.

Finally, social preferences such as altruism, reciprocity, intrinsic motivation and a desire to uphold ethical norms have also been shown to interact with conventional economic incentives, such as fines or payments. Evidence indicates that depending on the situation economic incentives may either complement or substitute, either crowding in or crowding out social preferences (Deci et al., 1999; Cardenas et al., 2000; Fehr & Falk 2002; Bowles, 2008; Bowles & Hwang, 2008). Field experiments have shown that payments might reduce landowners' private conservation incentives, and thus weaken their overall instincts to conserve (Cardenas et al., 2000). Similar results – that starting to provide incentives undermine might undermine existing social preferences – have also been documented in laboratory experiments (Deci et al., 1999; Heyman & Ariely, 2004).

Governance and Institutions

Institutions are defined by North (1990) as: "the rules of the game in a society or, more formally, ... the humanly derived constraints that shape human interaction". Organisations are groupings of individuals that operate within the institutional framework. This framework includes property rights, monitoring, enforcement, governance and contracting arrangements. Economic historians (North, 1990 & 1994) have argued that differences in societies' institutional frameworks are probably a major reason for differences in economic growth and human welfare. In many developing countries land ownership and resource tenure are unclear, with land and resources technically owned and managed by the state (Agrawal et al., 2008); natural resources have high rents thereby attracting resource grabs and corruption; powerful individuals can often act with impunity; and government agencies have poor capacity and may receive little political support. These conditions are known to lead to high rates of habitat destruction and over-exploitation of natural resources (Chomitz et al., 2007; Geist & Lambin, 2003). Institutional failure is a central challenge for biodiversity conservation (Barrett et al., 2001) and one that is not easily rectified.

A considerable body of literature exists on the conditions that might lead to the evolution of institutions for governance of common pool resources (Baland & Platteau, 1996; Agrawal 2001; Ostrom, 1990, 1999 & 2003). This research suggests that institutions are more likely to evolve and be sustained for resources where property rights, particularly the ability to exclude others, are clear and enforced (Schlager & Ostrom, 1992; Robinson et al., 2011); users are highly dependent on the resource for their livelihoods; groups are small and cohesive; and the area is sufficiently small that users can develop accurate knowledge of the resource and monitor it relatively cheaply. The marginal value of forest land and forest resources has to be sufficiently valuable to the livelihoods of local people so as to provide an incentive to develop and enforce local rules to manage and conserve forests. Self-organisation is more likely if social norms already exist that promote cooperation, such as low discount rates, users trust each other, and individuals have the ability to participate in decision-making. The latter is thought to have been particularly important in the case of community-based wildlife management (Child & Dalal-Clayton, 2004). Agrawal (2001) provides a summary of 35 key factors that promote the establishment of institutions, based on a synthesis of 15 years of research and three comprehensive literature reviews (Baland & Platteau 1996; Ostrom, 1990; Wade, 1988). Many of these 35 factors are consistent with the results of the various experiments and games performed by economists and psychologists presented above – e.g. the importance of repeated interactions (communication, participation, etc.) and mechanisms for punishment.

Ostrom identified that for any society to move from over-harvesting of an open-access resource to a collective management system requires navigating three social dilemmas. The first is the incentive for individuals to over-harvest, as identified by Hardin. The second is the investment needed to establish a system for sustainable management (e.g. a local institution to manage land-use), which is itself a second-order public good dilemma (with incentives to free-ride, or defect, etc…). Individual decisions to then invest in monitoring and sanctioning activities in order to increase the likelihood that participants follow the agreements they have made also generates a public good, and these investments therefore represent a third-level dilemma (Ostrom, 1990 & 1999). The difficulty in navigating these three dilemmas perhaps accounts for why institutional failure is so common.

External pressures

Although often maligned, local people are a relatively weak actor in comparison with the external pressures that drive the loss of biodiversity and ecosystem services, such as the activities of companies, corruption and international trade. These external drivers interact with the livelihood strategies of local people and influence the effectiveness of local institutions. Conservation interventions must therefore be designed based upon an understanding of external drivers and incorporate measures to mitigate these drivers. The activities of companies, acquiring large areas of forest land for logging concessions or clearance for agri-industrial crops, is one of the most important drivers of deforestation and forest degradation (Geist & Lambin, 2002; Chomitz et al., 2007; Rudel et al., 2009; DeFries et al., 2010). Immigration into forest frontier regions is another important external driver of deforestation (Geist & Lambin, 2002) and biodiversity loss (Scholte, 2003). Widespread corruption may allow powerful elites to act with impunity, disregarding national laws, enabling them to expropriate land and natural resources (Chomitz et al., 2007). Corruption is also associated with higher rates of forest loss (Wright et al., 2007) and species declines (Smith et al., 2003). Bushmeat is an important component of diet and culture in many parts of the world (Milner-Gulland et al., 2003), but over-hunting is causing some species to become locally extirpated (Robinson & Bennett, 2000; Nooren & Claridge, 2001). The pressure in Southeast Asia is particularly high (Corlett, 2007). In addition to household consumption, hunting is driven by trade to local, national and international markets (Blundell & Mascia, 2005; Chaber et al., 2012), which is increasing with changes in urban wealth (Robinson & Bennett, 2002). The international trade in wildlife is now worth billions of dollars a year (Blundell & Mascia, 2005), and a critical driver is demand for products for Asian traditional medicine (Corlett, 2007; Graham-Rowe, 2011). International markets are also becoming the key driver of both agricultural expansion (Rudel et al., 2009; DeFries et al., 2010) and logging (Barney & Canby, 2012).

1.5 Thesis Overview

This thesis investigates the implementation of two common conservation interventions – Protected Areas and Payments for Environmental Services – in the Northern Plains of Cambodia, considering the two main research themes: understanding impacts and underlying processes. Figure 1.1 provides the basic conceptual framework for the thesis, based upon the principal research questions and the specifics of the study site in northern Cambodia, with numbers in bold referring to individual chapters. The principal research questions are:

<u>Theme (1) Impacts:</u> what are the conservation outcomes attributable to a particular implementation of a policy tool, in a specific environmental and social context? And what are the positive and

negative impacts of the intervention, both socially and environmentally, and from whose perspective?

- What are the appropriate comparisons to make when assessing impacts?
- What are the additional environmental outcomes that can be attributed to implementation of Protected Areas and Payments for Environmental Services, with a focus on forest and wildlife conservation?
- What are the impacts of Protected Areas and Payments for Environmental Services on local human wellbeing, considering multiple aspects of human wellbeing and disaggregating outcomes according to different groups of people?
- Is the effectiveness of Payments for Environmental Services at delivering additional conservation outcomes related to the conditionality of the payments?

Theme (2) Processes: how does policy implementation influence the dynamics of complex socio-ecological systems in order to bring about observed changes which can be attributed to that policy?

- What are the drivers of deforestation and the decline in species populations in the study site?
- How do Protected Areas and Payments for Environmental Services affect these drivers, focusing on both internal and external drivers and the proximate and underlying causes of change?
- Do Protected Areas and Payments for Environmental Services cause changes in the behaviour of local people?
- To what extent to considerations of fairness and equity influence the response of local people to Payments for Environmental Services?
- Does Payments for Environmental Services promote collective action, through providing an additional incentive for villages to establish local institutions for collective management of natural resources?

Chapter 2 introduces the study site in northern Cambodia and describes the Protected Area and the three Payments for Environmental Services programme that were implemented there. It also outlines the framework for the impact evaluation, including selection of the controls and the different outcome measures used to assess the impact of the interventions (human wellbeing, deforestation rates and wildlife populations).

Chapters 3, 4, 5 and 6 analyse the impacts of the programmes, considering both social (Chapters 3 and 6) and environmental (Chapters 4 and 5) outcomes. **Chapter 3** analyses the livelihoods of local villages in 2008, three years after the Protected Area intervention started and corresponding to the year that the Payments for Environmental Services programmes were scaled-up, providing a baseline. **Chapter 6** returns to this baseline to understand whether Protected Areas and Payments for Environmental Services have exacerbated or alleviated local poverty, and for which subsets of the local population. **Chapter 4** analyses the conservation impact of the first payment programme – direct payments for protection of globally threatened nesting bird species from harvesting – considering whether the programme did succeed in changing human behaviour and thereby causing increases in bird populations. **Chapter 5** turns to the second conservation target for the programme – reducing deforestation rates by local communities – and assesses the extent to which Protected Areas and Payments for Environmental Services did succeed in protecting forests.

Chapters 4, 5 and 7 also consider how the Protected Areas and Payments for Environmental Services programmes influenced the dynamics of the system in order to bring out the observed changes in outcomes. **Chapter 4,** on the longest-running of the payment programmes (for bird nest protection), investigates how payments were distributed and perceived by local people, and evaluates whether local concerns about the fairness or otherwise of payments might have undermined the programme. **Chapter 5** focuses on the drivers of deforestation and how the Protected Area and Payment for Environmental Services interventions influenced these drivers in order to bring about the observed impacts. Finally, **Chapter 7** investigates how the different payment programmes influenced human behaviour, and considers the extent to which behavioural changes were caused by the payments alone or other factors, including socioeconomic status of the recipients, perceptions of the fairness of the programmes and the activities of local institutions. **Chapter 8** concludes.

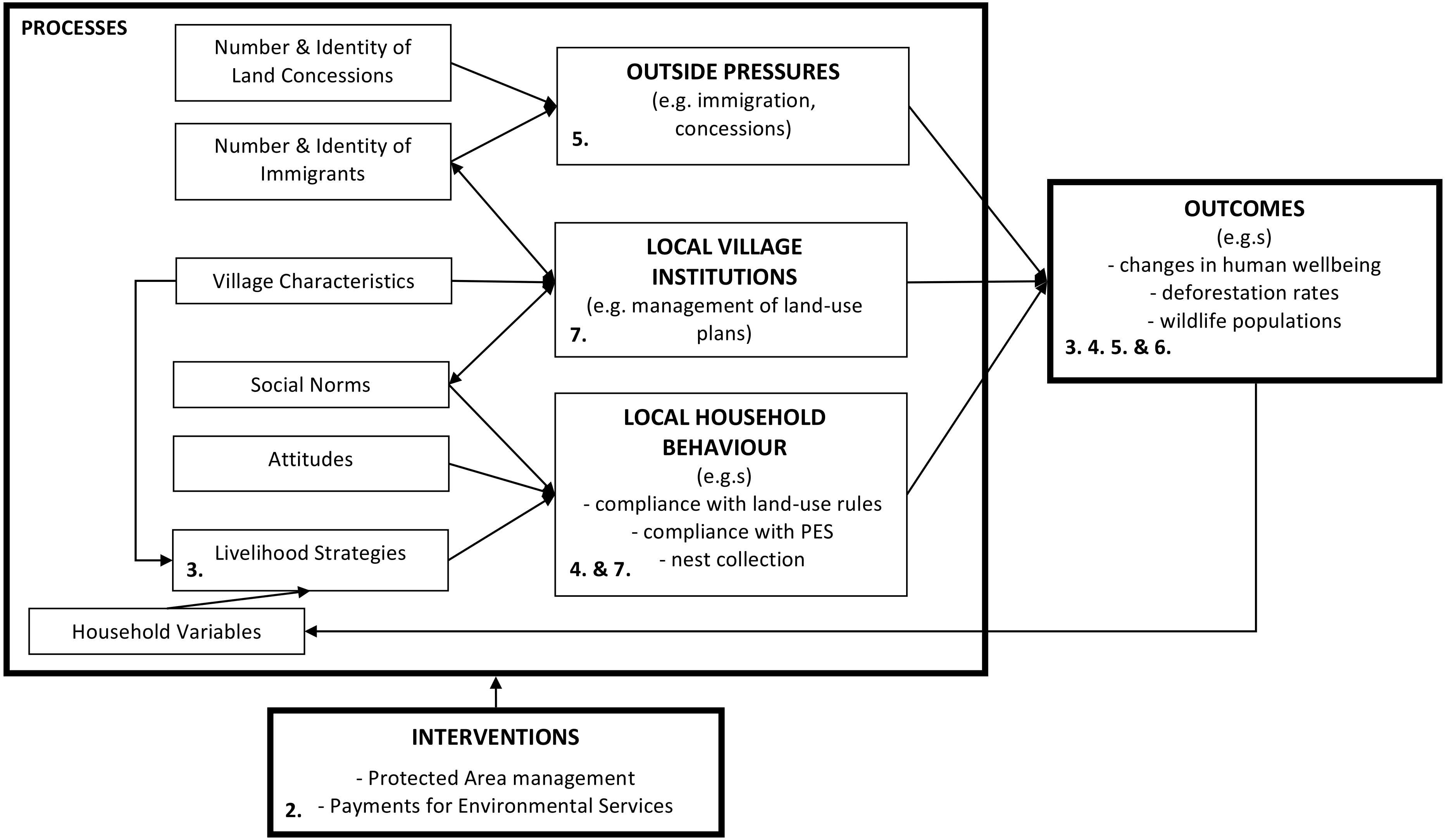


Figure 1.1. Conceptual Framework for the thesis. Chapter numbers are shown in bold.

Chapter 2. Study Area, Conservation Interventions, and Design of the Impact Evaluation[4]

2.1 Study Area: Northern Plains of Cambodia

Cambodia and the Northern Plains

Cambodia lies within the Indo-Burma hotspot (Myers et al., 2000) and contains four of the Global 200 Ecoregions (Olson & Dinerstein, 1998). The country is of global conservation importance because it contains the largest remaining examples of habitats that previously spread across much of Indochina and Thailand, which still contain nearly intact species assemblages, albeit at heavily reduced densities (Loucks et al., 2009). These include the deciduous dipterocarp forests once supported the greatest aggregation of large mammals and waterbirds outside the African savannahs (Wharton, 1966), of which the Northern and Eastern Pains of Cambodia are the largest remaining areas. Many of these species are listed on the IUCN (International Conservation Union) Red List (WCS, 2009), including 45 mammals (7 Critically Endangered or Endangered), 46 birds (12 Critically Endangered or Endangered, including the Giant *Pseudibis gigantea* and White-shouldered Ibises, *P. davisonii*) and 17 reptiles (9 Critically Endangered or Endangered). Conservation strategies are therefore frequently focused on remnant populations of highly threatened species where there is little room for error.

Hunting, habitat destruction and human disturbance, by both residents and immigrants, are the major and urgent threats to biodiversity conservation. Species populations have declined rapidly since the 1970s (Loucks et al., 2009). Annual collection by local people of eggs and chicks from nests is a particular threat to breeding populations of large bird species, particularly colonial nesting waterbirds and vultures (Goes, 2005; Clements et al., 2007; Clements et al., 2012b). Collection is generally undertaken opportunistically, often during trips to collect forest resources. Another major recorded source of mortality for wildlife species is poisoning, mainly relating to the misuse of poisons (possibly organophospates normally used in agriculture) for hunting or fishing (Clements et al., 2012a). Poisons are either emptied into waterholes (*trapeangs*), or delivered in bait such as rice, fish or fruit to catch scavenging species (such as storks) and frugivores. Reported national annual

[4] The description of the Study Area and the three PES programmes is taken from: Clements et al. (2010) *Ecological Economics*, 69, 1283-1291. The description of the impact evaluation design is taken from: (1) Clements et al. (2012c) *World Development*, in press; and (2) Clements et al. (2012b) *Biological Conservation*, in press.

deforestation rates were 0.7% during 1973-1997 (DFW, 1998) and 0.8% during 2000-2005 (Forestry Administration, 2008) and 2005-2010 (Forestry Administration, 2011), despite the fact that since 2002 most forest clearance has been illegal under the Forestry Law. Global assessments estimate that deforestation rates are significantly higher (1.3% during 2000-2010; FAO 2011). Based on these statistics Cambodia has one of the highest rates of land-use change globally. Deforestation is driven by a variety of processes including large-scale development projects, such as agro-industrial concessions, improved road access, population growth and smallholder encroachment, both by landless in-migrants and established communities (Cambodia R-PP, 2011). Encroachment is attractive to local people because land is an easily available secure form of wealth, which is viewed as an open access resource and enforcement of laws is rare. Many plots are claimed but not cleared, forcing new farmers to move further into the forest (An, 2008).

Initial conservation strategies in Cambodia focused on Protected Area (PA) management. The PAs were established in two phases. The boundaries of the Nature Protected Area system were decided in the early 1990s and gazetted by Royal Decree in 1993, and include Cambodia's National Parks, Wildlife Sanctuaries, Protected Landscapes (for cultural heritage areas) and various Multiple-Use Areas. All of these PAs are managed by the Ministry of Environment. Later, from 2002 onwards, various Protected Forests have been declared by Prime Ministerial Sub-decree to be managed by the Forestry Administration of the Ministry of Agriculture, Forestry and Fisheries. Due to on-going conflicts in the 1990s, very little was known about many of these areas (especially with respect to the Nature Protected Area system) at the time when PAs were declared. Boundaries were therefore decided based upon habitat types, historical records and some fieldwork in the case of the Protected Forests. In general PAs are located in remote forested areas, where road access is poor and local poverty is higher than the national average (World Bank, 2009).

Regardless of the date of designation, all Cambodian PAs have very poorly paid staff with limited capacity or infrastructure, i.e. they are 'paper parks' (Wilkie, et al., 2001). Ministry of Environment managed PAs have dedicated staff as part of the General Department for Administration of Nature Conservation and Protection; these are normally a park director and a few local rangers on annual contracts. Forestry Administration managed PAs have no dedicated staff and are instead administered through the Forestry Administration's line offices at Provincial and District level. PAs usually contain existing human settlements since the location of villages was not known when the PA boundaries were drawn. The Cambodian PAs are therefore very different from strictly enforced PAs, but share many characteristics with the estimated 56-85% of PAs in developing countries that also contain local people (Bruner et al., 2001; Brockington & Igoe 2006). These villages were not resettled, and have unclear property rights based upon national laws. Under Cambodian Law, local uses of natural resources within PAs are legal, although forest clearance, commercial logging, and hunting or trade of threatened species are illegal. Initially, large-scale concessions could not be legally declared within PAs, a restriction that was relaxed for Nature Protected Areas under the Ministry of Environment by the 2008 Protected Areas Law, but not for Protected Forests under the Forestry Administration. Since PAs were declared based on relatively little information the PA network excludes many areas of importance for biodiversity conservation, again not an uncommon situation (Brooks, et al., 2004), emphasising the importance of working both inside and outside PAs to achieve biodiversity conservation.

Protected Areas in the Northern Plains of Cambodia

This study focused on the core management zones of two PAs in the Northern Plains of Cambodia (Figure 2.1): 1,811km^2 of the 4,025km^2 Kulen Promtep Wildlife Sanctuary (KPWS) and 1,776km^2 of the 1,900km^2 Preah Vihear Protected Forest (PVPF). KPWS was declared in 1993 as part of the Nature Protected Area network managed by the Ministry of Environment, and PVPF in 2002 as a Protected Forest managed by the Forestry Administration of the Ministry of Agriculture, Forestry and Fisheries. The core management zones corresponded to the areas of greatest importance for biodiversity conservation and were entirely within Preah Vihear province. Sixteen villages were located inside the core management zones of KPWS and PVPF (Figure 2.1), all of which had existed since at least the 1960s, although there was considerable disruption in the 1970-90s, due to the civil war and forced resettlement by the Khmer Rouge. Resettled people subsequently returned to their original villages from the 1990s onwards. Although the villages are permanent and sedentary, they are often not socially homogenous and may contain a mixture of displaced peoples, ex Khmer Rouge and demobilized soldiers, indigenous peoples and more recent immigrants. Consequently, the villages are best defined as groups of people who happen to live in the same place, rather than 'communities' (Agrawal & Gibson, 1999). Local people are primarily subsistence farmers, practicing either rain-fed paddy rice cultivation or shifting cultivation, and are dependent upon forest resources as a crucial safety net and for cash income (McKenney & Prom, 2002; McKenney et al., 2004). One of the most important sources of cash income is the sale of liquid resins from dipterocarp trees, which comprises 16-23% of household income, with resin-tapping households earning $100-$340/year (Evans et al., 2002; McKenney et al., 2004).

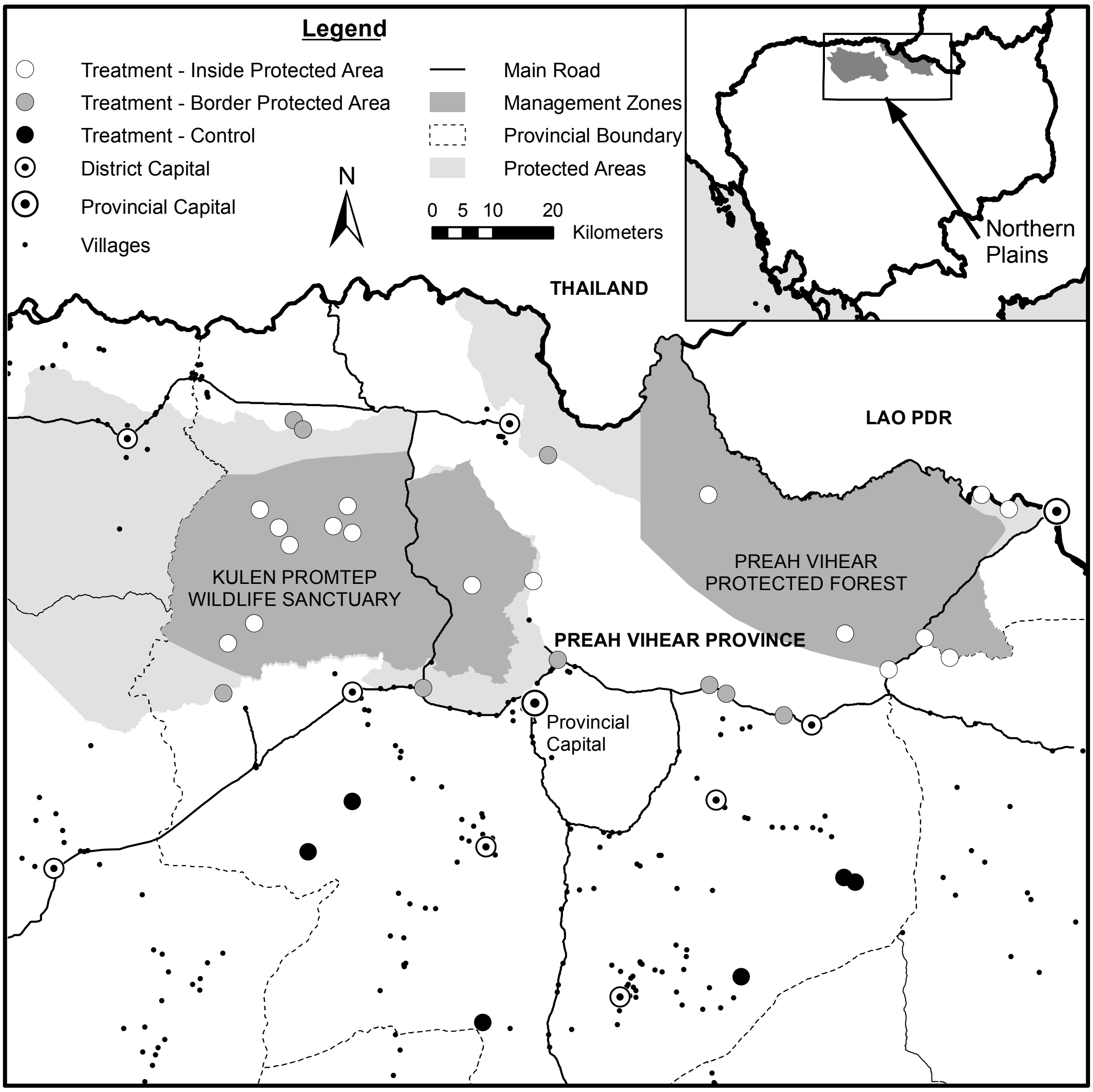


Figure 2.1. Map of Preah Vihear province, Cambodia, showing the two Protected Areas (Kulen Promtep Wildlife Sanctuary and Preah Vihear Protected Forest) and their Management Zones, and the location of the three types of villages selected for study – those inside the Protected Areas, those bordering the Protected Areas and the Controls.

Gazettment of the PAs conserved those sites to some extent from development pressures (such as forestry concessions in the 1990s). However both PAs remained essentially paper parks until the start of a long-term PA management capacity-building programme in 2005 titled "Establishing Conservation Areas through Landscape Management (CALM) in the Northern Plains of Cambodia", which was implemented by Government agencies and the Wildlife Conservation Society (WCS), and funded by the Global Environment Facility (GEF) through the United Nations Development Program (UNDP), with co-financing from WCS, the Critical Ecosystem Partnership Fund[5], the Danish International Cooperation Agency (Danida), the UK Department for International Development (DfID), New Zealand's International Aid and Development Agency (NZAID) and UNDP. As a consequence of the PA management capacity-building programme, during 2005-2012 both PAs had funding of around $2-3/hectare for the core management zones, which is broadly comparable to PAs in developing countries (Bruner et al., 2004). PA authorities were charged with enforcement of Cambodian Law, under which local uses of natural resources are legal, although land clearance, cutting of timber for sale, and hunting or trade of threatened species are illegal. Villages were permitted by PA authorities to expand agriculture to a limited extent within agreed land-use plan boundaries. The land-use plans were developed through a participatory process over a period of two or three years, which established forest management zones and clarified ownership over land and natural resources (Rock, 2001). The land-use plan is approved by the relevant Government authorities and is managed by an elected village committee of nine people. It specifically sets out which areas can be used for agriculture and residential land, including expansion areas that are currently forest.

2.2 The Three Payments for Environmental Services (PES) Programmes

Three PES programmes were designed to complement PA management in the Northern Plains (Clements et al. 2010). These programmes were: a community-managed ecotourism programme that linked income to bird and habitat protection; providing premium prices for agricultural goods if households limited field expansion to within agreed land-use plans (Ibis Rice); and direct payments conditional upon protection of nests of globally threatened birds. The first two programmes were managed at the village level, and were based upon the institutional foundation provided by the land-use plans and the village committees.

[5] The Critical Ecosystems Partnership Fund is a joint initiative of l'Agence Française de Développement, Conservation International, the Global Environment Facility, the Government of Japan, the MacArthur Foundation and the World Bank. A fundamental goal is to ensure civil society is engaged in biodiversity conservation.

Community-based Ecotourism

The community-based ecotourism programme was initiated in 2005 in the village of Tmatboey in Kulen Promtep Wildlife Sanctuary, and was replicated in Dangphlat village in Preah Vihear Protected Forest from 2008, and in Prey Veng village in Kulen Promtep Wildlife Sanctuary from 2010. The description here focuses on Tmatboey, although the programme operates in a similar manner at the other village sites. Tmatboey is a small village of 236 families, located in a large mosaic of deciduous dipterocarp forest, seasonally flooded grasslands and wetlands. The total village area is about 25,780 hectares, of which only a small proportion (620 hectares) is currently used for agriculture. The site fulfils many of the criteria for a successful ecotourism location (Wilkie & Carpenter, 1999): it contains rare species that are high profile targets for international birdwatchers (e.g. the Giant Ibis); sightings are reliable year-round; accommodation standards have improved as village capacity has increased; prices are moderately inexpensive and access is relatively easy from the major tourism centre at Siem Reap, which receives more than 2 million visitors annually to visit the temples at Angkor and has an international airport. The ecotourism programme aims to conserve the globally threatened wildlife through establishing local village-level tourism enterprises that directly link revenue received to long-term species conservation (Figure 2.2). This link is provided by the agreement between the PA authorities, WCS and the village, which stipulates that tourism revenue is subject to the villagers stopping hunting of key species and abiding by the land-use plan. This is reinforced by fees that are paid by all visitors: $30 per person if all key species are seen and $15 if only a subset are. A detailed description is given in Clements et al. (2008).

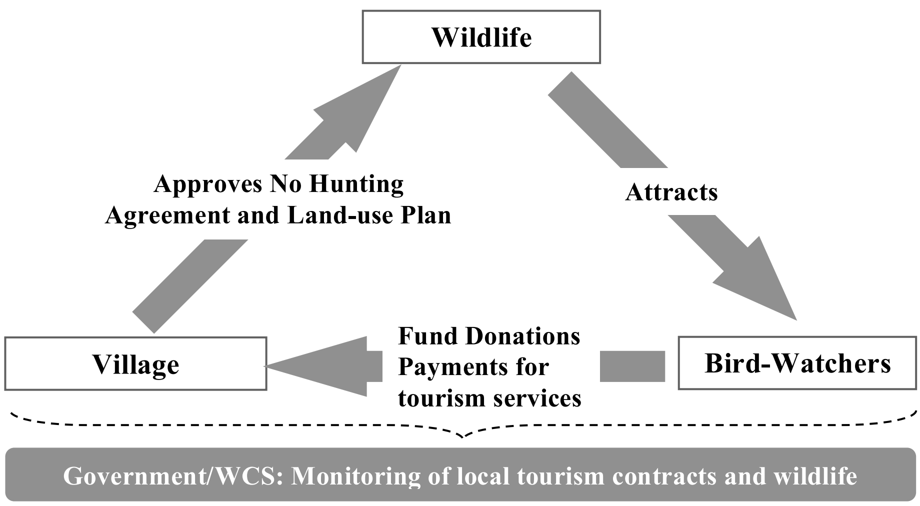


Figure 2.2. Design of the Community-based Ecotourism programme.

Institutionally, the programme relies on four parties, each of whom plays a key role:

- Elected village committees: site management of tourism services, management of income received and fund disbursements, local enforcement following no-hunting agreements and land-use plans, report serious violations to PA authorities;
- Protected Area authorities: legally approve tourism agreements and local land rights, law enforcement;
- Sam Veasna Centre: a local civil society partner based in Siem Reap that is responsible for marketing, site promotions, tourism bookings management and monitoring on behalf of the village-level enterprises;
- Private Sector: tourist bookings provide revenue.

WCS plays a general support role to all parties, and monitors the agreements.

Agri-Environment payments: Wildlife-Friendly products or 'Ibis Rice'

Tourism has limited potential for replication because all villages support a similar species mix, and the international birdwatching market is of restricted size. The agri-environmental payment programme 'Ibis Rice' was therefore initiated in 2008 in Tmatboey and Dangphlat, the first two tourism villages, as an alternative community-based payment programme that could be replicated widely. The programme as expanded to Prey Veng (the third tourism village) and a fourth village in Preah Vihear Protected Forest, Narong, during 2008-2011. Under the programme, farmers that keep to the land-use plan and no-hunting rules are allowed to sell their rice through the village committee responsible for management of the land-use plan to a marketing association called Sansom Mlup Prey (Figure 2.3).

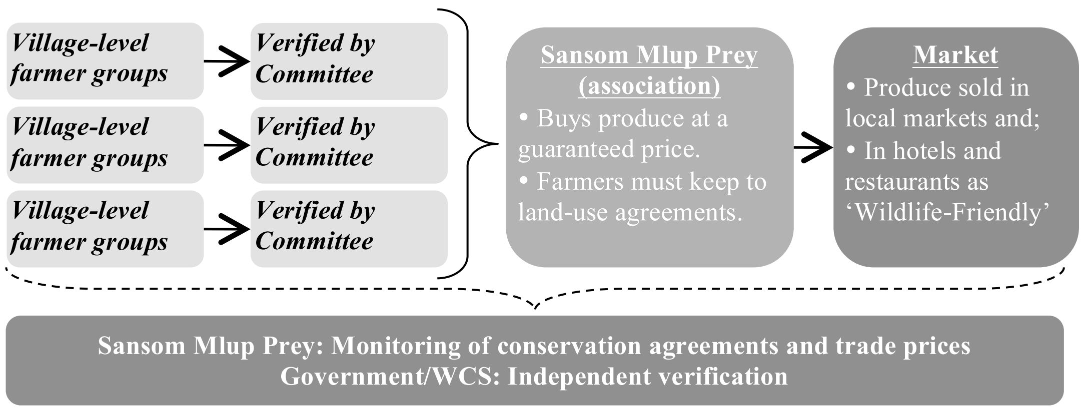


Figure 2.3. Design of the Ibis Rice programme.

Sansom Mlup Prey offers preferential prices to the farmers, who are supported by directly selling the rice to national market centres, bypassing middlemen who previously monopolised village trade, and through selling to tourist hotels under the 'Wildlife-Friendly' certification system, a new global brand. Sansom Mlup Prey also provides start-up capital and training in new agricultural techniques. In addition, the village scales are used to weigh produce, rather than the middleman's, which is seen as fairer and more transparent, since the middleman's scales are generally perceived to be biased. All profits are shared between the farmers and the village organisations, after deducting the costs of Sansom Mlup Prey. For participating farmers, the village committee determines their eligibility to receive payments by monitoring compliance with the village land-use plans. These results are then externally verified by Sansom Mlup Prey and WCS. Local self-enforcement is based on verbal or written contracts between farmers and the committee to stop illegal activities or relocate ricefields within land-use plans, rather than strong punishments. The payment value was set based on the market premium available for the products, not based on assessment of the opportunity costs to farmers of further encroachment. For farmers with sufficient labour or access to machinery these opportunity costs are likely to be high, since land is very valuable and alternative forms of employment are limited. The committee also receives a share of the profits, which provides added motivation (and income) for their work.

Direct Contracts for Bird Nest Protection

The globally threatened large birds found in the Northern Plains of Cambodia are heavily threatened by human disturbance and particularly collection of nests for eggs and chicks by local people for local consumption or trade. In order to address this specific threat, the Bird Nest Protection programme was initiated in 2003 by WCS in collaboration with the Ministry of Environment and the Forestry Administration of the Ministry of Agriculture, Forestry and Fisheries. The programme was designed to rapidly locate, monitor and protect the remaining nesting sites as a complement to longer-term activities to strengthen institutions for environmental protection, such as protected areas. Under the programme, local people are offered a reward of up to US$5 for reporting nests, and are then employed to monitor and protect the birds until the chicks successfully fledge. Prior to 2008 protectors received a payment of $1 per day for their work and an extra $1 per day upon completion if chicks successfully fledged. The total payment of $2/day was judged an acceptable daily wage based on village consultations. Since 2008 payments have been increased to $2.50/day total due to rising food prices based upon requests from local nest protectors. The protection teams

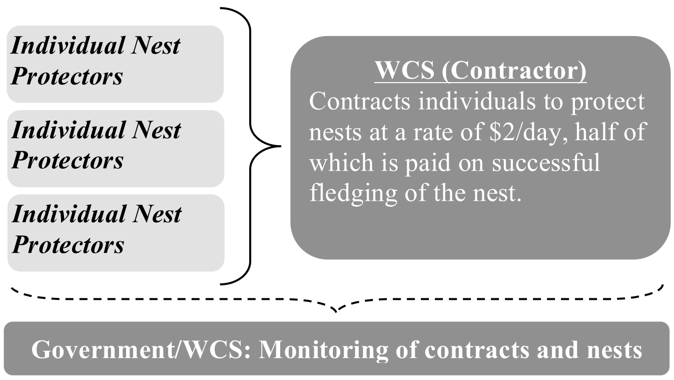


Figure 2.4. Design of Bird Nest Payments programme.

are regularly visited every 1-2 weeks by village rangers employed by WCS and by WCS monitoring staff to check on the status of the nests and for the purposes of research and data collection. The programme operates year-round, as some species nest in the dry season and others during the wet season. It started in four pilot villages in 2003 in Kulen Promtep Wildlife Sanctuary and was extended to Preah Vihear Protected Forest in 2004. By 2009 the programme was operating in 24 villages across both protected areas, including the villages targeted by the Ecotourism and Ibis Rice programmes. Unlike the previous two examples the bird nest protection programme works entirely through individual contracts; it is not community-based (Figure 2.4).

2.3 Design of the Impact Evaluation

Introduction: Impact evaluation methodologies and reducing bias

A fundamental goal of this thesis was to understand the impacts of the different conservation interventions – both protected areas and the three PES programmes – on key outcome variables such as conservation outcomes (deforestation rates, species populations), livelihoods (poverty and other aspects of human wellbeing) and drivers of ecosystem loss (immigration, land clearing behaviours). Understanding impacts requires moving beyond simplistic analyses based upon correlations between an intervention and changes in an outcome variable, in order to be able to attribute impacts. Rigorous impact evaluation survey designs can be used to untangle the impacts of forest conservation policies from the wider dynamics of the system, by assessing the degree to which changes in outcome variables can be attributed to policy interventions as opposed to other factors (Ferraro, 2009). Typically this is achieved by evaluating outcomes in comparison with the counterfactual – what would have happened in the absence of the intervention. Standard approaches use randomized control trials with policy interventions assigned randomly to intervention and control sites in order to eliminate other sources of bias. In econometrics the difference between the intervention and the control groups is called the *average treatment effect*.

However placement of forest conservation interventions, such as protected areas (Joppa & Pfaff, 2010), is usually non-random with respect to other areas in the landscape. In these cases, quasi-experimental survey designs such as matching can be used to control for other sources of bias by ensuring that intervention and control areas or groups are comparable in all aspects except that the controls have not received the intervention (Rosenbaum & Rubin 1983; Ferraro & Pattanayak, 2006; Ravallion, 2006; Pattanayak, 2009). Matching minimizes differences in observed covariates between intervention and control groups. The comparison between the intervention and control groups is therefore equivalent to what would have happened to the intervention groups had they not been subject to the intervention. In econometrics this is called the *average treatment effect on the treated*.

For matching to be applied properly, the underlying dynamics of the system being investigated must be well understood in order to select appropriate covariates (Ravallion, 2006). The matching covariates should not be affected by the interventions (i.e. they are exogenous to the system that is being studied). A critical assumption is that for the matching variables, potential outcomes are independent of assignment to control or intervention groups, called conditional independence. A second critical assumption is to ensure covariate balance (i.e. covariates have similar values for controls and intervention groups) and to ensure that the area of common support (covariate values found in both control and interventions groups) is large. A key problem is that matching can only be done on observables (i.e. observed covariates), leaving open the possibility of sources of unobserved bias. If an unobserved covariate does affect outcomes but was not controlled for by the matching process, then a difference between the intervention and control groups would be found when in fact the intervention had no effect.

Another method that can be used to reduce unobserved sources of bias is to follow trends in time for intervention and control groups, and to use this data to calculate the difference-in-difference estimator (also called the before-after-control-intervention survey design). The difference-in-difference estimator controls for time-invariant unobservable characteristics by using data from the same treatment units over time (Wooldridge, 2002). A key assumption is that the expected trend in the outcome variable for the control group is equal to the expected trend for the intervention group, in the absence of the conservation intervention. Combining the difference-in-difference method with matching, by using matching to select the control groups, can ensure that this assumption is met. Smith and Todd (2005) found that the difference-in-difference matching estimator performs best among other matching estimators, and Imbens and Woolridge (2009) recommend combining methods in this way. At least one other recent study (Arriagada et al. 2012) combined both matching and difference-in-difference estimators to evaluate the environmental impact of PES programmes. The same approach was used in this study, to calculate the impact of protected area and PES programmes in the Northern Plains of Cambodia.

A second methodological issue considered in the survey design was the scale at which interventions were implemented and outcome variables could be measured. Impacts were deliberately investigated at the local scale, in order to avoid the potential biases associated with large-scale studies (Arriagada et al. 2012). The primary unit that was affected by both the protected area and PES programmes was the village, because this was the scale at which local people experienced the conservation interventions and responded to them. It was assumed that the different villages could be considered as independent replicates, because local people were generally sedentary farmers that used forest resources within a day's walk (5-8 km) from their village. Within each village, different households would respond to the conservation interventions in different ways depending upon their livelihood strategies and attitudes and the extent to which they benefited from or were restricted by the conservation interventions. Consequently, a nested survey design was used: sampling independent replicates at the village-level, and then sampling different types of households within each village. The village-level sampling is considered here, and the household-level is addressed in the individual chapters.

Methods: use of matching to select the control villages

Village selection was only undertaken within Preah Vihear province within the Northern Plains, because different provinces are subject to different political and socio-economic factors. In Preah Vihear, for example, conflicts continued up until the death of Pol Pot in 1998 and Preah Vihear consequently received much less external investment (by Government, development agencies and companies) than other adjacent provinces whilst conflicts were on-going. In total, 6 villages were located within Preah Vihear Protected Forest and 10 villages were located within Kulen Promtep Wildlife Sanctuary, excluding villages around provincial and district towns. These included one village (Prey Veng) of more than 75 households, which was not recognised as an independent official

Cambodian village by the Ministry of Interior, instead being considered part of a the nearby village of Sambour, which is also within Kulen Promtep Wildlife Sanctuary. Prey Veng was counted as a separate village for the purposes of the analysis because it is an old village site (being present on the 1960s maps), has its own village chief, and is requesting official status from the Ministry of Interior.

The matching dataset compromised the 208 official villages recorded by the Commune Database (NCDD, 2011), plus 3 unofficial villages of >50 households (including Prey Veng), a total of 211 villages. Matching procedures are generally more accurate if there are a large number of candidate controls to select from relative to the number of intervention units. The purpose of the matching was to ensure that the selected control villages were as similar as possible to the villages inside the protected areas for observed baseline characteristics in 2005 (prior to the initiation of the interventions). The matching variables used therefore correspond to the main factors thought to have influenced protected area placement (see Study Area description), and the main determinants of poverty status at the village level (World Bank, 2009). The variables chosen were: (*1*) Number of families in the village in 2005 from the Commune Database and updated by field surveys; (*2*) Distance to nearest all-weather road in 2005, from the Cambodia Reconnaissance Survey Digital Data (MPWT/JICA, 2003) and updated by field surveys; (*3*) Distance to nearest full-day market in 2005 based on field surveys; and (*4*) Percentage of forest cover within 5 km of the village based on the national forest cover assessments from 2005/2006 (Forestry Administration, 2008). The 2005/2006 assessment has an overall classification accuracy of 74%, mainly referring to classification errors between the main forest types (Evergreen Forest, Semi-Evergreen Forest, Deciduous Forest and Other Forest). For the purposes of the matching analysis, all these forest types were combined into a single variable, and expressed as the percentage of land forest cover within 5 km of the village within Cambodia. 5 km was chosen as the buffer area, because detailed mapping of agriculture around villages within the protected areas indicated that almost all fields are located within this distance of the village (unpublished data).

All the variables selected were exogenous to the interventions being evaluated, i.e. they are not likely to have been affected by the protected area or PES interventions over the timeframe of the analysis. The conservation interventions were not likely to have any impact on the distance to an all-weather road or full-day market (roads are, for example, routinely built through protected areas in Cambodia). The conservation interventions are hypothesised to have an impact on the percentage forest cover, however this effect would have been predominantly after 2005/2006 (the date of the imagery used for the analysis) as protected area management started in 2005. Other similar analyses (e.g. Andam et al. 2008) have included variables relating to soil type and geography (e.g. slope). These variables were not included in this analysis because Preah Vihear is basically flat, with the exception of 3-4 uninhabited hills, and the soils are relatively uniform in the study area.

Due to the small number of villages inside the protected areas two matches were found for village, to ensure that a sufficiently large number of potential matches were found across the landscape. Analyses were carried out in R 2.14.2 using the package 'matching' (Sekhon, 2007; R Development Core Team, 2012). Nearest-neighbour covariate matching was used (Abadie & Imbens, 2006), allowing matching against multiple variables with equal weighting. The Mahalanobis distance (Abadie & Imbens, 2006) was used to measure distance in the multivariate space, as in previous studies (Andam et al., 2008, Andam et al., 2010, Joppa & Pfaff, 2011). All matching was with replacement and ties were handled deterministically by weighting the tied matches (Abadie & Imbens, 2006).

Balancing tests were used to evaluate the results of matching estimators, by comparing the matching covariates for the intervention and matched control groups. Statistics calculated included the means for each group; the mean, median and maximum difference in the empirical quantile-quantile (eQQ) plot of intervention and control groups on the scale in which the variable was measured; the mean, median and maximum difference in the empirical cumulative distribution function (eCDF); the variance ratio of intervention over control groups (which should equal 1 if there is perfect balance); *t*-tests comparing the samples before and after matching (the two sample *t*-test was used pre-matching and the paired t-test was used post-matching); and the bootstrap Kolmogorov-Smirnov (KS) test, which tests for a significant difference across the entire distribution (as indicated by the eQQ plots).

Results: selected control villages

Matching selected 15 possible control villages, and balancing statistics and tests indicated that balance had been achieved in the matched sample (Table 2.1). Two of the villages were within 20 km of the protected areas and were excluded from the sample to prevent spill over effects from the protected areas onto the control villages. Random stratified sampling, by district, was then used to select up to 2 matches per district, all of which were 20-60 km from the boundaries of the protected areas. The random stratified sampling by district ensured that the final control villages selected were distributed across the landscape, rather than being clustered in one area, making the survey design robust in the future against the potential loss of control villages due to large-scale development or other events.

Most previous studies compare outcomes within protected areas or villages subject to PES programmes with adjacent areas. Consequently, a second comparison set of villages bordering the protected areas was incorporated into the study. Unlike the control villages, these villages were only 4-12 km from the protected area boundary and were generally located along main roads. The intervention villages, control villages and the villages bordering the protected areas are shown in Figure 2.1.

Conclusion: Impact Evaluation survey design

The impact evaluation methodology used in subsequent chapters is based upon the methods introduced here. Outcomes of the conservation programmes were assessed for three treatment groups: around the intervention villages within the Protected Areas (including those where the PES programmes were implemented), the control villages, and the villages bordering the Protected Areas. Where possible difference-in-difference estimators were calculated, measuring effects both before and after the conservation programmes were initiated, for the intervention, control and bordering protected area groups. Precise details of the methodologies used for data collection in each case are given in the individual chapters.

Table 2.1. Balancing statistics and tests for covariate matching for the unmatched and matched samples of villages inside and outside the protected areas. The matching process ensured that the differences between protected areas and control villages for the matched sample were not significant. Statistics calculated included the means for each group; the mean, median and maximum difference in the empirical quantile-quantile (eQQ) plot of treatment and control groups on the scale in which the variable was measured; the mean, median and maximum difference in the empirical cumulative distribution function (eCDF); the variance ratio of treatment over control (which should equal 1 if there is perfect balance); *t*-tests comparing the samples before and after matching (the two sample *t*-test was used pre-matching and the paired *t*-test was used post-matching); and the bootstrap Kolmogorov-Smirnov (KS) test, which tests for a significant difference across the entire distribution (as indicated by the eQQ plots).

Variable	Village Size (families)		Distance to all-weather Road (km)		Distance to full-day Market (km)		Forest Cover in 2005/6 (%)	
Statistic	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched
Mean PA villages	131.2	131.2	27.6	27.6	24.3	24.3	94.7	94.7
Mean Control villages	167.2	129.5	13.9	22.6	9.6	23.0	74.4	92.2
Std Deviation Mean diff	-65.9	3.0	61.0	22.1	271.9	23.6	657.0	81.4
Mean raw eQQ diff	90.0	17.8	15.5	8.0	14.3	1.3	20.8	2.5
Median raw eQQ diff	22.0	18.0	11.6	5.1	15.3	1.0	16.5	2.6
Max raw eQQ diff	963.0	38.0	31.6	25.0	18.7	5.3	50.5	5.9
Mean eCDF diff	0.078	0.095	0.259	0.109	0.431	0.068	0.433	0.160
Median eCDF diff	0.065	0.100	0.235	0.100	0.464	0.067	0.474	0.133
Max eCDF diff	0.222	0.233	0.540	0.267	0.764	0.200	0.801	0.333
Variance ratio (Treatment/Control)	0.182	2.13	1.179	2.189	0.502	1.045	0.0344	0.619
t-test *p*-value	0.041	0.857	0.036	0.114	< 0.001	0.055	< 0.001	0.102
KS Bootstrap *p*-value	0.408	0.310	<0.001	0.194	< 0.001	0.463	< 0.001	0.058

2.4 Environmental and Social Outcome measures

Both environmental and social criteria were used to assess the outcomes of the Protected Area and PES programmes, using the impact evaluation framework. Broadly, the criteria used were divided into:

- Environmental criteria, including:
    - Deforestation rates
    - Protection of Globally Threatened bird populations
- Social criteria, including:
    - Poverty
    - Agricultural productivity
    - Food security
    - Education

For the environmental criteria, data was collected from all of the selected controls and all the villages inside the PAs. For the social criteria, data was collected from only a subset of villages (11 within the PAs, 4 border villages and 5 controls), due to funding constraints. Figure 2.5 shows the villages used for the social surveys.

Precise details of the methods used to collect the data are described in Chapters 3-6. The methods used to conduct social surveys, assess household poverty and calculate deforestation rates were more complex and are described in detail here.

Social surveys of households and villages

Three social surveys were conducted as part of this thesis: (*1*) An initial assessment of livelihoods and poverty in 2008 for 871 households, four years after PA management started, for villages inside the PAs, the matched controls, and the villages on the border of the PAs (Chapter 3); (*2*) A follow-up assessment using the same methods in 2011 for 1053 households, including households that had received payments from the three PES programmes during the intervening years (Chapter 6); and (*3*) Surveys of the attitudes and behaviour of local people towards the different PES programmes in eight villages inside the PAs where the programmes were operational conducted in late 2009 and early 2010 (Chapters 4 and 7). The methods used to conduct these surveys are described here.

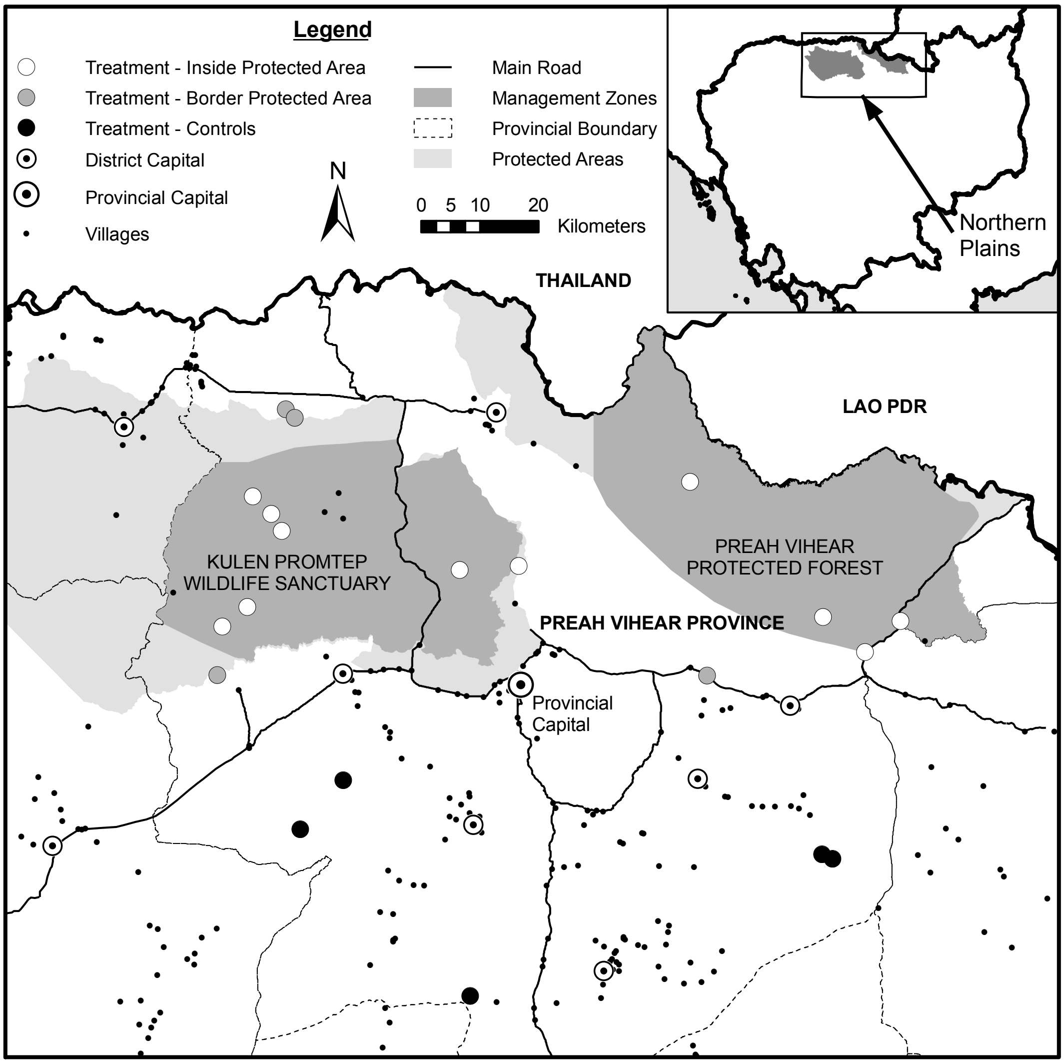


Figure 2.5. Map of Preah Vihear province, Cambodia, showing the two Protected Areas (Kulen Promtep Wildlife Sanctuary and Preah Vihear Protected Forest) and their Management Zones, and the location of the three types of villages used for the social surveys in Chapters 3 and 6 – those inside the Protected Areas, those bordering the Protected Areas and the Controls.

Any social assessment is challenging in Cambodia, because local people are accustomed to participatory rural appraisal exercises, which are generally conducted by organisations and donors prior to launching new programmes. Consequently they may vary their responses depending upon what information they think the interviewer wants to hear or the key conclusions they want the assessment to form. In designing the social surveys used in this thesis, a number of steps were taken to ensure that the data collected was as accurate as possible.

- The surveys were designed with the Centre for Development Orientated Research in Agriculture and Livelihood Systems (CENTDOR). CENTDOR originated as an internal monitoring and evaluation department within the Cambodian Centre for Development and Study in Agriculture (CEDAC), one of the largest Cambodian rural development NGOs, and was then established as a separate organisation in 2008. CENTDOR specialises in the monitoring and evaluation of rural development projects, with a focus on maintaining independence between the researchers and the programme that is being evaluated, and provides these services for a several organisations and donors in Cambodia. By using trained Cambodian social scientists as the primary researchers, CENTDOR aims to ensure that the data collected is of the highest quality possible. For each of the surveys, the interviewers identified themselves as independent Cambodian researchers. Where additional survey team members were needed, students were recruited from the Royal University of Phnom Penh and they were provided with a letter of introduction from their professors.

- The author's affiliation with WCS was well known in some villages in the study landscape. Consequently, the author did not accompany the survey team; instead pilot surveys were conducted in up to four other villages in the landscape that were similar to the study villages but where the author was not known. The purpose of the pilot surveys was to refine the proposed questionnaires and for the purposes of training the survey team to ensure consistency between interviewers. The author accompanied the survey team on each of the pilot surveys, and then conducted regular meetings with the team during the full survey to ensure that the work was proceeding appropriately. This included an initial review and analysis of the data received after the first and fourth villages were completed.

- Livelihood and poverty assessments focused on objective measures that could be verified visually (e.g. the type of house or owning a mini-tractor) rather than collecting data on incomes, which would be easy to falsify. Consumption data was also not collected, due to the problems with recall periods (Wilkie, 2007) and because obtaining reliable consumption data would necessitate multiple visits to the survey households in different seasons. Instead, the Basic Necessities Survey (BNS; Davies & Smith, 1998; Pro-Poor Centre & Davies, 2006) was used as a measure of household poverty. The BNS calculates an index of poverty for every household in the sample, relative to a locally-derived definition of poverty, and is based on previous methods that have been used in both developed and developing countries (Mack & Lansley, 1985; Hallerod, 1994; Noble et al., 2008, for further details see below).

- Data were triangulated where possible, by asking the same question in different ways, and avoiding asking any leading questions (for which the standard response in Cambodia is 'yes'). For example, during the attitude surveys, respondents were asked if payment programmes operated in their villages and were then asked to describe how they worked and what they thought about them. Subsequently, the interviewer then asked specific questions about each of the individual named PES programmes.

- For potentially sensitive issues, such as yields of resin or rice (which correlate closely with household income), questions were designed based on prior research that had assessed the accuracy of self-reports in similar villages (Evans et al., 2002; McKenney et al., 2004). This research had demonstrated that local people's recall of yields using local units (such as number of cans of resin collected per trip, or number of sacks of rice at the harvest) was a more accurate way to assess total yields. Since the local units can be different (the size of rice sack, for example, will vary between villages), this necessitated doing a calibration for each village.

Using the Basic Necessities Survey as a measure of Household Poverty

Poverty is a multi-faceted concept incorporating multiple dimensions (Scoones, 1998; Sen, 1999; McGregor, 2007), which can be measured in several aspects (Agrawal & Redford, 2006). An important methodological problem in social impact assessment therefore concerns how to define and measure poverty in order to assess trends (Ravallion, 2003). Detailed assessments of incomes were not possible, because these are time-consuming and it is questionable how accurately Cambodians report income data given the history of conflict and abuses of the past few decades. Instead, at the first assessment in 2008, data was collected using three measures of poverty: (*1*) the Basic Necessities Survey (BNS; Davies & Smith, 1998; Pro-Poor Centre & Davies, 2006); (*2*) a standard basket of assets, as a measure of absolute household wealth (Wilkie, 2007), and (*3*) a participatory wealth ranking exercise.

The Basic Necessities Survey incorporates multiple aspects of poverty into a single score for each household in the sample, relative to a locally-derived definition (Davies & Smith, 1998; Pro-Poor Centre & Davies, 2006). The Basic Necessities Survey is based on previous methods that have been used in both developed and developing countries (Mack & Lansley, 1985; Hallerod, 1994; Noble et al., 2008). Basic necessities are defined as assets or services that 50% or more of respondents agree "*are basic necessities that everyone in the community should be able to have and nobody should have to go without*". During the survey, respondents are asked to choose which items from a list meet the basic necessities definition, and they are then asked if they have the item currently. Items are weighted for importance according to the percentage of respondents who say an item is a basic necessity, discarding items that <50% of people thought met the basic necessity definition. Household poverty scores are based on the sum of the weightings of the basic necessities they have, as a percentage of the sum of the weightings for all basic necessities. The Basic Necessities Survey has the advantage that the population sampled defines the poverty score weightings, i.e. poverty is locally defined. Including a large variety of assets and services in the Basic Necessities Survey list means that the final score captures a wide range of the dimensions of poverty. The Basic Necessities Survey list of assets and services was defined during initial focus group discussions in villages not selected for the full surveys and district or provincial towns.

The list was constructed to contain a mixture of 35 items (assets or services) that included: (*i*) Items everyone in the study area agreed were basic necessities, and the majority of households had (e.g. a knife, having three meals a day); (*ii*) Items everyone in the study area agreed were basic necessities, but only some people had (e.g. draft animals, access to secondary schools, a toilet); (*iii*) Items <50% of people in the study area thought were basic necessities, but which were basic necessities to people in towns (e.g. mobile phones, electricity, a television); (*iv*) Items no one in the study area thought were basic necessities, and which people in towns did not rate as basic necessities, but which might become basic necessities in the future (e.g. health insurance, having a holiday). The list therefore deliberately included items no one would consider necessities, in order to encourage respondents to consider their answers rather than just marking all items as basic necessities. Including items that might become basic necessities as the aspirations change also allowed the same list to be used to measure future trends. Thirty-five items was thought to be an appropriate length for the list in order to keep the questionnaire short.

At the first assessment (in 2008 see Chapter 3), three measures of poverty status were calculated for each of the households in the sample: the Basic Necessities Survey score, the value of the basket of assets and the participatory wealth ranking. All three measures were highly correlated for the 871 households surveyed (Figure 2.6 and Table 2.2). The Basic Necessities Survey score was therefore used as the measure of household poverty in all the models for the 2008 (Chapter 3) and 2011 (Chapter 6) data.

Table 2.2. Non-parametric (Spearman's) correlation coefficients for the Basic Necessities Survey score, the Basket of Assets, and the Participatory Wealth ranking, for the 871 households and for models including the random effect of village only. All coefficients are significant at $P < 0.001$.

	Households only		Including random effect of village	
	Participatory Wealth Ranking	Value of the Basket of Assets	Participatory Wealth Ranking	Value of the Basket of Assets
Basic Necessities Survey score	0.531	0.628	0.694	0.744
Participatory Wealth Ranking		0.548	0.548	0.588

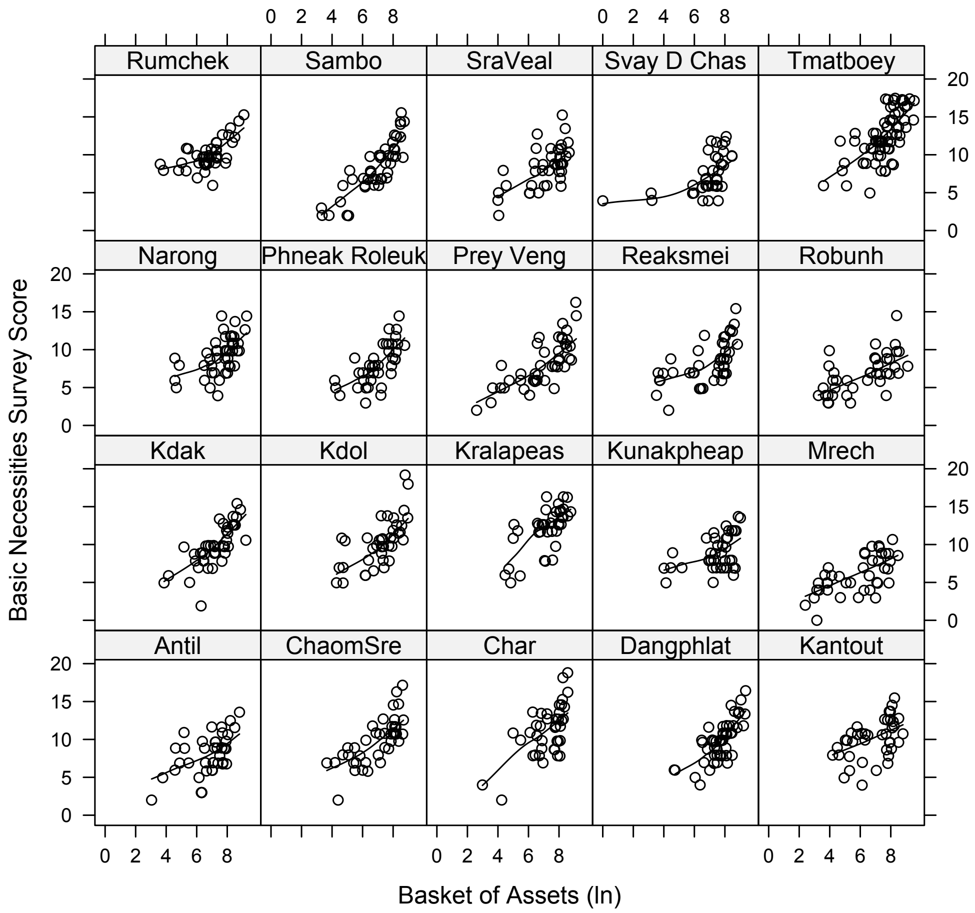


Figure 2.6. Scatter graph of household scores from the Basic Necessities Survey Score and the Basket of Assets for the 20 villages. There is a high level of correlation between household scores under the two techniques.

Calculating Deforestation Rates

Deforestation in the landscape was mapped between 2001/2 and 2005/6, the four years immediately prior to establishment of the PAs, and between 2005/6 and 2009/10, the subsequent four years when the PAs were actively managed and the PES programmes that focused on habitat protection (the Ecotourism and Ibis Rice) were being implemented. The analysis focused on areas around all 16 villages inside the PAs, 7 controls and 11 villages that bordered the PAs (Figure 2.1).

Landsat and Aster satellite images were used to estimate forest cover at three nominal time points: 2001/2, 2005/6, and 2009/10 (Rainey et al. 2010). Imagery from the Cambodian dry season (December-March) was used as this is likely to show greater contrast between forest and non-forest areas and lower cloud cover. Imagery covering the whole of the landscape in the 2002 dry season was not available, so for some sections imagery was used from a different sensor and from the 2001 and 2003 dry seasons (Table 2.3). For 2010, the best data were available from Landsat 7 (ETM+), which suffers from scratch lines due to a persistent sensor error. Gaps were filled with temporally adjacent images using standard techniques. The mosaic method was used to combine the various image fragments into one single image for the study area for each year. Two other types of satellite image were often available for each dry season, and these have been used as ancillary data the visual assessments.

Interpretation delineated two vegetation classes, forest and non-forest. The official national definition[6] of forest is an area of land covering at least 0.5 ha with at least 10% cover of trees taller than 5 m. This is a slightly lower percentage of crown cover than is used by the Forestry Administration in their assessments (20%; Brun 2009). Non-forest is all land with canopy forest cover less than this, and so includes natural grassland, bare land, water, swidden agriculture, rice paddy, other agricultural land, settlements and deforested areas. Mature tree plantations were not found in the landscape. The minimum mapping unit for forest was 1 ha, due to the limitations of medium-resolution imagery, since an area of 3x3 pixels (90 m x 90 m, 0.81 ha) was the smallest unit that could realistically be identified. This differs slightly from the 0.5 ha criterion used in national definitions but at a landscape level did not result in any bias in deforestation statistics. The image pre-processing and interpretation processing were done using the software package ERDAS IMAGINE. Subsequent data analysis was performed using ArcView 3.3 and ArcGIS 9.3. Images were geometrically corrected to an image from a reference year, which was in turn corrected to the rivers

[6] As submitted to the United Nations Framework Convention on Climate Change under the Kyoto Protocol's Clean Development Mechanism.

in the national hydrology dataset. A general accuracy of +/-1 pixel was achieved. The same methods gave overall accuracy of 93% when applied in Seima Protection Forest landscape in Mondulkiri which has a similar range of habitats (Evans et al. 2009).

Non-forest patches were identified visually and then delineated using a semi-automatic approach with the "Seed Tool" extension in ArcGIS 9.3. Data from the three sensors (Landsat TM, Landsat ETM+ and ASTER) were analysed in comparison to high quality ancillary datasets for the whole study area, such as ground-truthed datasets and aerial photographs (for sources see Table 2.4). For the baseline year, 2001/2, high resolution aerial photographs were available for the entire study area, which clearly showed areas of forest and non-forest. For years after 2002, the non-forest polygons of the previous time point were taken as the baseline and any observed changes were incorporated by editing and recapturing polygon boundaries.

The forest cover maps were resampled using a 1 km square grid, to give estimates of the number of hectares of deforestation in the two time periods in each of the 1 km grid squares. 1 km was judged an appropriate size for the grid given the resolution of the data (1 hectare). Only squares with complete forest cover maps were used.

Table 2.3. Master images and reference dates for polygon delineation.

Dry seasons	Reference year	Master image dates	Sensor (resolution)
2001/02	2002	10 Jan 2002 and 02 Feb 2002 with 17 Feb 2001,19 Jan 2002, 20 Feb 2002, 06 Jan 2003	Landsat 7 (30 m) ASTER (30 m)
2005/06	2006	21 Jan 2006 07 Feb 2006 23 Feb 2006	ASTER (30 m) Landsat 5 (30 m)
2009/10	2010	24 Dec 2009 with 09 Jan 2010* 05 Mar 2010 with 17 Feb 2010* 18 Feb 2010	Landsat 7 (30 m) Landsat 7 (30 m) Landsat 5 (30 m)

*The second image was used to fill sensor defects in the first image.

Table 2.4. Ancillary datasets, used for interpretation and validation, where compiled from a variety of sources.

Data	Sources	Additional Information
Aerial photographs	Ministry of Land Management, Urban Planning and Construction - Department of Geography	2001
Administration data (location of villages, towns, etc.)	Ministry of Land Management, Urban Planning and Construction - Department of Geography (updated by field surveys)	2005
Land use	Japan International Corporation Agency (JICA)	2000
Roads	Ministry of Public Works and Transportation (updated by field surveys)	2008
Forest cover	Forestry Administration	2002/2006
Agricultural parcel boundaries	Field surveys as part of participatory land use planning work in selected villages	various years
Rivers and wetlands	Ministry of Land Management, Urban Planning and Construction - Department of Geography	2005

Chapter 3. Baseline analysis of the impacts of Protected Areas on local livelihoods in Cambodia[7]

Abstract

Impact evaluation methods were used to investigate the effect of Protected Areas (PAs) on poverty and livelihoods in Cambodia, comparing households inside two PAs with border villages and controls in other remote forest areas. Both mixed effects models and matching methods were used to calculate effect sizes. There was no evidence that PAs exacerbated local poverty or reduced agricultural harvests in comparison with controls. Households bordering the PAs were significantly less poor, but this effect was due to greater access to markets and services. Non-timber forest product (NTFP) collectors inside PAs were significantly less poor than controls and had greater harvests. The PAs in Cambodia therefore have some positive impacts on households that use forest and land resources for their livelihoods.

3.1 Introduction

The impacts of protected areas on local poverty – both negative and potentially positive – have been widely debated (Adams & Hutton, 2007; Roe, 2008). Although the global benefits of biodiversity and ecosystem services are well recognized (Balmford et al., 2002; TEEB, 2010), the costs of protected areas (PAs) may be disproportionately borne by local people (Cernea & Schmidt-Soltau, 2006; West et al., 2006; Adams & Hutton, 2007). Debates have focused on whether the environmental goals of protected areas are compatible with poverty alleviation goals, especially in developing countries (Adams et al., 2004). There is now widespread acceptance that conservation policy should, at the very least, do no harm, and where possible should contribute to poverty alleviation (CBD, 2008). Accurate understanding in policy choices is limited by the paucity of information that exists regarding the impacts of current interventions on local poverty (Agrawal & Redford, 2006). For example, high poverty rates have been documented around PAs, but very few studies have attempted to quantify whether this is due to the PA or other factors (Andam et al., 2010; Sims, 2010; Naughton-Treves et al., 2011). The need to better understand the relationship between forest conservation policies and local poverty and the lack of information on impacts has led to repeated calls for the adoption of rigorous impact evaluation methods (Wilkie et al., 2006; Ferrao & Pattanayak, 2006; Pattanayak et al., 2010). Measuring impacts is also necessary during

[7] This manuscript is being published as: Clements (2012c) *World Development*, in press.

implementation to ensure that interventions do not negatively affect local people (Schreckenberg et al., 2010).

Whether PAs benefit or impose costs on local people depends upon the underlying relationship between local poverty and forest resource use (Angelsen & Wunder, 2003), external drivers, and the costs and benefits of the conservation interventions. The forest-poverty relationship is dynamic and may be different for different groups of people, implying that social impact assessment needs to consider who gains or loses, and when. Forest resources may contribute to local livelihoods through: (*1*) a poverty-driven forest reliance, whereby local poor people depend on low-value forest resources to some extent for their livelihoods, perhaps in response to shocks ('safety nets'), or (*2*) because they are unable to make the transition out of this resource dependent mode ('poverty traps'); and (*3*) an opportunity-driven forest reliance, whereby local people use higher-value forest resources as a source of a cash products in order to get richer ('pathways out of poverty', Angelsen & Wunder, 2003; Ruiz-Pérez et al., 2004). PA interventions can forcibly influence these relationships by either placing restrictions on forest resource use (Coad et al., 2008), displacing and resettling people (Brockington & Igoe, 2006), or increasing costs due to wildlife conflicts (Woodroffe et al., 2005). Alternatively, interventions may encourage and promote local forest resources use, for example through improved marketing or safeguarding access rights, exclusion of outsiders creating local monopolies, and may provide alternative pathways out of poverty through employment and business opportunities (Wunder, 2001; Scherl et al., 2004; Coad et al., 2008).

Rigorous impact evaluation survey designs can be used to untangle the impacts of forest conservation policies from the wider dynamics of the system, by assessing the degree to which changes in poverty can be attributed to policy interventions as opposed to other factors (Ferraro, 2009). Standard approaches use randomized control trials with policy interventions assigned randomly to intervention and control sites in order to eliminate other sources of bias. However placement of forest conservation interventions, such as PAs, is usually non-random (Joppa & Pfaff, 2010). In these cases, quasi-experimental survey designs such as matching can be used to control for other sources of bias by ensuring that intervention and control groups are comparable in all aspects except that the control groups have not received the intervention (Rosenbaum & Rubin, 1983; Ferraro & Pattanayak, 2006; Ravallion, 2006).

A second methodological problem in social impact assessment concerns how to define and measure poverty in order to assess trends (Ravallion, 2003). Poverty is a multi-faceted concept incorporating social, political, cultural, institutional and environmental dimensions (Scoones, 1998; Sen, 1999; McGregor, 2007), which can be measured in several aspects: incidence, intensity, inequality, temporality and spatiality (Agrawal & Redford, 2006). Standard approaches include household consumption and income surveys usually with multiple visits to the same households over the sampling period (Wilkie et al., 2006; Angelsen et al., 2011). These detailed methods can be expensive and time-consuming, and may neglect other non-economic dimensions of poverty. If measuring the impact of environment and development interventions is to become common practice there is a need to develop accurate and cost-effective methods that capture multiple dimensions of poverty and are appropriate for widespread use (Schreckenberg et al., 2010).

This paper uses matching and regression estimators to evaluate the impact of two PAs on the livelihoods of local people in Preah Vihear province, Cambodia. Both PAs contained established villages, and have been the focus of a long-term PA management and development programme since 2005 (Clements et al., 2010). The objective of this study was to investigate impacts due to the PAs since their establishment. It also established a baseline against which the subsequent implementation of three Payments for Environmental Services schemes, which were initiated in 2008, could be evaluated. The principle research questions addressed in this paper are: (*1*) what factors affect household poverty status and agricultural productivity; (2) what has been the overall impact of the PAs on local poverty and agricultural productivity in comparison with bordering villages and controls; and (*3*) have the PAs had different impacts on different types of livelihood strategies in comparison with controls.

3.2 Impact Evaluation Framework

Study Sites

The study focused on the core management zones of two PAs in northern Cambodia (see Chapter 2 for more details; Figure 2.5): Kulen Promtep Wildlife Sanctuary (gazetted 1993) and Preah Vihear Protected Forest (gazetted 2002). The PAs are in remote forest areas and contain 16 long-established villages. The impact evaluation took place in 2008, four years after the PA management activities were initiated.

Village and Household Matching Methods

Matching methods were used to select appropriate controls for households and villages inside the PAs against which to measure the impacts of PA management. A nested survey design was used, with two levels of matching: (*1*) selecting control villages that were similar to villages inside the PAs in order to measure overall impacts (see Chapter 2 for details); and (*2*) selecting control households within the control villages that had similar characteristics to households inside the PAs. The first dataset considered overall effects of PAs, whereas the second investigated specific impacts for different livelihood strategies. Prior to matching, a detailed qualitative analysis of the factors influencing the placement of PAs and household livelihood strategies was undertaken to ensure that the appropriate covariates were used to select controls (Ravallion, 2006).

For households, matches were chosen from a database of 504 households from 11 villages within the PAs, and 205 households from the 5 control villages selected by the village-level matching. The matching variables chosen were household characteristics that were not likely to be affected by PAs in the short-term, but were known to be strong determinants of household livelihood strategies and poverty status based on existing studies of rural Cambodians in forest areas, including in Preah Vihear (Evans et al., 2002; McKenny & Prom, 2002; McKenny et al., 2004; Hansen & Top, 2006; World Bank, 2009). The variables chosen were: (*1*) Adult Male Equivalents in the household, based on the recommended dietary calorie allowances for Southeast Asia (Barba & Cabrera, 2008), as an estimate of the daily nutrition needs of the household; (*2*) Number of working adults, defined as people ≥15 years old and <60 years; (*3*) Number of years of education that the household head has completed; (*4*) Whether or not the household taps resin, one of the most important sources of household income from the forest; and (*5*) Whether or not the household has at least one hectare of rice paddyfields. After matching, tests were also done to ensure that covariate balance was achieved for other key livelihood strategies: (*6*) Whether or not the household owns a shop, is engaged in service provision, or works as a trader; (*7*) Whether or not a member of the household is employed, either in the public, private or nongovernmental sectors; and (*8*) The number of livelihood strategies the household is engaged in, since households engaged in more livelihood strategies are likely to be richer and may have more resilience against shocks.

Only one match was found for each household within the protected areas because the pool of potential controls was much smaller. Matching was performed with a calliper, which defines a distance that is acceptable for any match. Any households within the protected areas that were outside the calliper were therefore dropped. The calliper was defined as 0.5 standard deviations of each matching covariate. Callipers reduce bias in the comparison, but at the cost of estimating differences on a subsample that may not be representative of all households in the villages inside the protected areas. In this case, this subsampling is justifiable because the intention was to investigate differences between households with the *same* livelihood strategies; not to draw conclusions about the entire population.

Analyses were carried out in R 2.13.0 using the package 'matching' (R Development Core Team, 2011). Nearest-neighbour covariate matching was used (Abadie & Imbens, 2006), allowing matching against multiple variables with equal weighting, which is appropriate when considering complex livelihoods that have multiple dimensions. The Mahalanobis distance (Abadie & Imbens, 2006) was used to measure distance in the multivariate space, as in previous studies (Andam et al., 2008; Andam et al., 2010; Joppa & Pfaff, 2011). All matching was with replacement and ties were handled deterministically by weighting the tied matches (Abadie & Imbens, 2006). Balancing tests were used to evaluate the results of matching estimators, by comparing the matching variables for the intervention and matched control groups.

Matched Datasets

Matching selected 15 possible control villages, and balancing statistics and tests indicated that balance had been achieved in the matched sample (see Chapter 2, Table 2.1). Random stratified sampling by district was then used to select five controls, with the controls distributed 20-60 km from the boundaries of the PAs (see Chapter 2, Figure 2.5). For households, the matched dataset contained 325 households within PAs (64% of 504 households), matched with 134 households from the control villages outside PAs (65% of 205 households). Households operate as discrete economic units in Cambodia, which is why households were selected as an appropriate sampling unit for the purposes of this analysis. Balancing statistics and tests indicated that balance had been achieved in the matched sample for all eight covariates (Table 3.1). The matched households were selected evenly across the villages inside the PAs and the control villages, with no particular bias towards any of the villages (Figure 3.1). As matching was done with replacement, some households outside the PAs were matched with several households inside PAs (Figure 3.2), with only two control households selected as matches ten times or more, suggesting a relatively limited effect. The calliper dropped 179 households within the PAs that did not have similar livelihood strategies to households outside the PAs, ensuring balance in the final matched sample.

Table 3.1. Balancing statistics and tests for covariate matching for the unmatched and matched samples of households inside and outside the protected areas. The matching process ensured that the differences between protected areas and control villages for the matched sample were not significant. Statistics calculated included the means for each group; the mean, median and maximum difference in the empirical quantile-quantile (eQQ) plot of treatment and control groups on the scale in which the variable was measured; the mean, median and maximum difference in the empirical cumulative distribution function (eCDF); the variance ratio of treatment over control (which should equal 1 if there is perfect balance); *t*-tests comparing the samples before and after matching (the two sample *t*-test was used pre-matching and the paired *t*-test was used post-matching); and the bootstrap Kolmogorov-Smirnov (KS) test, which tests for a significant difference across the entire distribution (as indicated by the eQQ plots).

Variable	Adult Male Equivalents		Number of Working Adults		Years of education of HH head (square-root transform)		Household has resin trees (1 = Yes, 0 = No)		Household has one hectare of rice paddy-fields (1 = Yes, 0 = No)	
Statistic	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched
Mean treatment	4.43	4.26	3.13	2.98	1.21	0.91	0.54	0.46	0.71	0.74
Mean Control	4.46	4.29	3.05	2.98	0.67	0.90	0.29	0.46	0.63	0.74
Std Deviation Mean Diff	-1.39	-2.07	6.12	0.00	51.01	1.32	48.68	0.00	17.72	0.00
Mean raw eQQ diff	0.114	0.080	0.146	0.000	0.532	0.041	0.244	0.000	0.078	0.000
Median raw eQQ diff	0.083	0.065	0.000	0.000	0.410	0.000	0.000	0.000	0.000	0.000
Max raw eQQ diff	0.850	0.537	1.000	0.000	1.732	0.488	1.000	0.000	1.000	0.000
Mean eCDF diff	0.016	0.015	0.014	0.000	0.082	0.013	0.122	0.000	0.040	0.000
Median eCDF diff	0.013	0.012	0.004	0.000	0.033	0.007	0.122	0.000	0.040	0.000
Max eCDF diff	0.046	0.052	0.066	0.000	0.254	0.049	0.243	0.000	0.080	0.000
Variance ratio (Treatment/Control)	0.952	0.983	1.028	1.000	1.244	1.030	1.198	1.000	0.877	1.000
t-test *p*-value	0.87	0.17	0.46	1.00	< 0.001	0.22	< 0.001	1.00	0.04	1.00
KS Bootstrap *p*-value	0.88	0.69	0.20	1.00	< 0.001	0.38	n/a	n/a	n/a	n/a

Table 3.1 (continued). Balancing statistics and tests for covariate matching for households.

Variable	Household runs a shop/service/trader (1 = Yes, 0 = No)		Household is employed (1 = Yes, 0 = No)		Number of household livelihood strategies	
Statistic	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched
Mean PA villages	0.15	0.14	0.07	0.06	2.03	1.92
Mean Control villages	0.14	0.14	0.03	0.03	1.72	1.91
Std Deviation Mean Diff	2.89	0.89	12.65	12.09	36.63	0.93
Mean raw eQQ diff	0.010	0.012	0.029	0.029	0.307	0.046
Median raw eQQ diff	0.000	0.000	0.000	0.000	0.000	0.000
Max raw eQQ diff	1.000	1.000	1.000	1.000	1.000	1.000
Mean eCDF diff	0.005	0.006	0.016	0.014	0.052	0.008
Median eCDF diff	0.005	0.006	0.016	0.014	0.023	0.007
Max eCDF diff	0.010	0.012	0.031	0.029	0.164	0.021
Variance ratio (Treatment/Control)	1.059	1.019	1.850	1.943	1.165	0.921
t-test *p*-value	0.72	0.90	0.06	0.07	< 0.001	0.87
KS Bootstrap *p*-value	n/a	n/a	n/a	n/a	< 0.001	0.84

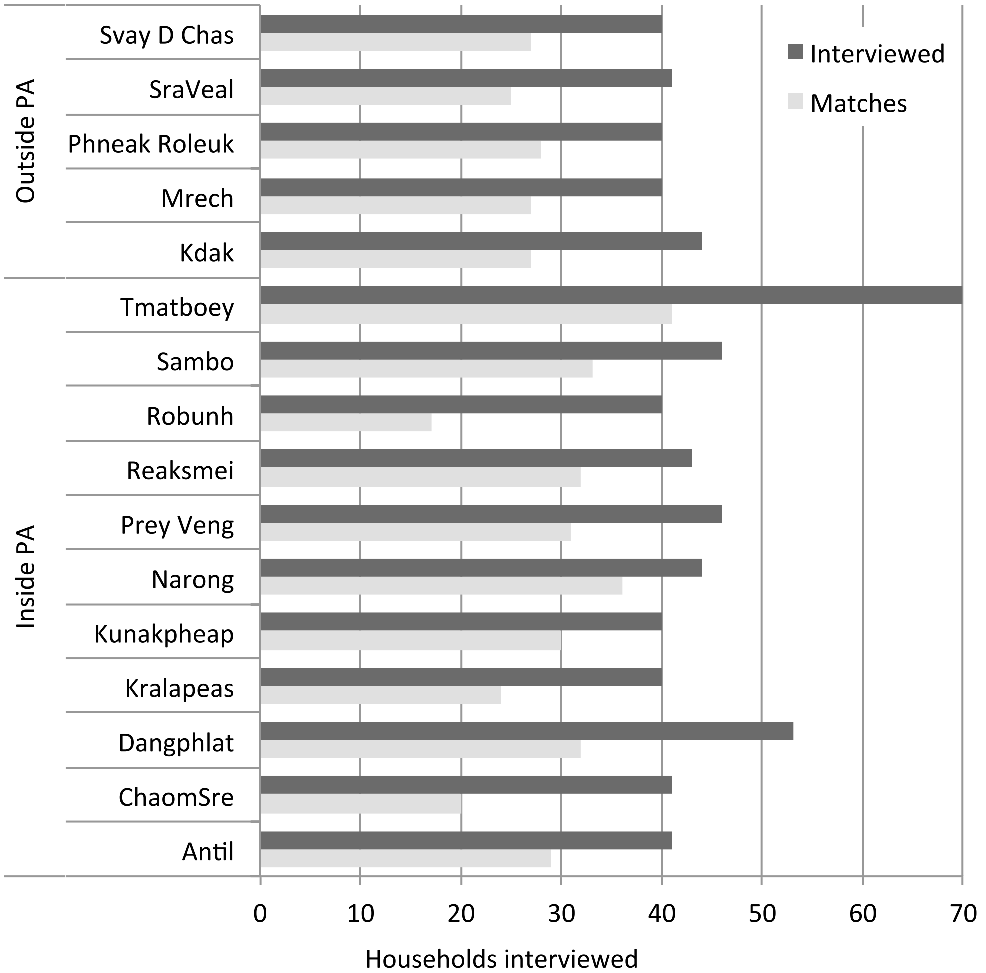


Figure 3.1. Number of households interviewed and number of households selected by the matching with replacement in the PA and control villages.

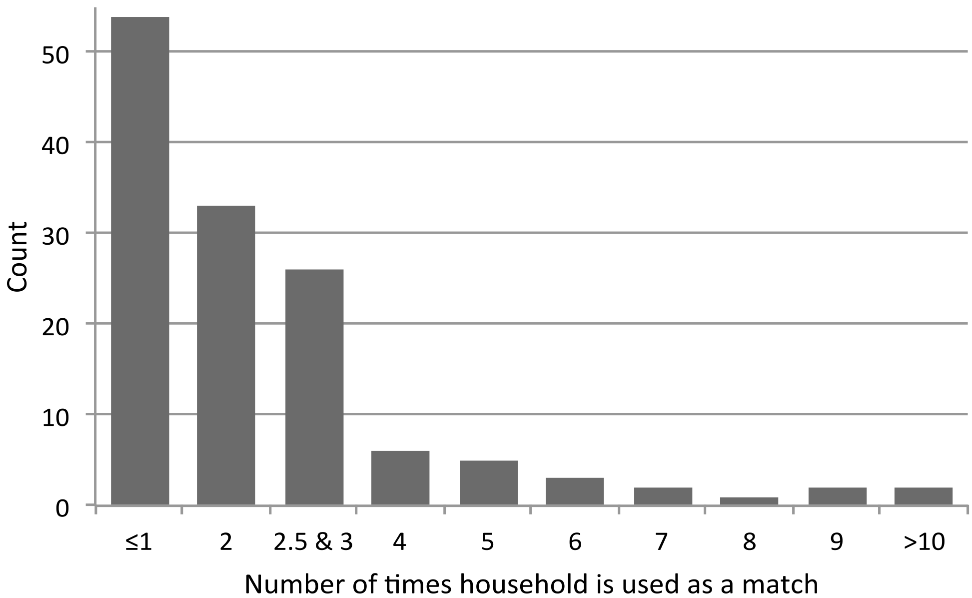


Figure 3.2. Histogram of the number of times a control household was selected as a match (total number of households selected = 134, partial matches = 0.5).

3.3 Survey Methods

Three survey methods were used: (*1*) household surveys of livelihood strategies and poverty status; (*2*) surveys of village-level characteristics; and (*3*) informal qualitative discussions around livelihoods and poverty and drivers of change. In total 871 households were sample from 20 villages; 504 from 11 villages inside the two PAs (selected randomly from the 16 inside the PAs), 205 from the 5 matched controls 20-60 km outside the PAs, and 162 from 4 villages 4-12 km from the border of the PA management zones (see Chapter 2, Figure 2.5). Surveys were conducted by trained social researchers, primarily from the Centre for Development Oriented Research in Agriculture and Livelihood Systems (CENTDOR). Prior to the initiation of data collection, pilot surveys were undertaken in the other four villages inside the PAs for training purposes and to evaluate the survey methods. Full surveys then took place during September-November 2008. Interviews were conducted with 40-45 households in each village, and additional households inside the PAs that had expressed interest in or were engaged in the Payments for Environmental Services programmes that was being established at the same time. Survey households were selected at random.

Household and village surveys of livelihood strategies and poverty status

A standard household questionnaire was developed during the pilot surveys, which collected data on key household characteristics, livelihood strategies, and the Basic Necessities Survey (BNS; Davies & Smith, 1998; Pro-Poor Centre & Davies, 2006) as a measure of household poverty. The questionnaire was deliberately kept short, taking 40-60 minutes to complete, by collecting salient information only. Respondents were household heads or another adult household member if the household head was unavailable. The BNS calculates a relative index of poverty for every household in the sample, relative to a locally-derived definition of poverty, and is based on previous methods that have been used in both developed and developing countries (Mack & Lansley, 1985; Hallerod, 1994; Noble et al., 2008, for further details see Chapter 2).

Each household was asked which livelihood strategies they engaged in, based on a list compiled during pilot surveys and allowing for free responses. Data on rice harvests (the staple food in Cambodian diet) and yields of liquid resin were collected using standard local units (e.g. sacks of rice, cans of resin collected per trip), which previous work had suggested encouraged accurate responses. The basket of assets list was developed based on the pilot surveys, and the value of the assets was determined using a village-level consumer price index. Village-level variables were collected using a standard questionnaire administered to a group of key individuals (e.g. the village chief, commune officials, shopkeepers, etc.). Qualitative discussions were also undertaken separately around key drivers of changes in livelihood strategies, the impacts of the PAs and external drivers, use of forest resources, and land in order to substantiate the quantitative findings of the models.

Analyses

The poverty score results (measured using the BNS) and the household rice harvest in 2007/8 were used as the dependent variables for all further analyses. The impact of PAs on human wellbeing at the household level was analysed using two different techniques: mixed effects models and matching estimators. Mixed effects models were undertaken in R 2.13.0 (R Development Core Team, 2011) using package “nlme” (Pinheiro et al., 2011), in order to account for the random effect due to repeated surveys of different households from the same village. The models were based on the entire dataset of 871 households from the 20 villages. Given the large number of possible explanatory variables, competing models were developed based on the *a priori* research hypotheses (Burnham et al., 2011). The initial model included all main effects and 2-way interactions relevant to the research question. Model selection was conservative, using second-order AICc (Akaike’s information criterion corrected) values to compare competing models, as is appropriate when the number of parameters being estimated is <40 (Burnham & Anderson, 2002). All terms with AICc Δ values of >4 (“considerably less” empirical support; Burnham & Anderson, 2002) were removed, with the exception of interactions that were relevant to the research hypotheses (e.g. those involving PA) that had some empirical support. Models were compared using maximum livelihood estimation, with coefficients for the final models estimated using restricted maximum likelihood (Crawley, 2007). Contrasts in R were used to compare differences between the types of village (Crawley, 2007): (*1*) comparing households inside PAs with households in control villages outside PAs, to evaluate the impact of PAs; and (*2*) comparing households on the border of PAs with households inside PAs. Model validation included plotting the residuals against the fitted values to check for the homogeneity of the variance, against the explanatory variables to check for any unexplained patterns, and histograms of the residuals and normal quantile-quantile plots to check for normality of the errors. The model selection procedure identified the most conservative model for both household poverty (measured using BNS Score; see Appendix, Table S3.1) and rice harvest (see Appendix, Table S3.2), which contained the least number of parameters for the highest level of explanatory power, and all the variables relevant to addressing the principle research questions.

Matching estimators were calculated only for the 325 households from the 11 villages inside the two protected areas, matched with the 134 households from the 5 control villages. The clustering effect of village was accounted for using the equations developed by Hansom and Sunderam (2011). The matched households were also subsampled also to compare specific effects for different types of livelihood strategies: resin-tapping, owning >1 hectare, and having a shop or family business.

3.4 Results

Factors affecting Household Poverty Status and Rice Harvests

Households varied significantly in their poverty status, depending upon household characteristics and livelihood strategies. Less poor households (those with higher BNS Scores) were more likely to be larger, with more working adults and a lower dependency ratio (the ratio of the number of working adults to the total household size); they were also more likely to have male household heads, that were more educated and older than poorer households (Table 3.2a). These results affirm the importance for household wellbeing of education and having sufficient labour for farming, collection of forest resources, and other livelihood strategies. 97% of households listed farmer as one of their occupations. Lower household poverty were associated with engaging in more livelihood strategies, such as being a resin-tapper, employment, and operating a village shop or providing a service (such as a carpenter, trader, etc.); or agricultural intensification by owning larger amounts of land, and having draft livestock or mini-tractors for ploughing, pulling carts, etc. Poorer households were more likely to rent out their labour, perhaps due to lack of other livelihood opportunities, and were more likely to practice shifting cultivation rather than permanent paddyfield rice. Less poor households also collected more resin and had more cattle. The majority of these variables were highly significant determinants of household poverty status, based upon the final selected mixed effects model (Table 3.3a).

Some of the same factors predicted household rice harvests (Table 3.2b), and were supported by the mixed effects model (Table 3.3b). There were some differences; households that were employed or operated a shop or business had significantly lower rice harvests (Table 3.3b), suggesting that they were diversifying into these nonfarm livelihood strategies.

Table 3.2. Household characteristics, poverty, livelihood strategies and assets for 871 households in Preah Vihear province, Cambodia, in 2008, showing the poorest and richest quintiles and the least productive and most productive rice harvest quintiles.

		(a) Poverty Score Quintiles		(b) Rice Harvest Quintiles	
	All	Bottom	Top	Bottom	Top
Households	871	174	174	174	147
Household Size (people)	5.7	5.3	6.1	4.9	6.5
Working Adults (people)	3.1	2.7	3.6	2.5	3.8
Dependency Ratio	1.0	1.2	0.8	1.1	0.8
Female Headed Households (%)	9%	12%	5%	14%	6%
Household Head Education (years)	2.3	1.4	3.8	2.1	3.0
Household Head Age (years)	41.1	38.3	42.7	39.8	44.9
Household Status					
Poverty Score[a]	9.3	4.9	13.9	7.0	12.0
Rice Harvest (kg)	1732	771	2857	163	4295
Livelihood Strategies					
Resin-tappers (%)	44%	30%	53%	24%	60%
Rice Farmers (%)	90%	78%	97%	51%	100%
Have >1 Hectare (%)	73%	36%	96%	40%	100%
Shifting Cultivation Farmers (%)	39%	52%	20%	39%	30%
Employed (%)	7%	3%	14%	8%	8%
Provide a Service or Shop (%)	17%	4%	37%	19%	24%
Rent out labour (%)	3%	8%	1%	10%	0%
Household Assets					
Resin yields (litres)	501	229	722	182	833
Cattle (heads)	3.9	1.5	7.2	1.5	6.1
Draft Cattle (%)	35%	10%	64%	14%	51%
Mini-tractor (%)	28%	1%	60%	9%	56%

[a] Measured using the Basic Necessities Survey

Table 3.3. Final mixed effects model for the effect of household and village-level variables on (a) household poverty status (measured using BNS score) and (b) household rice harvest in 2007/8. The table shows the coefficient values for the final model, all of which have a high level of empirical support based on the AICc Δ values. The model selection tables are given in the Appendix, Table S3.1 and Table S3.2.

Coefficients	(a) Poverty Score model	(b) Rice Harvest model
Village Type [Border PA]	2.156	35.633
Village Type [Inside PA]	2.223	29.494
Village Type [Outside PA]	2.376	34.385
Female-headed household [Yes]	0.922	
Female-headed household [Yes] * Number of livelihood strategies	-0.613	
Education of household head (yrs)	0.441	
Number of Working Adults		9.085
Rent out labour [Yes]	-0.756	-11.759
Own >1 hectare [Yes]	1.120	5.169
Own mini-tractor/draft animals [Yes]	0.233	6.512
Own >1 hectare [Yes] * Own mini-tractor/draft animals [Yes]		8.891
Employed [Yes]		-9.025
Operate a business [Yes]	-0.194	-5.320
Education of household head (yrs) * Operate a business [Yes]	0.427	
Resin-tapper [Yes]	-0.814	-0.006
Rice harvest in 2007/8 (kg)	0.001	
Rice harvest in 2007/8 (kg) * Time to Provincial Capital (hours)	-0.000	
Number of Cattle	1.150	5.436
Number of Cattle * Own mini-tractor/draft animals [Yes]		-4.130
Number of livelihood strategies	0.887	5.041
Village Population Size (households)	0.006	0.490
Years of schooling in the village	0.171	
Time to Provincial Capital (hours)	-0.079	-17.092
Time to Secondary School (hours)		-33.812
Time to Provincial Capital (hours) * Time to Secondary School (hours)		13.151
Village Type [Inside PA] * Number of cattle	-0.481	
Village Type [Outside PA] * Number of cattle	-0.816	
Village Type [Inside PA] * Resin-tapper [Yes]	0.371	-1.035
Village Type [Outside PA] * Resin-tapper [Yes]	-0.677	-6.741
Village Type [Inside PA] * Own mini-tractor/draft animals [Yes]	1.137	
Village Type [Outside PA] * Own mini-tractor/draft animals [Yes]	1.301	
Village Type [Inside PA] * Own >1 hectare [Yes]		4.713
Village Type [Outside PA] * Own >1 hectare [Yes]		-2.716
% residual variation due to the random effect of Village	6.2%	4.3%

The two primary development paths were therefore (*1*) intensification of agriculture by adopting permanent paddyfield rice (rather than practicing shifting cultivation), acquiring greater land holdings, and increased mechanization; and (*2*) diversification into non-agricultural livelihoods such as employment, operating a shop or providing a service. More educated households were more likely to diversify into non-agricultural strategies (Table 3.3a).

Village characteristics had a strong effect on household poverty and rice harvests. Less poor households were found in villages that were closer to the Provincial Capital, were larger, and had more years of schooling available in the village (Table 3.3a). Similarly, greater rice harvests were found in villages that were closer to the Provincial Capital or secondary schools, and were larger (Table 3.3b). Villages that were remote from the Provincial Capital were therefore less able to profit from higher agricultural harvests, probably due to restricted market access. The travel time to the provincial capital was a suitable proxy for access to major services such as hospitals ($r = 0.908$, $n = 20$, $P < 0.001$), high schools, large markets, and was highly correlated with the distance to all-weather roads ($r = 0.689$, $n = 20$, $P = 0.001$). Travel time to the nearest secondary school was a suitable proxy for access to the nearest major population centre, where full-day markets (correlation $r = 0.644$, $n = 20$, $P = 0.002$), shops and health services were more frequent.

Effect of Protected Areas on Household Poverty and Agricultural Productivity

Households bordering PAs were considerably less poor than households inside PAs (Table 3.4, Figure 3.3a, Difference = 0.93, $P < 0.001$), using poverty measured by BNS Score. Analysis of the livelihood strategies practiced by border households suggests that they are further advanced along the two development pathways identified in comparison with the other village types: (*1*) agricultural intensification, through having greater land holdings and mechanization; and (*2*) diversification, through employment, operating a shop or providing services (Table 3.4). However, the mixed effects model, which includes the village-level variables, indicated that these differences could be fully explained by the village characteristics (Figure 3.3a, Model Coefficient = 0.06, $P = 0.408$). Border villages were larger, closer to the Provincial Capital, and had better schools than villages inside PAs (Table 3.5), which is sufficient to explain the difference between the village types. It is unlikely that these differences can be explained by the PA intervention, because there was no evidence that the presence of PAs had influenced infrastructure development decisions (e.g. on roads, school-building, etc.).

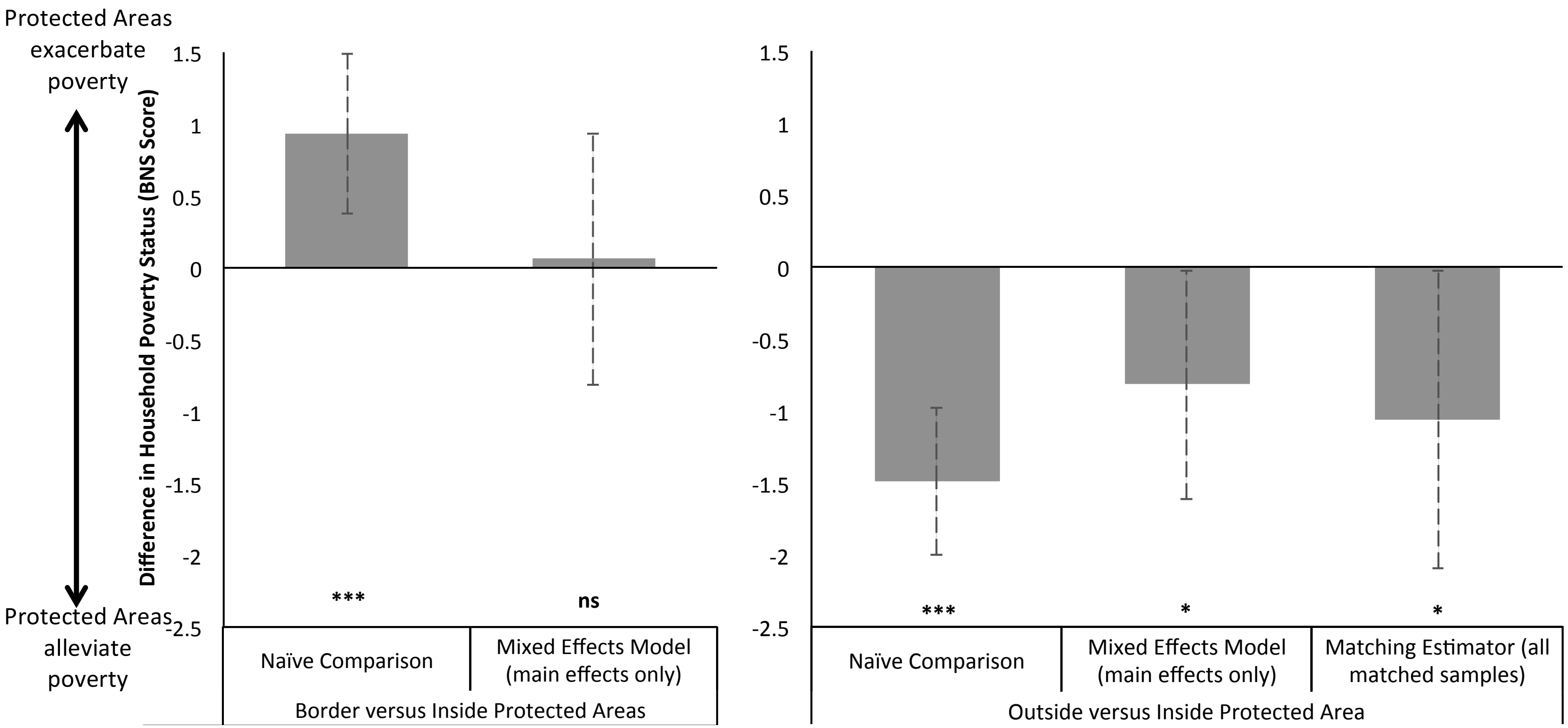
Table 3.4. Differences in household status and livelihood strategies between households bordering, inside and controls outside Protected Areas in the Northern Plains of Cambodia in 2008.

	Border PA	Inside PA	Controls	Tests of difference[a] (Inside PA vs Controls)
Villages	4	11	5	
Number of Households	162	504	205	
Household Characteristics				
Household Size (people)	5.5	5.7	5.9	
Working Adults (people)	3.2	3.1	3.0	
Dependency Ratio	0.8	1.0	1.1	
Female-headed Households (%)	12%	9%	7%	
Household head education (years)	2.8	2.6	1.3	
Household head age (years)	40.9	42.2	38.6	
Household Status				
Poverty Score[b]	10.4	9.4	8.0	*
Rice Harvest (kg)	1999	1828	1286	ns
Livelihood strategies				
Resin-tappers (%)	31%	54%	29%	***
Rice Farmers (%)	93%	89%	92%	ns
Have >1 hectare (%)	88%	71%	63%	*
Shifting Cultivation Farmers (%)	38%	37%	45%	*
Employed (%)	14%	7%	3%	ns
Provide a Service or Shop (%)	28%	15%	14%	ns
Household Assets				
Resin yields (litres)	357	626	307	*
Cattle (heads)	2.8	4.5	3.5	ns
Draft Cattle (%)	20%	39%	37%	ns
Mini-tractor (%)	32%	28%	26%	ns

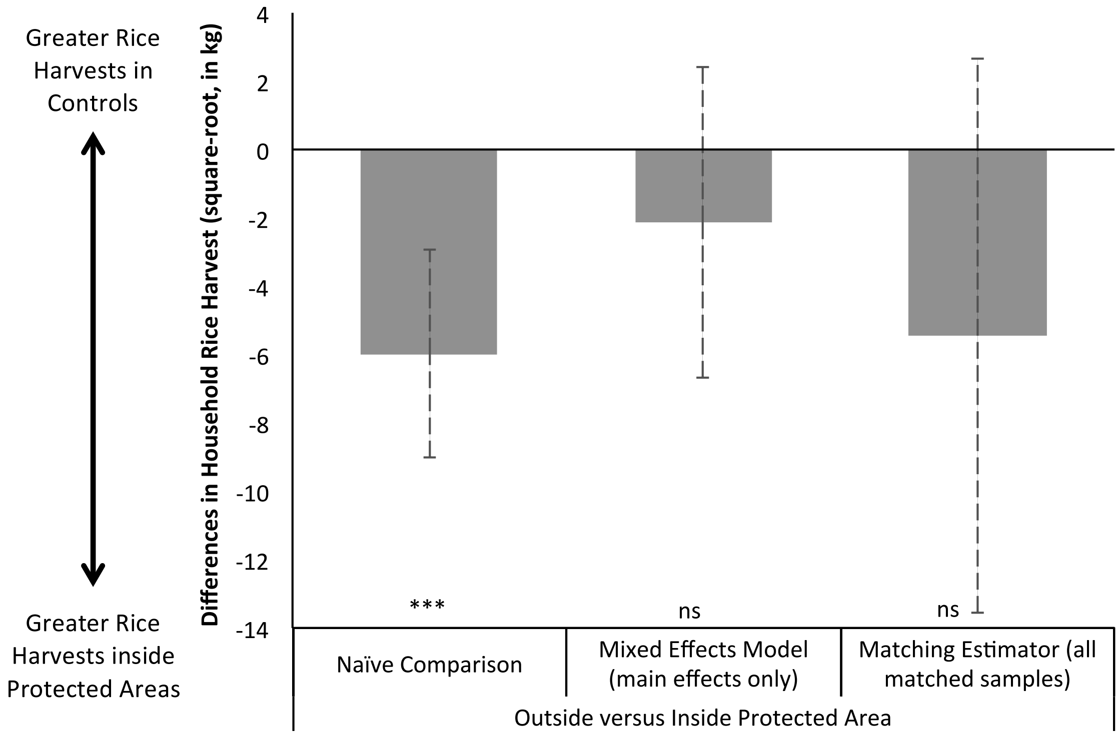
Notes:
[a] Tests of difference are mixed effects models for continuous variables (BNS Score, Rice harvest, Resin yields, Cattle), and chi-squared tests for categorical variables. The BNS Score and Rice harvest models are given in Table 3.3.
[b] Measured using the Basic Necessities Survey
Significance values for the null hypothesis of zero impact: ns = not-significant, * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$.

Figure 3.3. Average differences between households bordering, inside and outside protected areas for (a) for the household poverty score, measured using the Basic Necessities Survey, and (b) household rice harvests.

(a) Differences in the poverty status of households bordering, inside and outside Protected Areas



(b) Differences in the rice harvests of households outside and inside Protected Areas



Notes: Significance values for the null hypothesis of zero impact: ns = not-significant, * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$.

Table 3.5. Differences between villages bordering PAs, inside and outside the PAs for village-level variables.

Variable	Border PA	Inside PA	Controls
Village Population Size (households)	164.8	140.6	148.6
Travel time to Provincial Capital (hours), dry season	3.3	5.5	4.4
Travel time to Secondary School (hours), dry season	0.8	3.1	2.7
Distance to nearest all-weather road (km)	9.9	26.9	25.7
Number of school years available in the village	6.3	5.4	4.0

Households inside PAs differed significantly from the control households in terms of the livelihood strategies practiced (Table 3.4). In particular, over half of the households inside PAs were resin-tappers, in comparison with only 29% of households in controls, and control households were more likely to practice less intensive agriculture, such as shifting cultivation, with smaller land-holdings. Once village-level variables were taken into account in the mixed effects models, households inside PAs were significantly less poor than households outside PAs (Figure 3.3a, Table 3.6a). Very similar results were obtained for the matching estimator, comparing households inside PAs with matched households from the control villages (Figure 3.3a, Table 3.6a). Unlike with the border villages, these differences could not be explained by village characteristics, and were more likely to be due to the PA intervention itself.

There were no significant differences between rice harvests for households inside and outside PAs based on the mixed effects model (Figure 3.3b, Table 3.6b) and the matching estimators (Figure 3.3b, Table 3.6b). PAs therefore had little impact on average household rice harvests.

Impacts of Protected Areas on Livelihood Strategies

Resin-tappers benefited most from PAs. Resin-tapping households inside PAs were significantly less poor than resin-tappers outside PAs, based on the mixed effects model and the matching estimator, using BNS Score as the measure of household poverty (Table 3.6a). Similarly, resin-tappers inside PAs had significantly greater rice harvests than resin-tappers outside PAs based on the mixed effects model (Table 3.6b); the equivalent matching estimator is significant at the $P < 0.10$ level (Table 3.6b). By contrast, for those that did not resin tap there was no difference between households inside and outside PAs in terms of poverty and rice harvests (Table 3.6).

Table 3.6. Average differences between households with different livelihood strategies in villages inside PAs and controls outside PAs for (a) household poverty, measured using BNS Score[a], and (b) household rice harvests. Results are based on mixed effects models and matching estimators.

Difference test	Matching Estimator				Mixed Effects Models		
	N	Coefficient	Standard Error	Significance	Coefficient	Standard Error	Significance
(a) Difference in Household Poverty Status (BNS Score[a])							
Inside PAs vs Controls (all)	325	1.06	0.53	*	0.82	0.37	*
Inside PAs vs Controls (resin-tappers)	150	2.19	0.41	***	1.05	0.37	**
Inside PAs vs Controls (do not resin-tap)	175	0.07	0.71	ns			ns
Inside PAs vs Controls (own >1 hectare)	240	1.01	0.53	†			ns
Inside PAs vs Controls (do not own >1 hectare)	85	1.22	0.67	†			ns
Inside PAs vs Controls (shop/business)	45	1.00	0.75	ns			ns
(b) Difference in Household Rice Harvest in 2007/8 (kg, square root transformed)							
Inside PAs vs Controls (all)	325	5.45	4.13	ns	2.03	2.12	ns
Inside PAs vs Controls (resin-tappers)	150	10.43	5.58	†	5.94	2.61	*
Inside PAs vs Controls (do not resin-tap)	175	1.02	3.71	ns			ns
Inside PAs vs Controls (own >1 hectare)	240	7.81	4.68	†	7.42	2.49	**
Inside PAs vs Controls (do not own >1hectare)	85	-1.72	3.33	ns			ns
Inside PAs vs Controls (shop/business)	45	12.68	8.88	ns			ns

Notes:

[a] Basic Necessities Survey Score (measurement of household poverty status)

Significance values for the null hypothesis of zero impact: ns = not-significant, † = $P < 0.10$, * = $P < 0.05$, ** = $P < 0.01$, *** = $P < 0.001$.

Positive coefficients indicate that protected areas alleviate poverty or increase rice harvests, whereas negative coefficients indicate that protected areas exacerbate poverty or reduce rice harvests.

Households with larger land holdings had greater agricultural productivity inside PAs than outside. Rice harvests for households inside PAs owning >1 hectare were significantly greater than households owning similar amounts of land outside PAs, based on the mixed effects model (Table 3.6b); the equivalent matching estimator is nearly significant (Table 3.6b). By contrast, there was no difference in rice harvests between households owning <1 hectare inside and outside PAs (Table 3.6b).

No differences between households inside or outside PAs were observed for other livelihood strategies (such as operating a household business; Table 3.6) for either household poverty, as measured by BNS Score, or rice harvests, indicating that the PA intervention had limited impact on these strategies.

3.5 Discussion

Measuring the social impacts of protected areas

A simple comparison of households inside the PAs with bordering villages would come to the conclusion that PAs exacerbate local poverty (Figure 3.3a). The results of the impact evaluation show that this would be an inappropriate comparison, because border villages were closer to market centres, other services, and main roads, all of which had positive impacts on local poverty status. This demonstrates the importance of using impact evaluation survey designs that make appropriate comparisons. Impact evaluation methods have been criticized as too expensive for widespread use in programme evaluation (Richards & Panfil, 2011). The cost of the 2008 surveys analysed here was US$50,000 (including technical assistance and analysis), which would be affordable in the context of many large conservation-development programmes. Adoption of impact evaluation methodologies does, however, require access to appropriate technical expertise to design the surveys and analyse the results. Matching methods only give robust results if the matching process controls for the other drivers of change in poverty. Using matching methods for social impact assessment therefore requires a strong prior understanding of the system in order to select appropriate matching variables (Ravallion, 2006). Poor matching designs might identify an effect when in fact none exists or mask effects. The accuracy of estimates can be improved by triangulation of results with other methods (such as regression models and qualitative assessments), and using repeat surveys to calculate difference-in-difference estimators (Ravallion, 2006). In this study, the matching estimators were broadly similar to the results for the mixed effects models on a much larger sample. However, the accuracy of the analysis is dependent upon the validity of the original identification of the matched control villages. This was addressed in the current study by selecting matching variables from 2005 (prior to the commencement of PA management), to ensure that they were exogenous to the PA intervention (see Supplemental Materials for details).

The Basic Necessities Survey (BNS) was a relatively inexpensive and rapid method to assess local perceptions of poverty and to collect data on household poverty status. Again the principal technical hurdles were during the design phase, since the BNS required considerable piloting before a suitable list of items was developed and to train field surveyors in the approach. Considerable prior knowledge of livelihoods in the study area was needed in order to develop an appropriate list. The BNS performed similarly to the two other measures of poverty described here – the basket of assets and the participatory wealth ranking – however its relationship to standard measures of poverty, such as household income or consumption, is unclear. Validating the results of the BNS against income or consumption data would give greater confidence in the survey results. The BNS also captured other salient aspects of wellbeing. For example, Cambodians ranked highly the number of ceremonies in their village as a basic necessity, even if they themselves could not afford to host a ceremony, as a measure of overall social wellbeing.

Factors affecting Household Poverty Status and Agricultural Productivity

Cambodia underwent rapid economic growth during 1998-2008 with annual GDP increases of 7-13% (World Bank, 2012), leading to a reduction of more than 1% a year in the poverty headcount (World Bank, 2009). Reductions in poverty have been greater for people in urban areas, rather than for the rural poor that make up the majority of the population (78%; World Bank, 2009). Nearly all the people in the study area were poor subsistence farmers, if the survey results here are compared with national-level indicators (World Bank, 2009). Average household rice harvests were 1,732kg in 2008, barely sufficient to support a family for a year, suggesting that the majority of households were in rice deficit and dependent on other sources of cash income to buy food.

The most significant source of non-agricultural income in the study area is collection of liquid resin from dipterocarp trees (McKenny et al., 2004). The importance of resin to the household economy of forest communities in Cambodia has been well documented, in particular as a source of cash income to buy food in times of rice deficit (Evans et al., 2002; McKenny & Prom, 2002; McKenny et al., 2004; Hansen & Top, 2006). Income from resin is also invested in livelihood strategies. Resin is therefore an example of a forest resource that is both a 'safety net' for vulnerable households and a critical source of cash providing a 'pathway out of poverty' (Angelsen & Wunder, 2003; Ruiz-Pérez et al., 2004). Resin is the only forest resource to have local traditional ownership and harvesting rules, indicating its importance to local people (Ostrom, 1990). Under these rules, trees are individually owned, can be inherited, and rights are maintained even if resin-tappers move or live far away. To some extent these rules are acknowledged in Cambodian Law, which recognizes the user rights of tappers and prohibits the clearing of resin trees (Prom & McKenny, 2003).

The usefulness of resin as a development pathway is, however, limited by several factors. Firstly, the majority of useable trees in the area are claimed or tapped (A. John, pers. comm.). New families and immigrants are therefore reliant upon inheriting or buying resin tress. Secondly, the resin trade is monopolized by a small number of traders that pay high formal and informal taxes to transport resin (Prom & McKenny, 2003). This constrains the price that resin tappers can receive; most of the profits are captured higher in the value chain. Finally, large-scale concessions for logging (prior to 2002) or agri-business (since 2005) clear resin trees, often despite strong local opposition and for inadequate compensation (GTZ, 2009; World Bank, 2006). Strengthening resin tree tenure and reforming the trade barriers for resin is likely to have significant positive impacts on local livelihoods.

There is evidence from this study that rural people are beginning to diversity out of subsistence agriculture and forest resource collection. Two development pathways are supported by the data: the agricultural path (the traditional rural development model) and the multiple-activity path (rural diversification into nonfarm activities; Wunder, 2001). Some households are investing in improved agriculture, through purchase of mini-tractors and expansion of areas under cultivation. Other households are diversifying into nonfarm livelihoods, such as commercial activities or employment, in a minority of cases leading to households abandoning agriculture all together. Expansion into business activities is heavily related to the availability of education. Both development pathways are strongly related to access to markets and services, through being closer to major towns and roads.

Impacts of Protected Areas on Poverty, Agriculture and Local Livelihoods

Protected Areas cover approximately 12% of the world's terrestrial surface, in almost every country (UNEP-WCMC, 2012). Reviews have suggested that between 56% and 85% of PAs in developing countries contain local people (Brockington & Igoe, 2006), including extractive reserves and community conserved areas (Berkes, 2009). Given the controversies about whether PAs exacerbate local poverty or might contribute to poverty alleviation (Roe, 2008), there is a critical need for evidence to inform the debate. Very few well-designed empirical studies have examined PA impacts on local people, and these have generally found limited positive effects (Andam et al., 2010; Sims 2010; Naughton-Treves et al., 2011). In the current study, there was no evidence that four years of PA management had overall negative impacts on local livelihoods, either in terms of household poverty or harvests of rice, the staple crop that provides the basis of local diets. These results should be seen in context, however. Under Cambodian law, local people are entitled to remain inside PAs and to continue to practice traditional subsistence livelihoods. PA enforcement activities primarily targeted wildlife hunting, logging for commercial purposes, and agricultural expansion. Densities of wildlife in the PAs were also low (O'Kelly et al., 2011), leading to limited human-wildlife conflict. This context is not dissimilar from many other PAs in developing countries, that contain local people and where management budgets are limited (Bruner et al., 2004; Brockington & Igoe, 2006).

The Cambodian PAs did provide some positive impacts for local people, by providing security of land tenure and forest resource access – effectively a 'resource pool protection effect' for villages inside the PA. During the 1990s, forestry concessions were designated covering 7 million hectares (or 70%) of Cambodia's forests, all located outside PAs (Cambodia R-PP, 2011). Resin trees are all dipterocarp species, which are highly valuable timber species and are preferentially targeted by loggers. The subsequent legal and illegal logging led to widespread protests, particularly by resin-tappers, causing all forestry concessions to be suspended by the Royal Government in 2002 (World Bank, 2006). Subsequently, selective illegal logging has continued to be widespread, particularly for high value species such as dipterocarps. PAs have successfully protected resin trees during this time, both from the commercial companies (since forestry concessions were not declared inside PAs) and from illegal loggers. This explains why resin-tapping was a much more important livelihood strategy inside PAs, and why resin-tappers inside PAs were significantly less poor than resin-tappers outside PAs.

In the 2000s, land clearance has replaced logging as the major driver of change. National annual deforestation rates were 0.5% during 2000-10 (Forestry Administration, 2007 & 2011), despite the fact that since 2002 most forest clearance has been illegal. Consequently Cambodia has one of the highest rates of land-use change in the region (FAO, 2011). Large-scale resource exploitation and land-use change is primarily driven by economic land concessions, primarily for cash crops and rubber (Cambodia R-PP, 2011), which appropriate and clear large areas of forest. Approved concessions are currently in excess of 1 million hectares (So, 2010), or 6% of Cambodia's area.

Concessions are often met with strong local opposition, and are thought to deliver few benefits to local people who lose access to land and forest resources for minimal compensation or development opportunities (UN OHCHR 2007; GTZ, 2009; So, 2010). Landlessness has risen to 20%, and 40% of rural households have farms less than 0.5 ha, i.e. less than half of the minimum area required to meet nutritional needs (GTZ, 2009). Insecure local tenure over land and forest resources provides little incentive for local people to engage in sustainable resource management (An, 2008). PAs provide some protection to local residents from the risk that land is appropriated for other causes, and this may explain why land-owners inside PAs had greater agricultural productivity than land-owners outside PAs. However, the security of tenure afforded by PAs in Cambodia is currently in doubt. Within the past two years, significant forest areas of Cambodia's PA networks have been degazetted for economic land concessions, a trend which seems likely to continue (Cambodia Daily, 2011). The degazetting of PAs has serious implications not just for biodiversity conservation, but also for local people's welfare.

Chapter 4. An evaluation of the effectiveness of a direct payment for biodiversity conservation: the Bird Nest Protection Programme in the Northern Plains of Cambodia[8]

Abstract

Direct payments for the protection of biodiversity (a type of payment for environmental services) have been proposed as an effective tool for delivering conservation outcomes, in a way that also delivers development benefits to local people. Using an impact evaluation framework, this paper analyses the effectiveness of a direct payment programme that was established for nine Globally Threatened bird species in the Northern Plains of Cambodia. The programme provided conditional payments to local people to protect nests, since most of the species were highly threatened by the collection of eggs and chicks. Since the programme's inception in 2003 it has protected >2,700 nests over >2000km^2 of habitat at a cost of $30,000 annually, with 71-78% of the costs paid directly to local people. Payments significantly improved the success rates of protected nests in comparison with control sites, leading to population increases for at least three species. However, payments did not influence other threats to species, such as land clearance, and have failed to arrest declines in at least one species' population. The average payment per protector was a significant contribution to incomes in remote rural villages. However, the programme only benefited a small proportion of people, causing some local jealousies and deliberate disturbance of nesting birds. The programme demonstrates that direct payments can be a highly effective conservation tool in those cases where payments correctly target the cause of biodiversity loss. The results also suggest that it is important to consider how decisions over beneficiaries are made, especially in situations where property rights over biodiversity are unclear, if payments are to be socially acceptable. This has important implications for the design of payment schemes in conservation more generally.

4.1. Introduction

The history of conservation and development for the past 30 years has been dominated by discussions over how to appropriately integrate conservation and poverty alleviation goals (Roe, 2008) and navigate trade-offs between these two objectives (McShane et al., 2011). Dominant discourses include viewing local poverty as a threat to conservation that must be addressed, for example leading to over-exploitation of threatened species, or emphasise a rights-based approach

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that conservation activities should not compromise local poverty reduction (Adams et al., 2004). Direct payments for biodiversity conservation – a type of Payment for Environmental Services (PES) – have been proposed by Ferraro (2001; Ferraro & Kiss, 2002) as a more effective mechanism for encouraging local actors to deliver conservation outcomes in a way that also provides local development benefits, in comparison with indirect interventions such as integrated conservation and development programmes. Based upon Ferraro (2001; Ferraro & Kiss, 2002) a direct payment scheme involves a negotiated payment provided to a seller conditional upon a particular conservation outcome being achieved. The approach assumes that the seller has partial or total control over the conservation outcome. This definition is consistent with the broad framework for analysing all types of PES proposed by Sommerville et al. (2009), which is less restrictive than the original PES definition of Wunder (2007). Direct payments, and PES approaches in general, have received a significant level of interest since they were first proposed, and a relatively large number of both government- and user-financed programmes have been identified (for reviews see Ferraro & Gjertsen, 2009; Milne & Niesten, 2009; Pattanayak et al., 2010). However, very few studies have analysed the extent to which payments are effective at conserving biodiversity (Pattanayak et al., 2010); the majority of evaluations that have been completed are focused mainly on habitat conservation and forest protection (Pattanayak et al., 2010). Similarly, very few studies report the extent to which payments contribute to local livelihoods. Evaluating existing direct payment programmes is particularly relevant given the rapid expansion of proposed PES programmes, both nationally and internationally (such as Reducing Emissions from Deforestation and forest Degradation, REDD+; Clements, 2010).

The effectiveness of direct payments at conserving biodiversity depends upon the extent to which they adequately address the principal threats to biodiversity, as with any conservation intervention (Salafsky et al., 2002). Proponents of direct payments have argued that a key advantage is that payments are targeted (Ferraro, 2001), however this is only appropriate when the activities targeted are appropriate to reduce biodiversity loss. This implies the importance of having a sound understanding of the underlying dynamics of the social-ecological system within which the direct payments interventions are implemented (Ostrom, 2007). The underlying causes of biodiversity loss are complex and operate at multiple scales – from local to national to global – and payments, due to their targeted nature, may only be effective at addressing some of these.

Payments also influence the social system, through the provision of economic incentives to people involved in the programme. Although economic considerations certainly influence individuals' decisions to engage in behaviours (Persky, 1995), additional factors including social norms (Bowles, 2008) and procedural and distributive fairness are known to impact individuals' motivation (Fehr and Falk, 2002). Perceptions of unfairness can undermine the effectiveness of incentives, even if they provide apparent net benefits (Proctor et al., 2009; Sommerville et al., 2010). In addition to providing economic incentives, developing positive local attitudes is therefore key to any direct payment scheme. Local perceptions of a direct payment programme may be particularly important when local property rights are unclear, and therefore the decision over who benefits is not straightforward. In many countries, land ownership and resource tenure are poorly defined, with land and resources technically still owned and managed by the state (Agrawal et al., 2008), and institutions are weak (Barrett et al., 2001). Unclear property rights and weak institutions are thought to make implementation of any payment programme considerably more difficult (Wunder, 2007; Engel et al., 2008).

This paper evaluates the effectiveness of a direct payments programme for protection of globally threatened nesting birds in the Northern Plains of Cambodia. The Northern Plains was considered an ideal landscape to trial a direct payments programme; the area supports a large number of bird species of high conservation concern that are heavily threatened by annual collection of eggs and chicks for consumption and trade. The effectiveness of the programme in conserving biodiversity was determined using impact evaluation methods; comparing the success rate of nests protected by the programme with those from matched controls without an intervention (Ferraro & Pattanayak, 2006). Its effectiveness in providing development benefits was determined by investigating the distribution of payments and local perceptions of the scheme. We address four research questions: (*1*) how have payments affected the threats to nesting birds?; (*2*) have payments for nest protection led to increases in species' populations?; (*3*) was the distribution of the protection payments fair and equitable?; and (*4*) to what extent have payments changed local attitudes towards bird conservation? Based upon the answers to these questions, we consider the extent to which the payments were achieving their goals in the context of the threats to the target species and the mechanism by which the social and economic incentives generated by the payments led to effective biodiversity conservation.

4.2 Methods

Bird Nest Protection Programme

The Bird Nest Protection programme was initiated in 2003 by the Wildlife Conservation Society (WCS) in the Northern Plains of Cambodia in collaboration with the Ministry of Environment and the Forestry Administration of the Ministry of Agriculture, Forestry and Fisheries. The programme was aimed to rapidly locate, monitor and protect the globally important bird populations found across the landscape, which were heavily threatened by nest collection by local people. The programmes was designed as complement to longer-term activities to strengthen institutions for environmental protection, such as protected areas, and to clarify land tenure and resource management rights of local people. Originally initiated on a pilot basis, by 2009 the programme was operating in 24 villages across both conservation areas. The same approach has subsequently been replicated at other sites in Cambodia by several other organisations (including WWF and BirdLife International).

Under the programme, nests were located by local people (usually resin-tappers or local farmers), or community rangers contracted by WCS seasonally to undertake research. The rangers were often well-known hunters, hired specifically to reduce hunting pressure and for their knowledge of species' ecology. Local people received a reward of US$5 for reporting a nesting site. For all species except Giant Ibises a permanent protection team of two people was established for each nest, or colony of adjutants or darters. The people who found the nest were invited to form the protection team, otherwise nest protectors were sought from local forest product collectors or the nearest village. Giant Ibises were not thought to be valued for trade or consumption and hence were not given intensive protection, but predator-exclusion belts were placed around the base of nesting trees from 2006 because these had been shown to increase nesting success (Keo et al., 2009). Prior to 2008 protectors received a payment of $1 per day for their work and an extra $1 per day upon completion if chicks successfully fledged. The total payment of $2/day was judged an acceptable daily wage based on village consultations. From 2008 payments were increased to $2.50/day total due to rising food prices based upon requests from local nest protectors. Community rangers received a monthly salary ($50-$70) plus the same daily payment. Protection teams remained in place until the last chick fledged, or until the eggs hatched in the case of Sarus Cranes (which are precocial). All of the costs of the programme were recorded, including the payments made to protectors, and other costs such as monitoring visits, travel and surveys.

Protection teams were visited every 1-2 weeks by the community rangers, and monthly by WCS monitoring staff to collect data on the location of each active nest, dates of laying, hatching and fledging, habitat type, nest characteristics, and the number of birds, eggs, and chicks present for each species on each visit. Nests were deemed to have failed if they became unoccupied prior to fledging. Monitoring staff investigated all cases of nest failure to determine the cause, and payments were not made if nests failed due to human disturbance or collection.

Evaluating the conservation impact of the programme

In order to evaluate the impact of the programme on nesting success, from 2009-11 nests of the same species were monitored, but not protected, by community rangers around seven control villages in the same landscape. Controls were selected using covariate matching, a technique used to select sites that share similar characteristics to intervention sites (Abadie & Imbens, 2006). Matching used four variables – the village population size in 2005, forest extent in 2006, and distances to nearest all-day market and all-weather roads. These variables were chosen because villages involved in the programme tended to be smaller and located in remote areas that had high forest cover. Covariate matching was carried out in R 2.13.0 using the package 'matching' (R Development Core Team, 2011) to select controls that were statistically indistinguishable with respect to the matching variables from villages where the bird nest programme was being trialled. Balancing tests were used to show that there were no significant statistical differences between the final matched sample and the villages engaged in the programme, for the variables used (see Chapter 2 for details; Figure 2.1 shows the location of the selected control villages). All nests found around the control villages were monitored using the same data collection techniques as used for the nests engaged in the protection programme, and cases of nest failure were investigated by monitoring staff to determine the cause.

Nest success rates during 2009-2011 were calculated for the 65 control nests of Lesser Adjutant and Sarus Crane, 527 protected nests of Lesser Adjutant and Sarus Crane, 22 protected nests of Greater Adjutant and 60 unprotected Giant Ibis nests. Daily nest survival rates were calculated in programme MARK, assuming a constant rate for each species (Rotella, 2011). Post-hoc tests were done using CONTRAST (Hines & Sauer 1989) comparing nest survival between controls and protected nests of Lesser Adjutant and Sarus Crane, between protected nests of Greater and Lesser Adjutant, and between the unprotected Giant Ibis and protected nests of Lesser Adjutant and Sarus Crane.

Population estimates for each species in each year were calculated based on the number of occupied nests observed, as a measure of the number of breeding pairs. Such population estimates can be problematic because the detectability (the proportion of nests present in the area but not seen) might vary over time and could not be calculated accurately. Detectability could fluctuate between years for a range of reasons, including changes in survey coverage or observers, or in nesting behaviour; it could also trend over time, for example if observers became more efficient at finding nests. Changes in survey coverage could be accounted for by estimating the area visited in each year, but changes in observer efficiency could not, and could be a source of bias in this dataset leading to trends appearing more positive than they really are. From 2005, all rangers and survey staff were ask to maintain tracks of their trips using Global Positioning System (GPS) devices, by recording one point every 30 minutes. Survey coverage was then estimated as the number of kilometre squares visited during these surveys from July-December each year, corresponding to the period in which nests were located. Population data were analysed using generalised linear models with quasi-poisson errors and a log-link function in R 2.12.2 (R Core Development Team, 2011) to investigate differences in trends over time for each species.

Evaluating the social impact of the programme

The distribution of payments to local people between and within villages was investigated during four seasons, from 2005 until 2009. For each village participating in the programme, data were recorded on the total number of households, the number of households with nest protectors, the identity and occupation of nest protectors and all payments made. These data were used to determine the percentage of households engaged in the programme, the distribution of the payments made between villages, and the distribution of payments made to individual nest protectors. The payments received by protectors were compared to standard estimates of household consumption in rural forested regions of Cambodia, available from the 2007 Cambodia Socio-Economic Survey (World Bank, 2009).

Local attitudes to the programme were investigated by conducting semi-structured interviews with 467 households from 8 villages where the programme operated between December 2009 and January 2010. The questionnaire design was informed by focus group discussions conducted during 2007-2009. Questions focused on respondents' knowledge of the programme, how they thought it operated and who benefited, and whether they considered the rules fair. Interviews lasted about 50 minutes, and were conducted by trained Cambodian social researchers. Anecdotal information on local conflicts over the programme were collected from WCS staff and discussions with other organisations that had replicated the programme in Cambodia.

4.3 Results

Bird Nest Protection Programme: species protected and costs

Over 2,700 nests of eleven Globally Threatened or Near-threatened species were located and protected during 2003-2012 (see Appendix, Table S4.1). Some of the species' populations are of high conservation significance. Minimum population sizes in 2011 in the Northern Plains are estimated at 40 breeding pairs of Giant Ibis (15% of the global population), 5 pairs of White-shouldered Ibis (one of four known nesting sites in mainland Southeast Asia), 50 pairs of Sarus Crane, 250-280 pairs of Lesser Adjutant (equal to the largest known population in Indochina), and 10 pairs of Greater Adjutant (one of two known nesting sites in Southeast Asia). Tables S4.2 and S4.3 in the Appendix provide details of species' differences in the nesting season and choice of nesting site.

The total cost of the programme was around \$26,000 per year in 2005-2008, increasing to \$32,000 from 2008-9 as a consequence of rising prices, particularly for food and transport (Table 4.1). The average cost per nest protected was \$65-\$120. The average cost declined as the number of nests increased, partly because monitoring costs were shared between adjacent sites and because a greater number of nests were found per colony. 71-78% of the total cost went directly to local people, either protectors or community rangers. 22-29% was spent on external oversight of the programme by trained WCS monitoring staff, including nest verification visits and administration of nest protection payments, but excluding higher-level oversight of the programme.

Impact of payments on nesting success and species' populations

The success rate of protected nests was 88.5% during the 2009-11, in comparison with a success rate of 36.9% for unprotected controls of the same species during the same period (Figure 4.1; Table 4.2). The difference in the success rates between the protected and control nests of Lesser Adjutant and Sarus Crane are highly significant (χ^2 = 26.3, d.f. = 1, P < 0.001). Giant Ibises, which were not protected but did have predator exclusion belts installed (Keo et al, 2009), had a success rate of 86.7%, similar to the rate observed in another study in the same area (Keo et al, 2009), and not significantly different from protected nests of Lesser Adjutant and Sarus Crane (χ^2 = 0.01, d.f. = 1, P = 0.914). Of all the protected species, only Greater Adjutant had a moderate nest success rate (68.2%), and this was significantly lower than that for protected nests of Lesser Adjutant (χ^2 = 4.35, d.f. = 1, $P < 0.05$).

Table 4.1. Bird Nest Protection Programme: Costs, 2005-2009. The programme cost $26-31 000 annually, of which 71-78% were payments made to local people, with monitoring costing 22-29%.

	2005-6	2006-7	2007-8	2008-9
Local Payments (%)	$ 20 350 (78%)	$ 19 289 (74%)	$ 19 508 (72%)	$ 22 556 (71%)
Nest Protection Payments	$ 10 425	$ 10 786	$ 10 933	$ 11 890
Community Rangers	$ 9 925	$ 8 503	$ 8 575	$ 10 666
WCS Monitoring (%)	$ 5 603 (22%)	$ 6 630 (26%)	$ 7 474 (28%)	$ 9 375 (29%)
Expenses	$ 2 506	$ 3 470	$ 3 914	$ 5 195
Salaries	$ 3 098	$ 3 160	$ 3 560	$ 4 180
Total	$ 25 953	$ 25 918	$ 26 986	$ 31 930
Nests Protected	217	342	416	360
Average Cost/Nest	$ 120	$ 77	$ 66	$ 89

Table 4.2. Nesting Success Rates during 2009-2011 for unprotected control nests, Giant Ibises (which were not protected by the programme) and for three species that were protected by the programme: Greater Adjutant, Lesser Adjutant and Sarus Crane. Daily nest survival rates were calculated using the programme MARK (Rotella, 2011).

Treatment	Species	Locations / Colonies	Nests	Success	Daily Survival Rate
Controls	All	28	66	36.4%	
	Lesser Adjutants	26	64	37.5%	98.81% ± 0.19%
	Sarus Cranes	2	2	0.0%	92.47% ± 6.01%
Protected	All	256	746	88.5%	
	Lesser Adjutant	64	431	94.4%	99.94% ± 0.01%
	Sarus Crane	96	96	87.5%	99.64% ± 0.10%
	Greater Adjutant	9	22	68.2%	99.71% ± 0.11%
Not Protected	Giant Ibis	60	60	86.7%	99.80% ± 0.07%

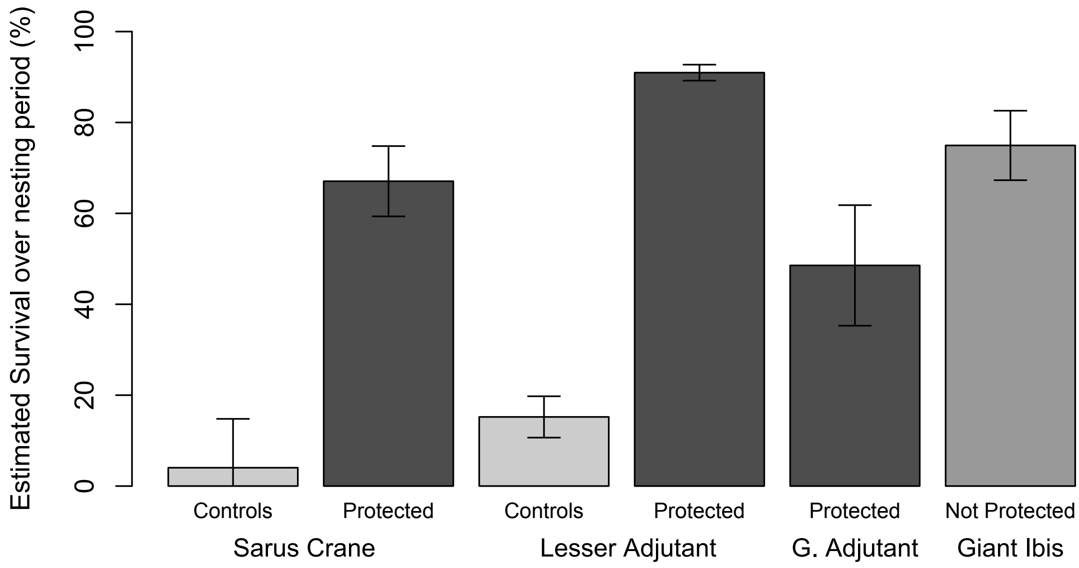


Figure 4.1. Estimated nest survival over the entire nesting period during 2009-2011 for unprotected controls and protected nests of Sarus Crane and Lesser Adjutant, protected nests of Greater Adjutant, and Giant Ibises (which were not protected by the programme). Estimates were calculated using the programme MARK (Rotella, 2011), and standard error bars are given.

Table 4.3. Causes of nest failure for species included in the Bird Nest Protection Programme and Giant Ibises (which were not protected) during 2006-2009 (Period I) and 2009-2011 (Period II), and for unprotected controls during 2009-2011 (Period II). Cases refer to individual nests for solitary species (such as ibises and cranes) and to incidents at nesting sites of colonial species (such as adjutants and darters). For colonies, a recorded incident does not necessarily mean that all nests at the colony failed. Results are given as the percentage of nests or colonies of the species affected, relative to the total number of nests or colonies of that species (in parentheses).

Cause of nest failure		Site & Year	Not protected		Species included in the Protection Programme							
			Giant Ibis (nests)		Sarus Crane (nests)		Lesser Adjutant (colonies)		Greater Adjutant (colonies)		All species (nests/colonies)	
			%	*n*	%	*n*	%	*N*	%	*n*	%	*n*
Human Origin	Human Disturbance *	Protected, Period I	5%	(58)	2%	(148)	3%	(76)	25%	(8)	3%	(271)
		Protected, Period II	2%	(60)	0%	(96)	0%	(52)	14%	(7)	1%	(180)
		Controls, Period II	25%	(4)	0%	(2)	5%	(21)		-	4%	(23)
	Nest Collection	Protected, Period I	0%	(58)	3%	(148)	1%	(76)	0%	(8)	3%	(271)
		Protected, Period II	2%	(60)	2%	(96)	4%	(52)	0%	(7)	2%	(180)
		Controls, Period II	0%	(4)	100%	(2)	76%	(21)		-	78%	(23)
Non-human Origin	Abandoned/ Weather †	Protected, Period I	9%	(58)	7%	(148)	3%	(76)	13%	(8)	6%	(271)
		Protected, Period II	7%	(60)	4%	(96)	6%	(52)	14%	(7)	8%	(180)
		Controls, Period II	0%	(4)	0%	(2)	0%	(21)	-	-	0%	(23)
	Natural Predation	Protected, Period I	0%	(58)	9%	(148)	3%	(76)	25%	(8)	8%	(271)
		Protected, Period II	2%	(60)	5%	(96)	6%	(52)	0%	(7)	6%	(180)
		Controls, Period II	0%	(4)	0%	(2)	0%	(21)	-	-	0%	(23)

* Human disturbance includes cutting of trees, land clearing, domestic dogs, etc.

† Abandoned/Weather includes flooding of Sarus Crane or Oriental Darter nests, loss of nests due to storms, and cases where nests were abandoned with the cause unknown.

For protected nests the most significant cause of nest failure was natural predation by crows, civets and other carnivores, and birds of prey, accounting for 6-8% of incidents over five years and over 120 nests in total (Table 4.3). A further 6-8% of nests or nesting colonies were accidentally lost due to wind, rain, flooding of Sarus Crane breeding sites, or chicks falling from trees (Table 4.3). It is possible that some of these nests may have been collected. Human disturbance, land clearance or tree cutting accounted for up to 3% of nest or colony failures, and eggs or chicks were collected from a further 3% whilst the protectors were absent. Similar causes of nesting failure were recorded for Giant Ibis, which was not protected, with the exception of natural predation, which was significantly reduced through the use of predator exclusion belts (Keo et al., 2009). By contrast, 77% of 22 unprotected (control) nests or colonies were harvested for eggs and chicks, and the trees used by one Adjutant colony were logged (Table 4.3). Of the protected species, only Greater Adjutant colonies had high rates of failure due to human causes (14-25%; Table 4.3).

The numbers of nests recorded by observers changed considerably between 2004-5 and 2011-12 for most species (Figure 4.2, Table 4.4). During this period survey effort declined by about 20% from approximately 2,400km^2 to 1,900km^2, suggesting that the recorded changes in nest numbers were not due to increased survey effort, although it is possible they were caused by changes in detectability. However, the fact the same group of observers recorded some species increasing significantly, whilst observing static or declining trends for other species breeding at the same time in the same habitats, suggests that the results indicate relative trends rather than simply observer bias. Survey coverage was lowest in the 2008-9 season, when surveys started a month later than usual and after some Giant Ibises had finished nesting. Data for the 2010-11 season were omitted because the onset of the wet season was considerably delayed, so most species started nesting 1-2

Table 4.4. Generalised linear model for trends in the number of breeding pairs recorded for each species at the two sites, using quasi-poisson errors and a log link function. Significance values for the null hypothesis of zero effect: ns = not-significant, † = $P < 0.1$, *** = $P < 0.001$).

Variable and Coefficient	Degrees of Freedom	F-value (significance)	Coefficient (significance)
Site	2	1055.7 ***	
Species	4	68.6 ***	
Species * Year	5	8.3 ***	
- Lesser Adjutant (protected)			0.106 ***
- Sarus Crane (protected)			0.110 †
- Greater Adjutant (protected)			-0.240 †
- Oriental Darter (protected)			0.343 ***
- Giant Ibis (not protected)			0.035 ns
Residual deviance = 296.5 on 46 degrees of freedom			

months later than normal, and numbers of all early nesting species (Giant Ibis, Lesser Adjutant and Sarus Crane) were low. Breeding populations, calculated as the number of nests observed, of Lesser Adjutant and Oriental Darter increased significantly through the study period (Table 4.4, $P < 0.001$ in both cases), and there was some evidence for increases in Sarus Cranes (Table 4.4, $P = 0.088$). The observed population increases for Lesser Adjutants, Oriental Darters and Sarus Cranes are consistent with internal recruitment, based upon what is known about the age at which birds reach sexual maturity (del Hoyo et al., 1996). Successful breeding by Sarus Cranes in the Northern Plains may account for the growing number of birds seen since 2007 at dry season feeding sites elsewhere (Evans et al., 2008). By contrast, there is no evidence for changes in the numbers of Giant Ibis, which was not impacted by nest collection (Table 4.4, $P = 0.644$), implying that other factors, such as natural predation and conversion of feeding habitats to agriculture, are the primary threats to this species (An, 2008; Keo, 2008). There was some evidence for population decreases in Greater Adjutants (Table 4.4, $P = 0.059$), probably due to a combination of disturbance of feeding sites, poisoning, and cutting of nesting trees. On several occasions the main colony at Antil village was deliberately disturbed, before the nest protectors arrived, by land grabbers who did not want the presence of a breeding colony to draw attention to their activities. The birds moved to another site but in diminished numbers.

Social impacts of the programme

The programme benefits about 100 households each year, of the approximately 4,000 households across the 24 villages where the programme operates. In the majority of villages, <5% of households were engaged in the programme (Figure 4.3a), although in a few villages up to 33% of households were involved. The majority of villages received <$750 per year, but with some villages earning >$2000 per year (Figure 4.3b). Total payments varied depending upon the number of key species present, or species with particularly long breeding periods. Antil village received the greatest amount, with >$14,000 of payments over the four years, mainly due to the presence of the Greater Adjutant colony, which requires at least 6 months of protection each year. The average payment per nest protector was $80-$160, but there was considerable variation in the payments made, depending upon the species protected (as different species needed protecting for different periods of time, Figure 4.3c). Some individuals were specialist protectors, switching species depending on the season and receiving continual employment for several months. Community rangers received

Figure 4.2. Breeding bird populations in the Northern Plains, for (a) Lesser Adjutant; (b) Sarus Crane; (c) Greater Adjutant, and (d) Oriental Darter, all of which were targeted by the payment programme; and (e) Giant Ibis, the only species which was not protected. The predicted values and 95% confidence intervals for the best fitting generalised linear model of nest numbers are also shown (see Table 4 for details). Survey coverage (final panel) was constant or declined slightly during the study period, suggesting that observed increases in species populations were not due to greater survey effort.

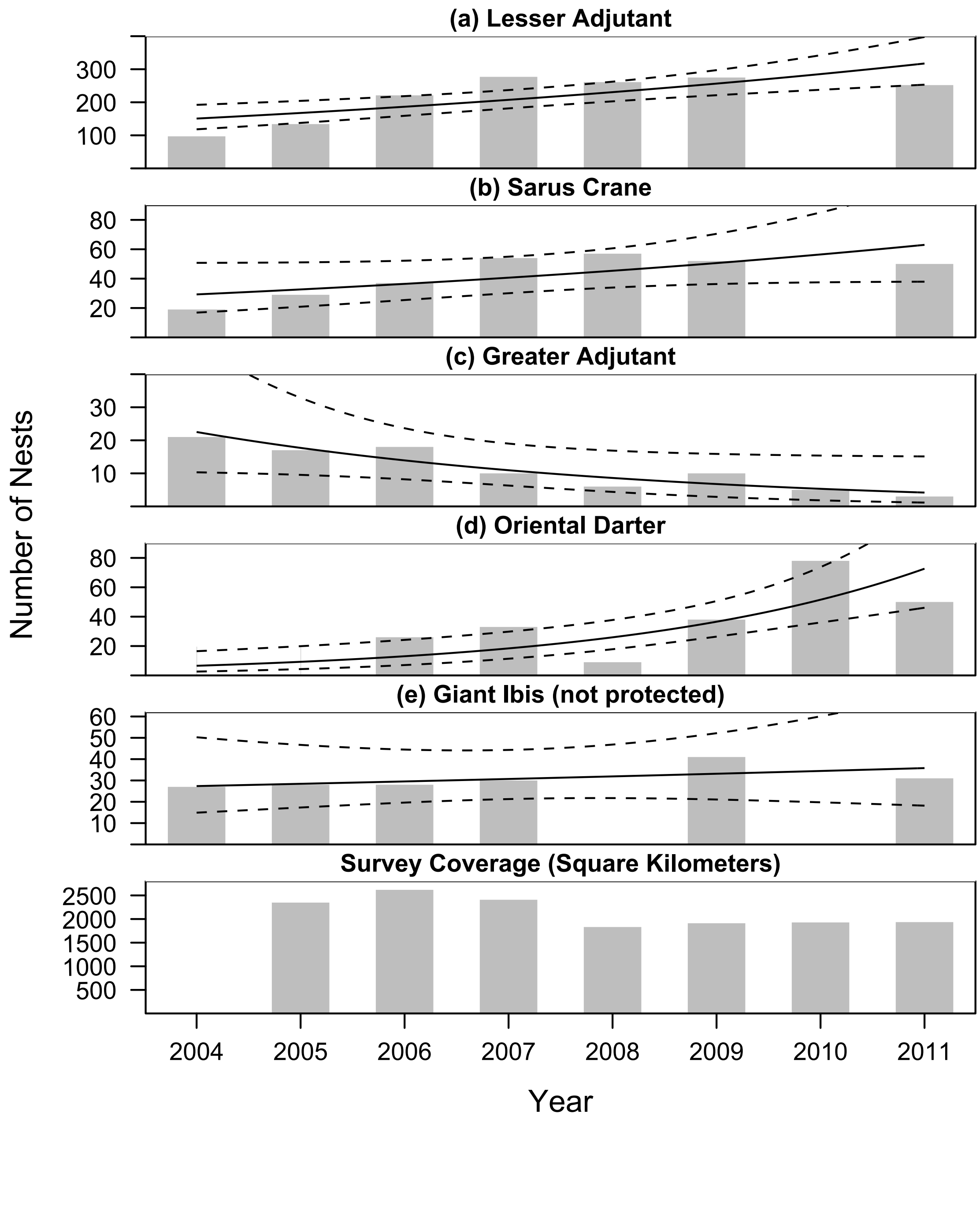
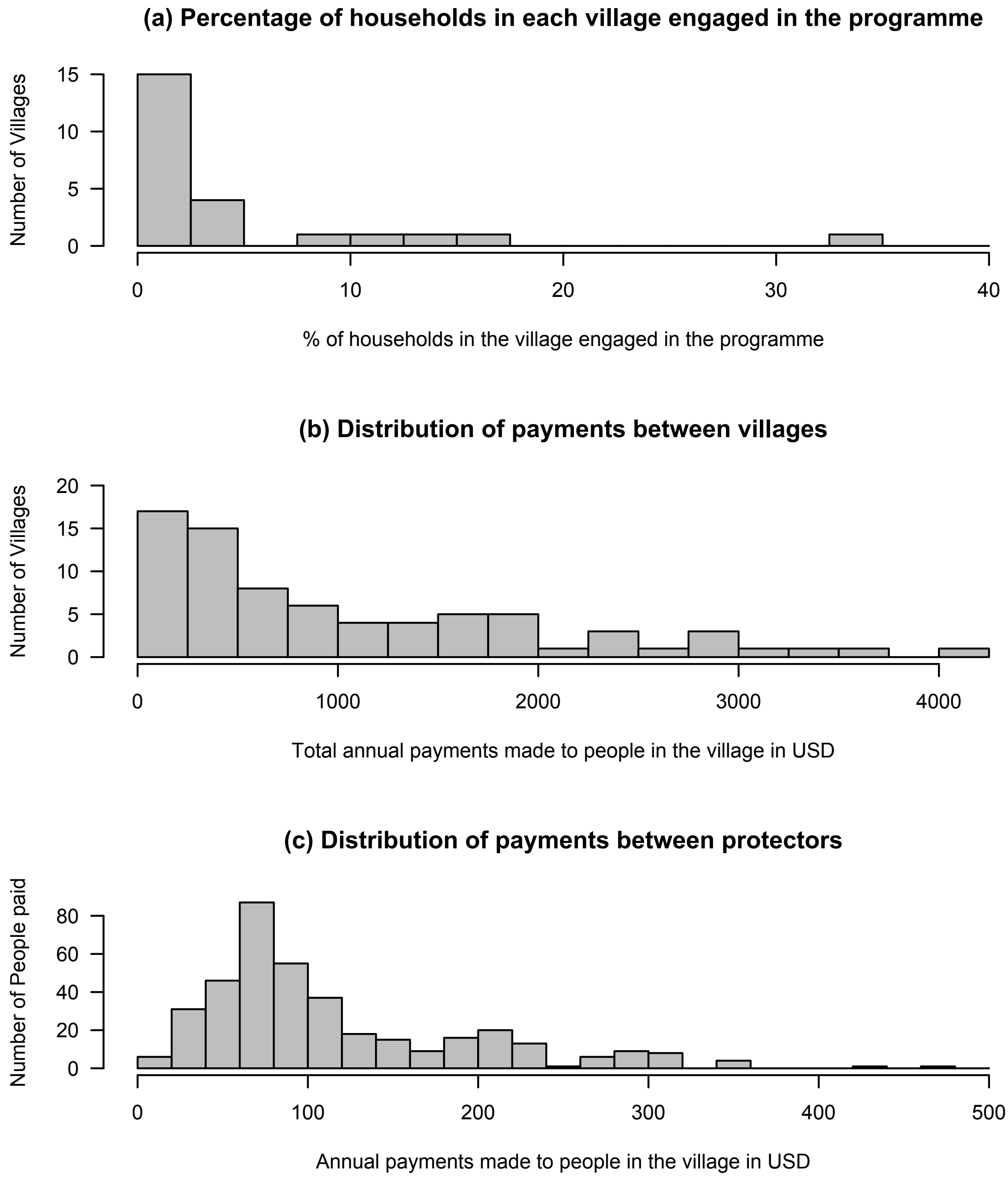


Figure 4.3. Who benefited from nest protection payments between 2005 and 2009? The histograms show (a) the percentage of households in each village receiving payments; (b) distribution of the total annual payments received by people in each village; and (c) distribution of annual payments to individual nest protectors. Histogram (c) has peaks at $80, for nest protectors that were employed protecting a single species over two months or less, $160 for protecting Lesser Adjutants (3-4 months) and $300 for protecting Greater Adjutants (up to 6 months).



significantly more, averaging $500-$800 per year with a maximum of >$1,200. The distribution of payments is therefore quite uneven both between and within the villages, with only a small number of people generating high incomes from nest protection. The average payment per protector is significant in comparison with the 2009 estimate of household consumption in rural forested regions from the 2007 Cambodia Socio-Economic-Survey of $329±16 (World Bank, 2009).

Despite the uneven distribution of benefits and the small number of people involved, 67% of 467 households interviewed were familiar with the programme and could accurately describe how it worked. Of these, the vast majority thought that the distribution of benefits was fair (95%, Table 4.5), and understood that the primary beneficiaries were individual households (93%). There was no suggestion that traditional rules existed regarding the management of birds, or that these might have been crowded out by the initiation of the programme. In villages where a moderate percentage of people (c.10%) were engaged in the programme, respondents thought that it benefited the village as a whole (67%), whereas in villages with limited involvement in the programme fewer respondents thought the village benefited (28%). Most people correctly saw the programme as being directly managed by WCS, especially in those villages with high involvement in the programme (71%), rather than by local people. Even so, it was universally understood that anyone could participate (100%). Participating households were similar to non-participants in most characteristics, with the exception of a slight bias towards male-headed households (Table 4.5). Despite this overall positive assessment of the programme, conflicts over who should receive payments and jealousy regarding the amounts paid were observed, particularly in Antil village (where people were paid for up to six months to protect Greater Adjutants). This type of resentment was also observed by other organisations piloting the same approach at other sites in Cambodia (WWF, pers. comm.). Antil differed from the other villages, because non-participants overwhelmingly saw the programme as providing no benefits to the village (76%), suggesting a substantial level of local disquiet.

Table 4.5. Attitudes of local people towards the bird nests protection programme, based on a sample of 467 households interviewed across 8 villages, 5 of which were regularly involved in the programme, and 3 of which had only limited involvement. 67% of respondents (315 households) were aware of the programme and could describe broadly how it worked, data are based on responses from these interviews.

			Involvement in Bird Nests Programme			
			Limited		Regular	
Variable	Question	Response	Result	%	Result	%
Villages interviewed						
Number of Villages			3		5	
Average Village Population (2008, Households)			116		146	146
Average number of Households engaged in the Programme/village/year		Yes (%)	1	(1%)	13	(10%)
Annual average value of payments made per village/year		$/village/year	$87		$2,103	
Aware of the programme (n = 467)		Yes (%)	76	(47%)	239	(78%)
For the 315 households that are aware of the programme:						
Existence of prior rules?	Can describe traditional rules regarding birds?	Yes (%)	0	(0%)	0	(0%)
Knowledge of the Programme	Can describe the conditions (to protect birds)?	Yes (%)	70	(92%)	221	(92%)
Household beneficiaries	Benefit directly from the programme?	Yes (%)	6	(8%)	62	(26%)
	Female-headed households? (divorced, widowed or single)	Beneficiaries (%)	0	(0%)	1	(2%)
		Non-Beneficiaries (%)	2	(3%)	13	(7%)
	Average Age of household head (years)	Beneficiaries	36		41	
		Non-Beneficiaries	41		41	
Perceptions?	Who manages the programme?	Village Authority? (%)	0	(0%)	0	(0%)
		Villagers? (%)	15	(20%)	54	(23%)
		WCS? (%)	47	(62%)	169	(71%)
	Who can participate?	Anyone? (%)	76	(100%)	239	(100%)
	Is the programme fair?	Yes (%)	73	(96%)	225	(94%)
Who benefits?	Village Authority?	Yes (%)	0	(0%)	0	(0%)
	WCS or WCS's friends?	Yes (%)	0	(0%)	12	(6%)
	Individual households?	Yes (%)	71	(93%)	222	(93%)
	Village?	Benefit a lot (%)	21	(28%)	160	(67%)
		No benefit (%)	54	(71%)	71	(30%)
		Lose out (%)	1	(1%)	8	(3%)

4.4 Discussion

The effectiveness of direct payments as a conservation intervention

Direct payments for conservation, and results-based incentive mechanisms in general, such as PES and REDD, have received considerable attention over the past decade, and a large number of such programmes exist in both marine and terrestrial environments in both developed and developing countries (Milne & Niesten, 2009; Pattanayak et al., 2010), including widespread use of payments for nesting turtles (Ferraro & Gjertsen, 2009) and birds (Verhulst et al., 2007). The Bird Nests protection programme analysed in this paper is consistent with Ferraro's definition of a direct payment programme, and Wunder's (2007) strict definition of PES. Proponents have argued that direct payments may provide an effective mechanism to deliver biodiversity conservation outcomes, in a way that also provides potentially significant contributions to local livelihoods (Ferraro & Kiss, 2002). The Bird Nests Protection programme meets many of these claims concerning the effectiveness, costs, and development benefits of payment programmes.

The evidence suggests that nest protection payments were an effective way to ensure that large numbers of globally threatened birds that were threatened by nest collection successfully bred in the Northern Plains. Leakage (displacement of bird harvesting activity to other sites) is unlikely to have occurred due to the large distances involved: villagers would have had to move significant distances (>10 km) to find unprotected bird populations. As a consequence of the programme, populations of some of these species may have increased considerably based upon the population data presented. However, the success of a targeted results-based payments programme depends upon the extent to which the outcome that is rewarded (nest protection) accurately reflects biodiversity conservation needs (Redford & Adams, 2009; Gibbons et al., 2011). Payments had limited impact on species such as the ibises and Greater Adjutants, for which the main threats to nesting birds were natural predation and habitat clearance by villagers or outsiders (An, 2008; Keo, 2008; Keo et al., 2009; Wright 2012; Wright et al., 2012). Protectors were unable to prevent any of these threats. This emphasises the importance of designing conservation interventions based on clear conceptual models of threats to biodiversity, how interventions affect these threats, and the resulting impacts of interventions on conservation targets (Salafsky et al., 2002; Margoluis et al., 2009). When the programme was designed in 2003, nest collection was the greatest threat to breeding bird populations. Since 2006, deforestation rates have increased considerably in Cambodia and in the study area (Forestry Administration, 2011), and the failure of the payment programme to incentivise habitat protection raises considerable concerns about its long-term effectiveness.

The nest protection programme was relatively inexpensive in comparison with the costs of other types of conservation interventions such as protected area management (James et al., 2001) or integrated conservation and development projects (Wells et al., 1999). The majority of funds went to local people. This substantiates theoretical claims that direct payment programmes would have low administrative costs, and would provide significant benefits at the local level (Ferraro & Kiss, 2002). The payments provided a legal income from the birds instead of illegal hunting and trade. Payment amounts were highly significant in poor remote rural villages relative to other sources of income, suggesting that they made a contribution to local livelihoods.

In conclusion, the bird nests protection programme was a highly effective conservation intervention to protect highly threatened globally significant biodiversity, in a way that was rapid to establish, cost-efficient and delivered significant benefits to participants. The sustainability of user-financed direct payments programmes, such as this one, are however a concern since they are reliant upon continual funding (Swart, 2003). If the payments ceased it is possible that some of the nest protectors and local rangers (many of whom had previously been well-known hunters) would return to nest collection.

Social acceptance of external payments: equity and fairness

The extent to which payments are socially-appropriate, equitable, fair, or designed to build local support for conservation is often not an explicit consideration in the design of PES programmes (Jack et al., 2008; Pascual et al., 2010). Critics have raised concerns that payments may ‘crowd-out’ local social norms, monetising behaviours and outcomes that may previously have had non-monetary local values (Bowles, 2008; Redford & Adams, 2009; Clements, 2010). There was no evidence in the bird nests case that prior social rules existed regarding management of breeding bird populations, or that these were crowded out by introduction of the payments.

Brown and Corbera (2003) distinguish between three elements of equity in PES programmes: equity in access, equity in decision-making and equity in benefits. The bird nests protection programme scores highly against only one of these three criteria. The programme was designed to be, and recognised by local people as, open to participation by anyone from the local villages. Local people were not, however, involved in any aspects of decision-making, as the programme was administered externally by WCS staff. Externally-imposed rules and incentives may lead to perceptions that incentives are unfair (Fehr & Falk 2002). Finally, the distribution of benefits was highly inequitable both between and within villages: only a small number of households in the villages benefited, and even fewer received high payments.

Researchers have suggested that there is a trade-off in programme design between efficiency, in terms of the cost for protecting biodiversity, and equity in the distribution of benefits (Proctor et al., 2009; Pascual et al., 2010). Payment programmes could be designed to be more egalitarian but that this would be less cost-efficient as the payments are likely to be less precisely targeted to those able to deliver conservation outcomes. Pascual et al. (2010) propose a range of fairness criteria for distribution of benefits from PES programmes, from simple compensation based on the costs of providing the ecosystem service, to pro-poor payments that aim to maximise net benefits to the poor, even at a cost of efficiency loss. The distribution of benefits under the bird nest protection programme was at the compensation end of this spectrum, rewarding protectors based upon the opportunity cost of their labour to protect the nests (their minimum willingness to accept).

Despite the uneven distribution of payments, however, the programme had broad support across all the villages, was generally seen to benefit the village as a whole, and was overwhelmingly viewed as fair. This is probably explained by three observations. Firstly, protectors were generally chosen from local forest users or farmers, who had the strongest claims to ownership of the area in the absence of property rights. Secondly, the payment levels were based on the number of days worked, with the daily rate based on an acceptable local wage. Differential payments are seen as fair so long as the payment level is commensurate with effort (Konow, 2003). Thirdly, in Cambodia international non-government organisations, such as WCS, commonly provide services usually provided by the state and tend to be viewed positively as service providers (Malena & Chhim, 2009). The evidence suggests that the bird nests protection programme was administered correctly: there was very little evidence for elite capture, or the programme being seen to disproportionately benefit 'friends' of WCS.

Nevertheless, interview reports suggested that a small minority of local people did not support payments, perhaps due to the uneven distribution of benefits. The level of local disquiet was greatest in the village where payments were made for Greater Adjutants, which were the most valuable species in the programme (due to their long nesting time), but where few people benefited to a great degree due to the small number of nesting sites. As a consequence, the Greater Adjutant colonies were not effectively protected by the programme and have continued to decline.

Design, implementation and evaluation of payment programmes

The last two decades have seen a rapid expansion of policy approaches that provide conditional incentives for provision of social and environmental services in developing countries, such as Conditional Cash Transfers (Fiszbein & Schady, 2009), Direct Payments for Biodiversity Conservation, and Payments for Environmental Services (PES), including REDD, with billions of dollars spent on such programmes globally (Fiszbein & Schady, 2009; Diaz et al, 2011). Whereas Conditional Cash Transfer programmes have incorporated and facilitated rigorous impact evaluations as part of their implementation, most PES or direct payment programmes have not been subjected to the same standards (Pattanayak et al., 2010). Consequently, a recent PES review concluded "we do not yet fully understand either the conditions under which PES has positive environmental and socioeconomic impacts or its cost effectiveness" (Pattanayak et al., 2010).

This study has shown that it is possible to incorporate rigorous impact evaluation into the implementation of a direct payment for biodiversity conservation programme. To be effective, evaluation needs to consider at least three aspects: (*1*) the details of programme implementation, including the cost and distribution of payments made, (*2*) the impact of payments on the conservation threats they were designed to address; and (*3*) the impact of the payment programme on conservation targets, such as increases in species populations (Wilkie, 2004). Such comprehensive evaluation at multiple levels is important in the context of complex socio-ecological systems, where it is challenging to separate out the impact of a single intervention. Experimental or quasi-experimental techniques are necessary in order to assign causation to conservation interventions in the context of other processes (Ferraro & Pattanayak, 2006).

Implementation of payment programmes in the context of weak institutional frameworks and unclear property rights creates significant challenges (Muradian et al., 2010). This study has shown how a simple direct payments programme implemented by an external agency, targeting only a single metric (nest success), without explicit consideration of the distribution of benefits or other social issues, can be extremely successful in conservation terms and also deliver important benefits to participants. It has also demonstrated two potential pitfalls with such a programme design: (*1*) targeting a single conservation metric is risky when there are multiple changing threats to species' populations; and (*2*) compensating individuals directly for species protection, ignoring issues of equitability, may lead to unintended consequences. Most significantly, in the context of rapid land-use change, weak institutions and unclear property rights over land and natural resources, this type of programme is best viewed as a complement, not a substitute, to other types of interventions, including protected area management, local management of natural resources, and development of sustainable financing (Clements et al., 2010).

Chapter 5. Impacts of Protected Areas and Payments for Environmental Services on deforestation in northern Cambodia

Abstract

Global efforts to reduce deforestation require accurate information regarding the effectiveness of forest protection policies, if these efforts are to be successful. This study investigated the effect of Protected Areas (PAs) and Payments for Environmental Services (PES) at reducing deforestation rates in northern Cambodia, using both impact evaluation methods and an analysis of the drivers of deforestation. PAs significantly reduced deforestation rates, even after accounting for the non-random placement of the PAs, at relatively low cost. PES, implemented within the same PAs, further reduced deforestation rates by about 50-100% over that achieved solely by PA management, albeit at much greater cost. The two different policies had complementary effects on drivers of deforestation: PAs deterred external drivers (large-scale development, population growth), whilst PES encouraged local people living within PAs to reduce land clearing behaviours. The PES programmes would have failed without the security of forestland tenure afforded by the PA. Command-and-control approaches and incentive-based mechanisms should therefore be viewed as complementary, rather than competing, policies to reduce forest loss.

5.1 Introduction

Understanding how forest conservation policies affect drivers and rates of deforestation and forest degradation, and whether these impacts are additional (more than might have been expected if the policies had not been implemented), is essential to efforts to mitigate global climate change and protect biodiversity (UNFCCC, 2010; Clements, 2010). Policy implementation also needs to be costed in order to determine the level of investment required to achieve global policy goals. Protected Areas (PAs) are one of the most widely adopted conservation policies, currently covering over 12 percent of the terrestrial area of developing countries (UNEP-WCMC, 2012), against a global target of 17 percent agreed by the Convention on Biological Diversity (CBD, 2010). More than a dozen impact evaluation studies, at landscape, national or global scales, have shown that PAs do protect forests more than might have been expected in the counterfactual case (reviews by Albers & Ferraro, 2006; Joppa & Pfaff, 2010; Ferraro et al., 2012). These studies show that the calculated effect of PAs, when compared to an appropriate counterfactual, is often lower than might have been concluded from a simple comparison (for example with adjacent areas), due to nonrandom factors that influence PA placement. PAs, for example, may be located in more remote areas away from roads and population centres, or at higher elevations, or on soil types that are marginal for agriculture (Andam et al., 2008; Pfaff et al., 2009), all of which would be expected to have lower deforestation rates regardless of the PA intervention itself.

Payments for Environmental Services (PES) has been recently proposed as a policy that recognises the public values of ecosystem functions, and attempts to internalise these within economic markets (Wunder, 2007). The popularity of PES is to some extent linked to the increasing use of conditionality and performance measures to distribute aid and subsidies (e.g. conditional cash transfers, Fiszbein & Schady, 2009). In economic theory, PES incentives are supposed to be conditional upon the beneficiaries delivering a particular environmental outcome, such as forest protection. In practice, PES programmes face a plethora of political, institutional, design and governance challenges, especially in developing countries, which may moderate the extent to which payments are targeted appropriately and made directly conditional upon environmental performance (Clements et al., 2010; Ferraro et al., 2012). Wunder et al. (2008) further suggest that PES may not be cost-effective in government-financed cases because side objectives, such as poverty alleviation, will affect programme design; by contrast they suggest that user-financed PES programmes may be more effective. Perhaps unsurprisingly, therefore, the few impact evaluation studies that have been completed for government-financed PES have suggested that these programmes have had limited additional environmental impact (Pattanayak et al., 2010). Ferraro et al. (2012) state that no impact evaluations of user-financed PES have yet been completed (Chapter 4 and Clements et al. (2012b) give a subsequent example). The costs of different policies are rarely compared in the same landscape.

The drivers of deforestation and forest degradation, and the potential effects of policy on these drivers, have been extensively researched (Angelsen & Kaimowitz, 1999; Geist & Lambin, 2002; Angelsen, 2010). Studies of drivers can be disaggregated into understanding the characteristics and motivations of deforestation agents, the proximate causes of deforestation due to the behaviour of these agents, and the underlying factors that influence their decision-making. Proximate causes of deforestation include infrastructure development, such as building roads, settlements or mining; agriculture expansion, either permanent or shifting, by residents or immigrants and by individuals or companies; and wood extraction, again either for local or commercial uses (Geist & Lambin, 2002; Wright et al., 2007; Meyfroidt & Lambin, 2009; Gibbs et al., 2010). Environmental factors, such as soil type or topography, affect the magnitude of these proximate causes. Underlying causes include demographic factors; economic factors such as prices and markets; technological factors such as mechanised agriculture; policy and institutional factors, including property rights and policies designed to reduce deforestation; and cultural factors (Geist & Lambin, 2002). Interactions frequently occur between multiple causal factors. Due to the complexity and multi-scale nature of these interactions the same policy prescription can lead to different outcomes at different sites, depending upon the agents of deforestation and their response to interventions (Angelsen, 2010). PAs might be expected to constrain the proximate drivers of deforestation, potentially imposing significant costs on the agents of deforestation (such as local people), whilst at the same time being vulnerable to wider national-level policy trends. PES increases the value of maintaining forests to those local people who are able to influence deforestation rates, potentially leading to a change in their behaviour if the incentives outweigh the returns from forest clearance.

In this study, impact evaluation methods were used to quantify the impact of PAs and PES on deforestation rates over an eight year period (2002-2010) in the northern forests of Cambodia. Northern Cambodia was an ideal location to test the impacts of PAs and PES because the interventions were initiated relatively recently, thereby allowing before-after comparisons to be made; the forested areas are extensive, allowing the selection of appropriate controls that were not affected by the PA and PES interventions; and because the PES programmes had been well documented (Chapter 2; Clements et al., 2010). The objectives of this study were to: (*1*) quantify the overall impacts of PAs and PES on deforestation rates in the landscape; (*2*) identify the local agents of deforestation and how PAs and PES affected the proximate drivers of deforestation; and (*3*) to investigate the underlying causes of deforestation and how these were affected by PAs and PES. The study focused on both internal drivers, principally the actions of local people, and external drivers caused by in-migration and large-scale expropriation of forested lands by concessions. From the mid-2000s, concessions have been declared across Cambodia either for economic development (economic land concessions) or resettling of people (social land concessions), and have become a key driver of deforestation (Cambodia R-PP, 2011).

5.2 Methods

Protected Areas and Payments for Environmental Services interventions

The study focused on the core management zones of two PAs in northern Cambodia (see Chapter 2, Figure 2.1): Kulen Promtep Wildlife Sanctuary (gazetted 1993) and Preah Vihear Protected Forest (gazetted 2002). Both PAs were paper parks until active management started in 2005. Villages were permitted by PA authorities to expand agriculture to a limited extent within agreed land-use plans. Initially, large-scale concessions could not be legally declared within PAs, a restriction that was relaxed for National Parks and Wildlife Sanctuaries (such as Kulen Promtep) by the 2008 Protected Areas Law, but not for Protected Forests (such as Preah Vihear). Two PES programmes were instituted from 2005 onwards in four of the villages inside the PAs with the goal of reducing local forest clearance as a complement to PA management: (*1*) community-managed ecotourism linked to wildlife and habitat protection; and (*2*) providing premium prices for agricultural goods to households that kept to the land-use plans (Ibis Rice; Chapter 2; Clements et al., 2010).

Selection of control villages

The impact evaluation focused on results at the village level, because most land clearance was focused around existing villages and that was the scale at which the PES interventions were implemented. Matching methods were used to select seven appropriate control villages >20km outside the PAs against which to measure the impacts of the interventions (see Chapter 2). A further sample of nine border villages were selected from the area within a 10 km buffer of the two PAs (see Chapter 2, Figure 2.1). This allowed the comparison of impacts inside PAs with nearby areas, which is the standard comparison made by many studies that do not use matching methods.

Deforestation rate analysis

Deforestation in the landscape was mapped between 2001/2 and 2005/6, the four years immediately prior to establishment of the PAs, and between 2005/6 and 2009/10, the subsequent four years when the PAs were actively managed and the PES programmes were being implemented. Forest and non-forest areas within 8 km of the selected villages were identified using high resolution aerial photographs and medium spatial resolution remote-sensing data (see Chapter 2 for details). The forest cover maps were resampled using a 1 km square grid, to give estimates of the number of hectares of deforestation in the two time periods in each of the 1 km grid squares. 1 km was judged an appropriate size for the grid given the resolution of the data (1 hectare). Only squares with complete forest cover maps were used. For the analysis of the impacts of PES within the PAs, squares were assigned a treatment type depending upon whether they were in the PA and within 5 km of one of the four PES villages (n = 217), or were in the PA and within 5 km of another village within the PA that was not receiving payments (n = 433). The radius of 5 km was chosen because mapping of household agriculture indicated that this was the maximum distance local people travelled to establish fields. This also allowed the separation of effects due to different villages within the PAs, where the minimum distance between two villages was 10 km. For the analysis of PA impacts a wider radius of 8 km was used in order to incorporate some grid squares that were too far from the villages to be viable for agriculture. Grid squares were then assigned a treatment type depending upon whether they were both within a PA and within 8 km of a village inside a PA (n = 1356), were within 8 km of a control village (n = 913) or were within 8 km of a village bordering a PA but completely outside the PA itself (n = 1035).

Matching estimators were used to compare deforestation rates across treatments, based on matching of squares subject to the PA or PES interventions with similar squares in the other treatments. Spatial autocorrelation was adjusted for by clustering the data according to the closest village using the equations developed by Hanson and Sunderam (2011). Matching was undertaken using similar procedures as for the village selection in order to control for observed characteristics that might influence deforestation rates between squares, ensuring that the comparisons made were as similar as possible. The matching variables selected were the base area of forest in 2001/02 (for the first four year period) or 2005/06 (for the second four year period), the distance to nearest village and the slope. Elevation, which has been used in similar analyses (Joppa & Pfaff, 2010), was considered but discarded because it was highly correlated with slope (the study area had little topographic variation).

Matching was conducted six times, to compare the PA squares to the controls and border areas in each of the two time periods, and to compare areas within PAs that were affected by PES interventions and areas within PAs that were not affected by PES interventions. The matching procedure used was similar to that followed for the village matching (see Chapter 2). One match was found for each grid square inside the PAs or affected by the PES interventions. Callipers were used where necessary in order to minimise differences between the samples and achieve a good match. Balancing statistics and tests were used to check that balance had been achieved in the matched samples, and the matching was re-run if the samples were significantly different in two or more characteristics. The balancing statistics and tests for the two most important comparisons are given in Table 5.1: (a) comparing PAs and the Controls during the second period (2005/6 to 2009/10) and (b) comparing PES areas with non-PES areas within PAs for the second period (2005/6 to 2009/10). The clustering effect of village was accounted for using the equations developed by Hansom and Sunderam (2011).

Table 5.1. Balancing statistics and tests for covariate matching for the unmatched and matched samples of squares for (a) Protected Areas and Controls, and (b) areas affected by PES interventions and areas that were not. The matching process ensured that the differences between the PA and Control squares for the matched sample were minimised. Statistics calculated included the means for each group; the mean, median and maximum difference in the empirical cumulative distribution function (eCDF); the mean, median and maximum difference in the empirical quantile-quantile (eQQ) plot of treatment and control groups on the scale in which the variable was measured; the variance ratio of treatment over control (which should equal 1 if there is perfect balance); *t*-tests comparing the samples before and after matching (the two sample *t*-test was used pre-matching and the paired *t*-test was used post-matching); and the bootstrap Kolmogorov-Smirnov (KS) test, which tests for a significant difference across the entire distribution (as indicated by the eQQ plots).

(a) Protected Areas and Controls

Variable	Slope (degrees)		Distance to nearest village (km)		Forest Cover in 2005/6 (hectares)	
Statistic	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched
Mean PA squares	1.478	1.396	4.508	4.496	95.451	95.424
Mean Control squares	1.914	1.400	4.232	4.493	94.978	95.491
Std Deviation Mean diff	-32.093	-0.548	15.227	0.170	4.060	-0.581
Mean raw eQQ diff	0.517	0.020	0.274	0.032	0.771	0.203
Median raw eQQ diff	0.290	0.010	0.284	0.028	0.075	0.000
Max raw eQQ diff	8.783	2.394	0.432	0.302	9.429	17.757
Mean eCDF diff	0.140	0.007	0.042	0.005	0.058	0.007
Median eCDF diff	0.154	0.007	0.048	0.004	0.032	0.004
Max eCDF diff	0.218	0.029	0.064	0.018	0.159	0.035
Variance ratio (Treatment/Control)	1.099	1.108	0.943	0.995	0.981	1.036
t-test *p*-value	<0.001	0.243	<0.001	0.539	0.346	0.039
KS Bootstrap *p*-value	<0.001	0.609	0.016	0.978	<0.001	0.218

(b) PES areas and non-PES areas, all within Protected Areas

Variable	Slope (degrees)		Distance to nearest village (km)		Forest Cover in 2005/6 (hectares)	
Statistic	Unmatched	Matched	Unmatched	Matched	Unmatched	Matched
Mean PES squares	1.338	1.255	2.809	3.094	87.243	95.334
Mean non-PES squares	1.335	1.258	2.708	3.076	92.730	95.268
Std Deviation Mean diff	0.676	-0.786	9.871	1.966	-31.870	0.539
Mean raw eQQ diff	0.070	0.013	0.118	0.036	5.420	0.247
Median raw eQQ diff	0.065	0.009	0.116	0.031	3.568	0.000
Max raw eQQ diff	0.850	0.067	0.302	0.166	16.259	3.076
Mean eCDF diff	0.048	0.010	0.030	0.013	0.139	0.015
Median eCDF diff	0.042	0.008	0.031	0.008	0.151	0.008
Max eCDF diff	0.131	0.041	0.068	0.049	0.228	0.049
Variance ratio (Treatment/Control)	1.298	0.995	0.890	0.974	1.420	1.008
t-test *p*-value	0.933	0.392	0.245	0.027	<0.001	0.508
KS Bootstrap *p*-value	0.011	1.000	0.486	0.998	<0.001	0.973

Understanding proximate drivers of deforestation

In 2009-2010 teams of local data collectors both inside the PAs and in the control areas conducted surveys around the villages for 10 days each month to record any occurrences of deforestation or forest degradation (such as forest clearance or logging). For each incident, data were recorded on who the actors were and where they came from. Local data collectors carried Global Positioning System (GPS) devices to record the locations of the observations, and were asked to record one point every 30 minutes regardless of what was seen, in order to estimate the area surveyed. Data collectors were local people and were not linked in any way to the PA management. It is possible that they might have under-reported activities around their own villages, however this potential source of bias was accounted for by asking teams to survey areas around each other's villages. Survey coverage was estimated as the number of kilometre squares visited during these surveys, using the same grid squares used for the deforestation rate analysis with the same radius (5 km for the PES impact evaluation, 8 km for the PA impact evaluation). Any grid squares visited (and observed human activities) outside of these radii were excluded from the analysis, to ensure that results were from exactly the same area as the data on deforestation rates, and because the prevalence of human activities would probably be closely related to distance from the village (i.e. grid squares far from villages and close to villages would not be comparable). Human activity data was expressed as a binary variable: whether or not an incident was observed during the visit to the grid square in that month. Participatory rural appraisal discussions with focus groups indicated significant differences in local behaviour through the year, hence the particular month was known to be important.

Data were analysed using mixed effects models with a binomial link function (logistic regression) in package "lme4" in R 2.14.2 (Bates et al., 2011) to investigate the effect of the intervention (PA or PES) on whether or not a human activity was observed during that visit. Models included the month the square was visited, and the random effect of village (as 1 km grid squares were clustered around villages) and the square itself (as each square was visited multiple times). It was not possible to analyse this data using matching methods (as was used for the equivalent data on deforestation rates from the same 1 km grid squares) because the clustering equations developed by Hanson and Sunderam (2011) only allow for one clustering variable not two (village and square).

Understanding underlying causes of deforestation

The analysis focused on internal drivers (population growth) and external drivers (immigration and land concessions).

Population growth. The number of households in each village during 2001-2011 was obtained from the Cambodia Commune Database (NCCD, 2011), which is updated annually based upon reports from village chiefs. These numbers were verified where possible by visiting the villages to review the formal 'village household list', which is maintained by the village chief. Verification visits were conducted for all villages in all years between 2008 and 2011, and in many cases historical lists kept by village chiefs enabled verification back to 2004 or even earlier. Data were analysed using mixed effects models in the package "nlme" in R 2.14.2 (Pinheiro et al., 2011), with the dependent variable as the number of households in each village square-root transformed.

Immigration and new villages. Details of immigrants that had come to settle in the villages were recorded during the village visits in 2008-2011 based upon the village household lists and discussions with key informants (such as village officials or teachers). Data included the year of arrival, where the immigrants had come from, why they had come to the village, whether they stayed or not, and, if they moved on, the reason why this happened. Based upon the historical village household lists and discussions with key informants it was possible to assemble the same data for the years 2005-2007. Chi-squared tests were used to make comparisons between the proportion of immigrants that were allowed to settle for the treatment types (border areas, controls, inside PAs; and for PES and non-PES villages within-PAs). In addition, all new settlements that were near the village but not formally included in the village household list were identified for the period 2001-2011 based upon discussions with the same key informants and by recording settlements encountered along the roads leading to villages. Data collected on new settlements included the approximate year of establishment, where the people came from and why, and the number of households present. Chi-squared tests could not be used to analyse the results because only nine new settlements were recorded. Instead, generalised linear models with a binomial link function (logistic regression) in R 2.14.2 were used to analyse whether or not a new settlement was recorded within 8 km a village (border PA, controls and inside PAs) before 2005 (before PA management started) and after 2005 (when PAs were actively managed).

Land concessions. From 2001 onwards, the location of any gazetted large-scale concession for economic development or land speculation was obtained from official documents and mapped. Concessions in the study landscape began to be declared from 2008, and the majority dated from 2011 onwards (i.e. beyond the timeframe of the deforestation rate analysis). Around the study villages, social land concessions were declared in border areas and nearby parts of PAs from mid-2009, and economic land concessions for rubber were declared around two of the control villages from mid-2009. The effect of these concessions on forest cover was investigated by calculating the deforestation rate from 2005/6 to 2008/9 (prior to the declaration of the concessions) and from 2008/9 to 2009/10, when the concessions had existed for 6-9 months, for the same 1 km grid squares used for the impact evaluation (i.e. within 8 km of study villages). Data were expressed as the annual deforestation rate because the time periods were different.

Models of drivers of deforestation. The effect of the different drivers on the deforestation rate for the two time periods (2001/2 to 2005/6 and 2005/6 to 2009/10) was modelled using mixed effects models, including both the village and the identity of the 1 km grid square as random effects (since data from two time periods were available for each grid square). Explanatory variables included the distance to each of the nearest village, all-weather road, and full-day market in each time period; whether or not a grid square was inside a concession, PA or affected by a PES programme designed to reduce deforestation; and the human population density in terms of number of households in the base year (2001/2 or 2005/6), interpolated as a surface using a kernel function with a radius of 5 km in ArcGIS v10. All continuous variables were log-transformed to stabilise the variance, and correlated explanatory variables were not included in the same model. The distance to the nearest all-weather road or full-day market was considered a suitable proxy for immigration pressure (Kaimowitz et al., 1998). Competing models were compared using second-order AICc values calculated using maximum likelihood (Burnham & Anderson, 2002). Final model coefficients were then estimated using restricted maximum likelihood for the most conservative model (Burnham & Anderson, 2002).

Data was also collected on the annual costs of the PA and PES interventions, based upon park budgets and annual donor reports.

Table 5.2. Changes in deforestation rates between the four years prior to establishment of the Protected Areas (2001/2-2005/6) and the subsequent four years after establishment (2005/6-2009/10) for Protected Areas, controls, border areas to Protected Areas, and, within the Protected Areas, for villages receiving payments and villages not receiving payments. Data are based upon the average deforestation rate (in hectares per 1 km grid square) in the areas surrounding the villages. The matching estimator indicates the significance of the difference between the deforestation rate within Protected Areas and controls or border areas, and between the deforestation rate around villages within PAs receiving payments and villages not receiving payments.

	Landscape-level interventions			Within Protected Areas only	
	PAs	Controls	Border Areas	PES	No-PES
Grid squares (villages)	1356 (16)	913 (7)	1035 (11)	217 (4)	433 (11)
2001/2 to 2005/6 deforestation rate in hectares/km^2	0.872	1.398	2.193	2.529	0.534
(standard error)	(0.105)	(0.173)	(0.167)	(0.477)	(0.086)
2005/6 to 2009/10 deforestation rate in hectares/km^2	0.636	2.001	3.595	0.734	1.298
(standard error)	(0.058)	(0.214)	(0.194)	(0.096)	(0.151)
Difference between periods [a]	-0.236 *	0.603 *	1.402 ***	-1.795 ***	0.765 ***
Differences prior to the implementation of the interventions (from 2001/2 until 2005/6):					
average difference (between PAs and controls or border areas; and PES and no-PES areas within PAs) [a]		-0.526 ***	-1.322 ***		1.966 ***
matching estimator clustered by village [a]		0.185 (ns)	-0.592 (ns)		1.446 (ns)
Estimated effect of interventions (from 2005/6 when implementation started until 2009/10):					
average difference (between PAs and controls or border areas; and PES and no-PES areas within PAs) [a]		-1.366 ***	-2.959 ***		-0.564 ***
matching estimator clustered by village [a]		-1.152 ***	-2.352 ***		-0.712 *

[a] Tests of difference are t-tests for the average difference and z-tests for clustered matching. Significance values: ns = not-significant; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$

5.3 Results

Deforestation Rates

Deforestation rates inside the PAs significantly decreased after establishment of the PAs in 2005/6, whereas deforestation rates increased considerably in both control areas and the border areas around the PAs (Table 5.2).

The differences in deforestation rates were generally much greater if the entire sample was used, in comparison with the matching estimators that compared similar squares only (Table 5.2). Other studies that have compared matching estimators with regression estimators have found similar differences (Andam et al., 2008), suggesting that the matching estimators give a more conservative estimate of the difference between treatment types. Based upon the matching estimators, the difference between the deforestation rate in the PA squares and control squares was not significant in the four years before the PAs were established. After PA establishment, deforestation rates in the control squares were significantly greater than the PA squares. According to the matching estimators, the control areas lost an additional 1.15% of their forest area during the four year period, in comparison with the PA squares, which had a deforestation rate of only 0.64%. PAs therefore reduced the deforestation rate to just under a third of that in the control areas.

The deforestation rate in border areas was much greater than the other two treatment types (Table 5.2). The difference between the deforestation rate in the border areas and the PAs was relatively minor during the first period, but increased significantly after PA establishment. According to the matching estimators, the border areas lost an additional 2.35% of their forest cover during the four-year period, in comparison with the PA squares. The PAs therefore reduced the deforestation rate to just under a fifth of that in the border areas.

Prior to 2005/6, villages that were later selected targeted by the PES programmes had greater deforestation rates in the surrounding area than other villages inside PAs (Table 5.2). To some extent, these villages were targeted for the PES intervention because they were known to be clearing important forest for biodiversity. During the period 2005/6 to 2009/10, when the PES intervention started to be implemented, the deforestation rates around these villages were significantly lower than the other villages inside the PAs (Table 5.2). According to the matching estimators, the difference is equivalent to a reduction in the deforestation rate of about 50% as a consequence of the PES intervention.

Proximate causes of deforestation

During 2009 and 2010, approximately four years after PA management started, nearly 12% of grid square visits around control villages recorded land clearance or logging, in comparison with only 1.3% of grid square visits around villages inside the PAs (Figure 5.1). Within the PAs, recorded incidents of activities around non-PES villages were much greater than for PES villages, particularly for land clearance (Figure 5.1). These differences are highly significant for the PA intervention (Table 5.3) and slightly significant for the PES intervention (Table 5.3). As the focus group discussions had suggested, the prevalence of activities varied through the calendar year, with the majority of activities carried out during the dry season months after the harvest (late January onwards) and the early wet season (May-June), when new land plots are cleared (Table 5.3). Incidents were far less frequent during July-August, when paddyfields are ploughed and rice is sown, and the late wet season or early dry season (November-December) when crops are harvested. In between these periods (September-October) rates of logging were particularly high.

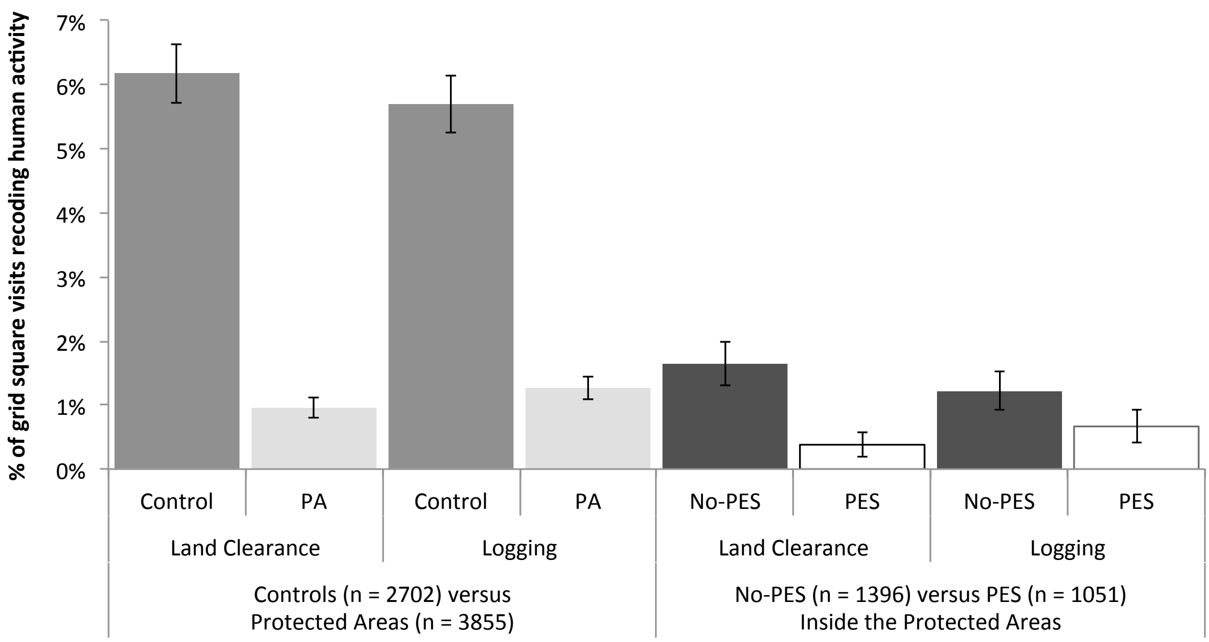


Figure 5.1. Recorded incidents of land clearing and logging in 1 km squares by local data recorders on a monthly basis during 2009-2010 for Protected Areas and matched controls, and inside PAs for villages engaged in PES and villages that were not engaged in PES. Data is from a 15-month period for controls and Protected Areas and from a 17-month period for PES and non-PES areas, with data collected approximately four years after Protected Area management started. Results are presented as the percentage of 1 km square visits that encountered a particular human activity.

Table 5.3. Mixed effects models with a binomial link function (logistic regression models) for the effect of PA and PES interventions on incidents of human deforestation and forest degradation activities (e.g. forest clearing and logging). The dependent variable was whether or not a particular human activity was observed during a visit by local data recorders to 1 km grid squares. The interventions are Protected Areas (for the column 'PA impacts') and Payments for Environmental Services (column 'PES impacts'). Data are from a 15-month period for controls and Protected Areas ('PA impacts'; 1 October 2009 – 31 December 2010) and from a 17-month period for PES and non-PES areas ('PES impacts'; 1 August 2009 – 31 December 2010). Other possible explanatory variables, such as distance from the village, slope, elevation, and forest cover, were excluded from the models based on the AIC scores during the model selection process.

	PA Impacts			PES Impacts		
	Coefficient	*z*-value	*p*[a]	Coefficient	*z*-value	*p*[a]
(Intercept)	-2.709	-7.699	***	-4.751	-6.217	***
Intervention	-1.571	-4.107	***	-1.101	-2.024	*
Month: January versus:						
- February	0.837	2.889	**			
- March	0.909	3.238	**	-0.396	-0.258	
- April	1.239	4.704	***	2.438	2.729	**
- May	0.453	1.496				
- June	0.568	2.097	*	0.870	0.994	
- July	-0.860	-2.309	*	-0.537	-0.455	
- August	-0.319	-0.998		0.242	0.284	
- September	0.823	3.031	**	0.398	0.450	
- October	0.369	1.482		0.471	0.564	
- November	0.108	0.417		-0.099	-0.109	
- December	0.059	0.224		-0.048	-0.051	
% of residual variation due to the random effect of Village	24.4%			0.0%		

[a] Significance values: ns = not-significant; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

Underlying drivers

Population Growth Rates. Prior to 2005 villages in border areas had significantly greater population growth rates than either the controls or villages inside PAs (Table 5.4a). After 2005, the population growth rates for border areas remained unchanged, and the growth rates of populations in control villages increased to be approximately the same (Table 5.4b), whereas the rate for villages inside PAs was significantly lower than either the border areas (Table 5.4b) or the controls (Table 5.4b). PA management therefore reduced the rates of population growth for villages inside PAs.

Table 5.4. Mixed Effects Models for changes in village population sizes inside Protected Areas, bordering Protected Areas, and controls, during 2001-2011. Village population was expressed as the number of households, square-root transformed, and the base year for the models was 2000. Two models are presented: (a) for the period 2001-2004 when the Protected Areas were not managed; and (b) for the period 2005-2011, when the Protected Areas were actively managed. During 2001-2004 there was no significant difference in the population growth rates in controls and villages inside Protected Areas (Difference in growth rates = -0.015, t = -0.189, df = 55, P = 0.851), whereas during 2005-2011 the population growth rate in controls was significantly greater than villages inside Protected Areas (Difference in growth rates = 0.0687, t = -2.517, df = 158, P < 0.05).

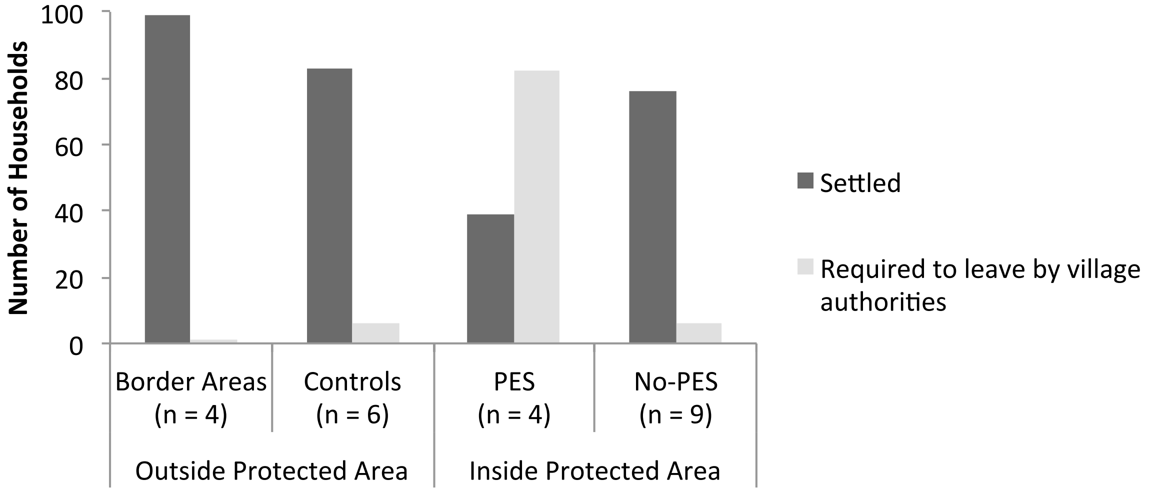
	(a) Before 2005			(b) After 2005		
Model terms	Coefficient	t-value	p[a]	Coefficient	t-value	p[a]
Intercepts (village population in year 2000):						
Border Areas	11.480			11.848		
Controls	11.572			10.965		
Protected Area	10.852			10.674		
Distance to Road (km)	-0.042	-2.226	*	-0.047	-2.286	*
Year × Border Areas	0.379	6.874	***	0.330	14.516	***
× Controls contrast	-0.189	-2.105	*	-0.045	-1.384	ns
× Protected Area contrast	-0.174	-2.557	*	-0.113	-4.164	***

[a] Significance values: ns = not-significant; * P < 0.05; ** P < 0.01; *** P < 0.001.

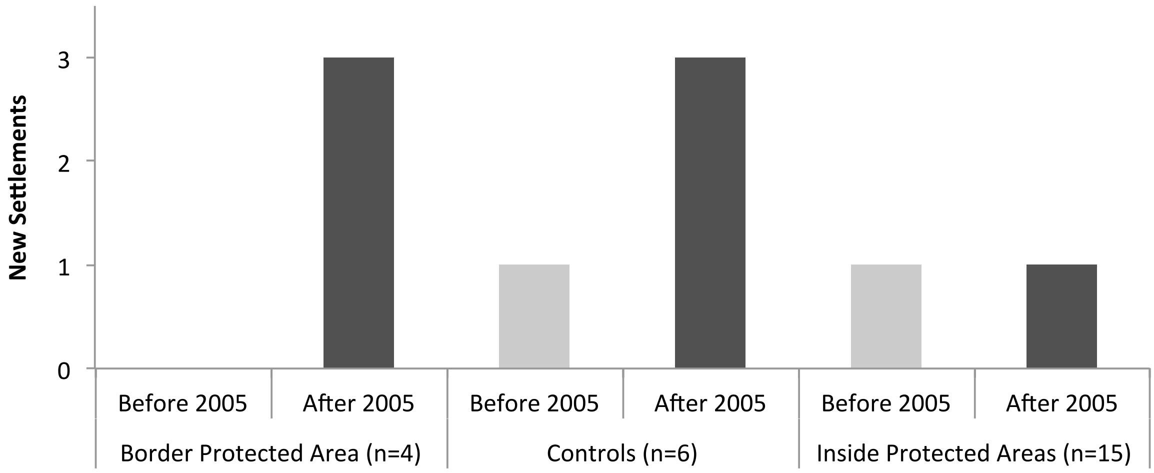
Immigration and new villages. Large numbers of immigrant households were recorded trying to settle in all the villages (Figure 5.2a). However a much larger number of immigrants were refused permission to settle in villages inside PAs than controls (χ^2 = 36.3, df =1, P < 0.001). This effect was not due to the PA per se, however, as the majority of the refusals originated from the PES villages inside the PAs. PAs did significantly reduce the creation of new settlements (Figure 5.2b). After 2005, significantly fewer villages inside the PAs had a new settlement nearby in comparison with the controls (generalised linear model with binomial link function, z = 2.002, P < 0.05), whereas before 2005 there was no difference between treatments (generalised linear model with binomial link function, z = 0.683, P = 0.495).

Land Concessions. Both Economic and Social Land Concessions dramatically increased the deforestation rate between 2008/9 and 2009/10, even when they were only operational for a few months (Figure 5.3). Economic Land Concessions were generally more extensive than Social Land Concessions, since the latter were primarily for resettlement of people whereas the former were for large-scale industrial agriculture or tree crops. During 2009-2012 more than 50 concessions were declared across the study landscape, generally outside of the core management zones of the two PAs (Figure 5.3 and Figure 5.4). Deforestation rates, particularly outside the core management zones of the PAs, are therefore expected to increase substantially in the future.

Figure 5.2. Effects of Protected Areas and Payments for Environmental Services on immigration to existing villages and new settlements during 2001-2005 (before the interventions started) and 2005-2011 (when the interventions were being implemented).



(a) Recorded incidents of immigration for villages inside Protected Areas receiving payments compared with villages inside Protected Areas that were not receiving payments, villages in border areas and controls, during 2005-2011. The proportion of immigrant households required to leave is significantly higher for villages receiving payments than other villages inside protected areas (χ^2 = 70.3, df =1, P < 0.001) and control villages (χ^2 = 76.0, df =1, P < 0.001).



(b) Creation of new settlements near to villages inside Protected Areas, in border areas, and controls before 2005 and after 2005 when protected area management started. After 2005 the number of new settlements created in border areas and controls was significantly higher than Protected Areas, before 2005 there was no significant difference.

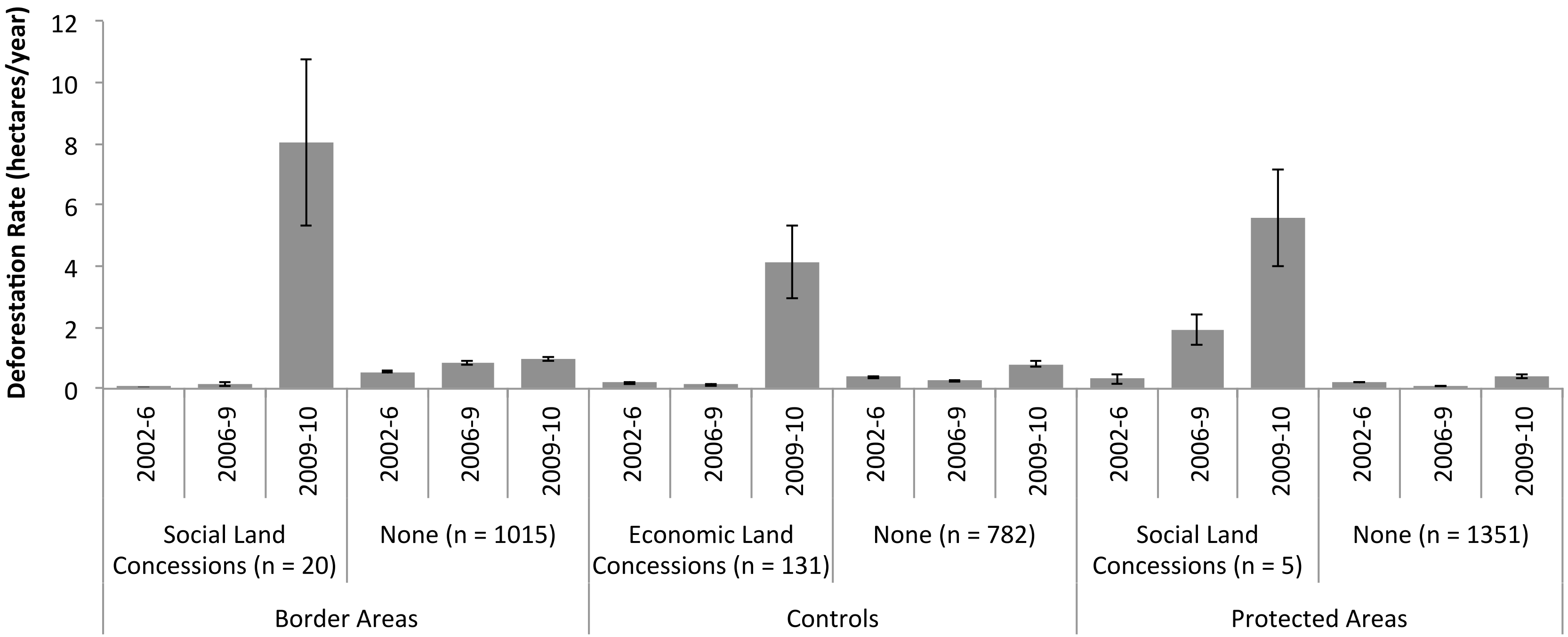


Figure 5.3. Impact of Economic and Social Land Concessions on deforestation rates, calculated as the number of hectares cleared per year in 1 km grid squares (also equivalent to the percentage annual deforestation rate). Social and Economic Land Concessions were declared across all three treatment types (border areas, controls and Protected Areas) from 2008/9 onwards, but the number of 1 km squares impacted was much higher in control areas (*n* = 131, 14.3% of squares) and border areas (*n* = 20, 1.9% of squares) than inside Protected Areas (*n* = 5, 0.4% of squares). The high deforestation rate during 2005/6 to 2008/9 inside the Protected Areas for the social land concessions grid squares was due to a new road built to the proposed concession site.

Table 5.5. Mixed effects model for the effect of drivers of deforestation and conservation interventions (Protected Areas and PES) on deforestation rates during 2001/2 to 2005/6 and 2005/6 to 2009/10 for the 3304 one kilometre grid squares (1035 bordering Protected Areas, 913 controls and 1356 inside Protected Areas, of which 217 were affected by PES programmes). Deforestation rate was expressed as the number of hectares cleared in each 1 km grid square in each time period, and grid squares were clustered around villages. All variables were log transformed.

Model term	Coefficient	*t*-value	*p*[a]
Border Protected Area during 2001/2 to 2005/6	1.294	5.103	***
- Control contrast	0.065	0.558	ns
- PA contrast	-0.068	-0.816	*
- PES areas contrast (additional to PA)	0.209	2.810	**
Change for Border Protected Area during 2005/6 to 2009/10	0.084	12.272	***
- Control contrast	-0.045	-4.462	***
- PA contrast	-0.086	-9.132	***
- PES contrast (additional to PA)	-0.043	-2.613	**
- Social Land Concession contrast	0.957	6.662	***
- Economic Land Concession contrast	0.257	3.525	***
Base area of forest	0.123	3.398	***
Population density	0.054	1.867	(*)
Distance to village	-0.343	-6.509	***
Slope	0.028	0.707	ns
Distance to road (in 2001/2 or 2005/6)	-0.182	-8.312	***
Distance to market	-0.171	-3.098	**
% of residual variation due to the random effect of Village	9.4%		

[a] Significance values: ns = not-significant; (*) $P < 0.1$; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

Models of drivers of deforestation. The mixed effects model combining all the drivers of deforestation and the PA and PES interventions confirms the individual results. There was little difference in the deforestation rate between the treatment types during 2001/2 to 2005/6 prior to initiation of the interventions (Table 5.5). PAs, and the PES programmes within the PAs, significantly reduced the deforestation rate during 2005/6 to 2009/10. The magnitude of the effect of each programme was approximately equal, although the matching estimators (Table 5.2) suggest that PAs had a greater effect than PES. Deforestation rates were significantly affected by the human population density and distance to village; the accessibility in terms of distances to roads and markets; and the availability of forest in the base year. Slope had limited effects, mainly because there is little topographical variation in the study area. Both types of land concessions significantly increased deforestation rates.

5.4 Discussion

Effectiveness and efficiency of Protected Areas and Payments Environmental Services at protecting forests

Actively managed PAs were very effective at reducing rates of deforestation. This key result was robust, even when accounting for the non-random placement of PAs, adjusted for using the matching estimators. Similar conclusions have been drawn by other studies (Andam et al., 2008; Joppa & Pfaff, 2010; Joppa & Pfaff, 2011; Nelson & Chomitz, 2011), and this study therefore contributes to an emerging body of evidence that suggests that PAs can be effective at reducing deforestation (Ferraro et al., 2012). PAs should therefore be viewed as an important component of any national strategy designed to reduce emissions from deforestation (REDD+; Angelsen, 2010).

The study was one of the first evaluations of the environmental impact of a user-financed PES programme (but see Chapter 4; Clements et al., 2012b). The results indicate that user-financed PES programmes, when used as a complement to PA management, can deliver further additional environmental outcomes (in this case measured as the contribution to forest protection) over that achieved by the PA intervention alone. In this study, the effect of implementing PES in villages inside PAs was approximately equivalent to reducing deforestation rates by an additional 50-100% over than achieved by the PA alone.

Although impressive, these results should be seen in the context of the costs of each intervention. Funding for each PA was approximately $2-3/hectare/year for the core management zones during 2005-2012, which is broadly comparable to the funding available for PA management in other developing countries (Bruner et al., 2004). By contrast, PES payments totalled between $4,000 and $20,000/village/year or $20-$100/hectare/year (assuming an 8 km radius around each village), plus the transaction costs of establishing the PES programmes. PAs were therefore considerably more efficient at delivering reductions in deforestation. Thus whilst PES may be more efficient at delivering additional environmental outcomes than programmes offering indirect incentives, such as Integrated Conservation and Development Programmes (ICDPs; Ferraro, 2001; Ferraro & Kiss, 2002; Wunder, 2007), PES in this study was considerably less efficient than very modest investments in command and control mechanisms.

Table 5.6. Summary of drivers and the impact of interventions on drivers based upon the results of this study.

Driver	Effect on Deforestation Rate	Impact of PA intervention on drivers	Impact of PES intervention on drivers
External			
Immigration to existing villages	Moderate	Low	High
Immigration to form new settlements	High	High	Low
Access to markets and roads	High	Low	Low
Economic Land Concessions	High	High	Low
Social Land Concessions	High	Moderate	Low
Land clearance and logging by outsiders	High	High	Low
Internal			
Intrinsic Population Growth	Moderate	None	None
Shifting agriculture for rice or cash crops	Moderate	Moderate	High
Expansion of paddyfields	Moderate	Low	High
Logging	Low	Low	Low

Effects of Protected Areas and Payments for Environmental Services on internal and external drivers of forest loss

During the decade 2000-2010, the deforestation rate in Cambodia was amongst the highest in the world (FAO, 2011), with the north-eastern border identified as an area with especially high deforestation (WRI, 2007; Rainey et al., 2010). A contributing factor is the demand from Vietnam for natural resources (Meyfroidt & Lambin, 2009). Three different agents cause deforestation in northern Cambodia: companies, people moving into the study area and local residents (Rainey et al., 2011). The activities of these agents can be divided into external drivers of deforestation, which includes immigration to existing villages and to form new settlements, access to markets and roads, establishment of large-scale land concessions, and land clearance and logging by people living outside the PAs; and internal drivers, which includes intrinsic population growth, and logging, shifting agriculture and expansion of paddyfields by local residents (Table 5.6; Rainey et al., 2010). These agents and drivers are broadly similar to those identified by other studies (Angelsen & Kaimowitz, 1999; Geist & Lambin, 2002). Agricultural expansion, in particular, is well known to be one of the primary drivers of tropical deforestation (Gibbs et al., 2010).

PAs and PES had different effects on the different drivers of deforestation. The evidence suggests that the principal effect of the PAs, since the start of active management, was to mitigate external drivers of deforestation, particularly immigration to form new settlements, large-scale concessions for agro-industrial development, and land clearing activities by people living outside the PAs who had to travel along main access routes and hence were easily detected by PA staff (Table 5.6). By contrast, the PES programmes were effective at changing the deforesting behaviour of local PA residents, but were broadly ineffective against external drivers that they had not designed to target. The PES programmes did also reduce immigration into existing villages, because local residents were concerned about the potential loss of payments if the local deforestation rate increased considerably. The analysis of drivers of deforestation therefore suggests that the two different policy mechanisms were highly complementary. Indeed, implementation of the PES programmes would have been impossible without the protective effect of the PAs. These results indicate that, in the Cambodia context, the principal effect of the PAs was to exclude outsiders, thereby securing forest and land resources for the use of local residents, and contributing significantly to local poverty reduction (Chapters 3 and 6; Clements et al., 2012c). Since 2008, however, the security of tenure afforded by PAs has been compromised by new policies that allow large-scale concessions to be declared within Cambodian National Parks and Wildlife Sanctuaries, which is leading to degazettment of many parks despite widespread opposition from local people (Figure 5.4; Vierze and Naren, 2012). Given the benefits to local livelihoods afforded by PAs (see Chapters 3 and 6; Clements et al., 2012c), the consequences of these policies for the livelihoods of local residents are unclear.

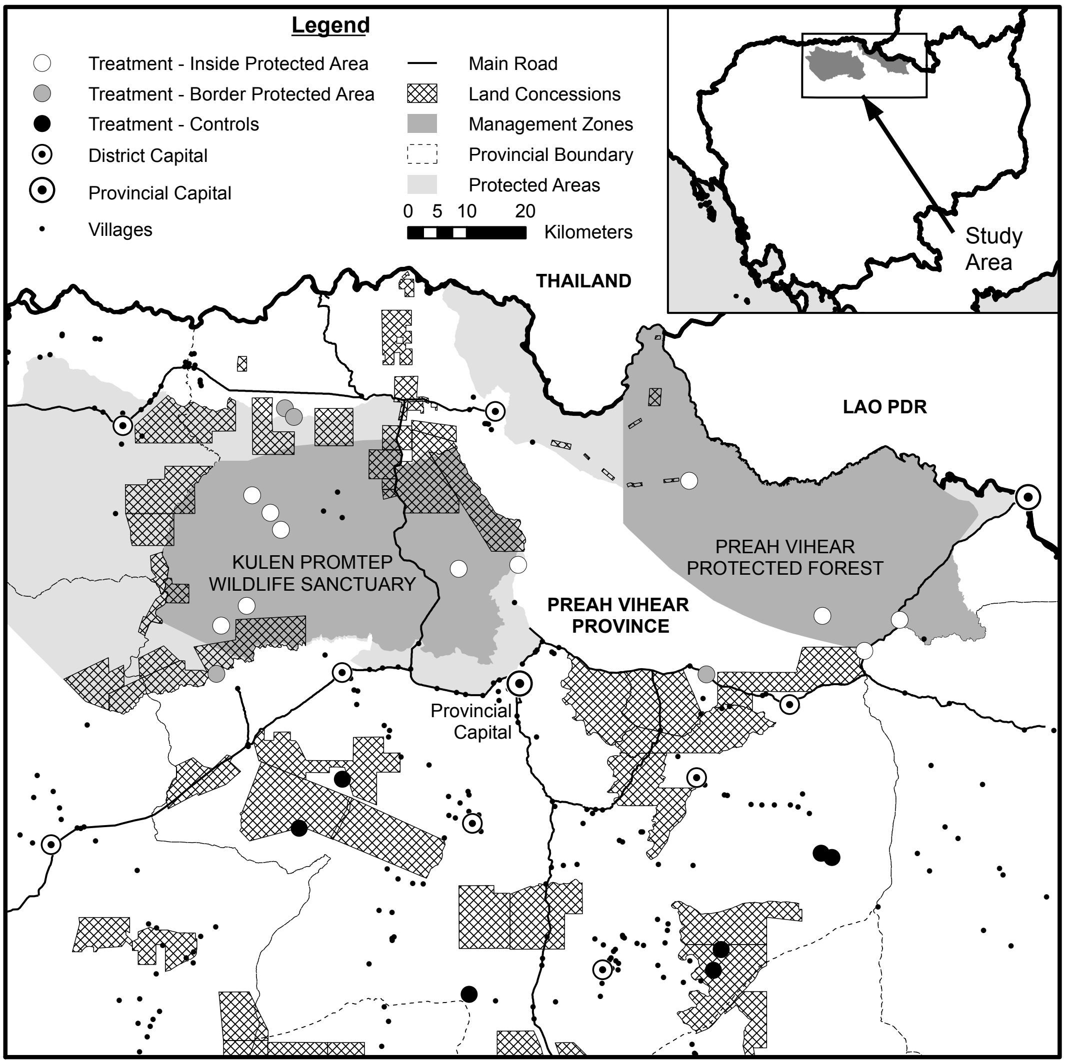


Figure 5.4. Map of the study area, showing the location of the study villages and all Economic and Social Land Concessions declared between 2008 (when the first concessions were announced) and 2012. The majority of concessions were approved after mid-2011. Deforestation rates under areas under concession increased rapidly after the concessions were approved, even in the short period for which data is available (Figure 5.3). The rapid increase in the number of concessions suggests that deforestation rates will continue to rise in the future.

Chapter 6. Do Payments for Environmental Services and Protected Areas support local livelihoods whilst conserving forests in northern Cambodia?

Abstract

The potential impacts of Payments for Environmental Services (PES) and Protected Areas (PAs) on local livelihoods in developing countries are contentious and have been widely debated. The available evidence is sparse, with few rigorous evaluations of the social impacts of PAs and none of PES. This study measured the impacts on human wellbeing of three different PES programmes instituted within two PAs in northern Cambodia, using a panel of intervention villages and matched controls. PAs increased security of access to land and forest resources for local households, benefiting forest resource users, but restricting households' ability to expand and diversify their agriculture. PES impacts on household wellbeing were related to the magnitude of the payments provided: the two higher-paying market-linked PES programmes had significant positive impacts for participants, whereas a lower-paying programme that targeted biodiversity protection had no detectable effect on livelihoods, despite its positive environmental outcomes. Households that signed up to the higher-paying PES programmes, however, typically needed more capital assets and hence they were less poor and more food secure than other villagers. Therefore, whereas the impacts of PAs on household wellbeing were limited overall and varied between livelihood strategies, the PES programmes had significant positive impacts on livelihoods for those that could afford to participate. The results confirm theories that PES, when designed appropriately, can be a powerful new tool for delivering conservation goals whilst benefiting participants.

6.1 Introduction

A critical question for conservation policy is whether interventions incur net costs or provide net benefits to the local people who are the most directly impacted. There is now widespread acceptance that conservation policies should, at the very least, do no harm, and where possible should contribute to poverty alleviation (CBD, 2008). Protected Areas (PAs) are one of the most widely adopted policies, currently covering >12% of the terrestrial land surface (UNEP-WCMC, 2012). The debate around the impacts of PAs has, however, been particularly contentious, with large numbers of case studies documenting costs for local people, such as restrictions on agriculture or access to natural resources (Brockington & Igoe, 2006; Cernea & Schmidt-Soltau, 2006; Coad et al., 2008; Roe, 2008). The concern about such negative impacts is a key reason why newer policies, such as Payments for Environmental Services (PES), which provide benefits to local people conditional upon achieving an environmental outcome or a change in behaviour, have gained in popularity (Ferraro & Kiss, 2002; Wunder, 2007). It is hypothesized that PES should improve local wellbeing due to the benefits provided (Ferraro & Kiss, 2002; Pagiola et al., 2005). This hypothesis has, however, never been tested using empirical data (Pattanayak et al., 2010).

Rigorous impact evaluation methods are widely credited with having transformed development policy, by quantifying the contribution that specific interventions have made to improvements in human wellbeing (Banerjee & Duflo, 2011). There have been widespread calls for adoption of the same methods in environmental science (Ferraro & Pattanayak, 2006). Most published studies to date have focused on assessing environmental rather than social outcomes, e.g. using impact evaluation methods to show that PAs and PES policies do indeed protect forests (Andam et al., 2008; Joppa & Pfaff, 2011). These studies have also shown the critical importance of making comparisons with appropriate controls in order to avoid over-estimating the effectiveness of interventions (Andam et al., 2008). Only three studies have evaluated the social impacts of conservation policies, focusing on PAs in developing countries, and in these cases PAs had no net impact or slightly positive impacts for local people (Andam et al., 2010; Sims, 2010; Naughton-Treves et al., 2011). There have been no impact evaluations of the effects of PES on wellbeing in developing countries.

Published social impact evaluations also focus on net impacts using single measures of poverty, whereas human wellbeing is complex and multifaceted (Scoones, 1998; Agrawal & Redford, 2006; McGregor, 2007). Interventions with minimal effects on income may nonetheless contribute to less tangible aspects of wellbeing such as access to resources and education. There is also an urgent need to disaggregate outcomes in order to understand the impacts of conservation interventions on different subsets of society (Daw et al., 2011), especially vulnerable groups. Impacts must also be understood in the context of a dynamic system – livelihoods are complex and changing, particularly as rural populations become increasingly linked to markets.

In this study, impact evaluation methods were used to quantify the impact of PAs and PES over time on a panel of intervention and matched control households practicing a range of livelihood strategies in villages in the northern forests of Cambodia. Northern Cambodia was an ideal location to test the impacts of PAs and PES because the interventions were initiated relatively recently, thereby allowing before-after comparisons to be made, the PAs had existing residents, and the PES programmes have been well documented (Clements et al., 2010). The study specifically considered (*1*) the impact of PAs on multiple aspects of local wellbeing; (*2*) the additional impact of PES programmes on local wellbeing; and (*3*) differential impacts of these interventions on different livelihood strategies.

6.2 Study Site and Survey Design

Protected Areas and PES interventions

The study focused on the core management zones of two PAs in northern Cambodia (see Chapter 2; Figure 2.5): Kulen Promtep Wildlife Sanctuary (gazetted 1993) and Preah Vihear Protected Forest (gazetted 2002). Both PAs were paper parks until active management started in 2005. Three PES programmes were instituted in villages within the PAs to complement PA management: (*1*) direct payments for protection of nests of globally threatened birds; (*2*) community-managed ecotourism linked to wildlife and habitat protection; and (*3*) providing premium prices for agricultural goods to households that kept to the land-use plans (Ibis Rice; see Chapters 2 and 3; Clements et al., 2010; Clements et al., 2012c).

Social Outcome measures

Outcomes were measured for three comparison groups: villages within PAs; control villages >20 km from the PA boundaries that were selected using quasi-experimental matching to be as similar to the within-PA group as possible; and villages bordering the PAs (4-12 km from the PA boundary), which is the standard comparison made by most studies. Five measures of household wellbeing were recorded: (*1*) poverty, using the Basic Necessities Survey which incorporates many of the multiple dimensions of poverty (Scoones, 1998; Agrawal & Redford, 2006; McGregor, 2007) into a single score, relative to a locally-derived definition (Pro-Poor Centre & Davies, 2006; see Chapter 2); (*2*) rice harvests, since rice is the staple crop in Cambodian diets; (*3*) food security; and (*4*) the education level of each household member. Households were categorized by which livelihood strategies they practiced (e.g. resin-tappers, shifting cultivators). Livelihoods data was collected in 2008, three years after PA management was initiated and before households were paid from any of the PES programmes, and in 2011, after households had been receiving payments for 1-3 years (depending on the programme and the household).

Controlling for observed and unobserved sources of bias

Controlling for potential sources of bias is necessary to provide confidence that observed differences are due to the interventions rather than other factors. Bias control was conducted at the village level, because this was the scale at which the PA and PES interventions were implemented. Matching methods were used to select seven appropriate control villages outside the PAs against which to measure the impacts of the interventions (see Chapter 2). Matching ensured that the control villages were as similar as possible to the within-PA villages for observed baseline characteristics in 2005 (prior to the initiation of the interventions) that would be expected to have significant effects on local livelihoods and deforestation rates. After matching, the covariate distributions of the sample of intervention and control villages were not significantly different (see Chapter 2, Table 2.1), thereby removing observable sources of bias. Potential spill over effects were controlled for by selecting matches that were >20 km from the PA boundaries.

Matching controlled for possible observable confounding factors, but unobserved factors might still bias the results. Therefore panel methods were used, following the same households over time to calculate difference-in-difference estimators. Panels eliminate the potential bias due to unobserved factors (e.g. unobserved household variables), assuming that the effect of these factors is similar over time for different treatment units (Wooldridge, 2002).

6.3 Methods

Household Surveys

Surveys were conducted by trained Cambodian social researchers in 20 villages: 11 within the PAs, 5 matched controls 20-60 km outside the PAs, and 4 villages 4-12 km from the PA boundaries (Chapter 2, Figure 2.5). The first assessment of 871 households was conducted in September-November 2008, three years after the PA management activities were initiated, and coinciding with the first year that the market-linked PES programmes were scaled up in four villages (see Chapter 3). Households were selected using random stratified sampling, based on a participatory wealth ranking exercise in each village. The second assessment took place in July-September 2011 for the same households and an expanded sample (1053 households total). 12% of the original sample could not be located, either because people had moved away or were absent. The final panel had 769 households that were interviewed in both time periods (443 within PAs, 185 controls, and 141 in border villages). Each household was asked a series of standardized questions on key characteristics, livelihood strategies, and a relative measure of locally-relevant poverty: the Basic Necessities Survey (Pro-Poor Centre & Davies, 2006), which previous research had indicated was highly correlated with other poverty measures (Chapter 3; Clements et al, 2012c). The PA impact evaluation used the entire panel of 769 households. The PES impact evaluations used a subset from villages within the PAs: for Ecotourism and Ibis Rice 174 households from 4 villages, of which 27 and 50 households respectively were paid during 2008-11; and for the Bird Nests 247 households from 6 villages, of which 28 were paid during 2008-11.

Analyses of wellbeing and livelihood variables

Mixed effects models in R 2.14.2 were used to analyse the covariate wellbeing variables (poverty, rice harvests and food security), with village as the random effect, using package "nlme" (Pinheiro et al 2011). Models were formulated for each variable in 2008 and 2011, and for the change in each variable between 2008 and 2011 ('first difference models'; Wooldridge, 2002). The latter control for initial conditions – such as baseline poverty status of households – by including this in the model. Competing models were developed based on the main research questions and compared using second-order AICc (Akaike's Information Criterion corrected) values calculated using maximum likelihood (Burnham & Anderson, 2002; see Chapter 3 for details of the model selection process). Final model coefficients were then estimated using restricted maximum likelihood (Burnham & Anderson, 2002). Contrasts tests were used to compare households within PAs with households in control villages and households on the border of PAs with households within PAs.

Binomial categorical variables (e.g. if a household was a resin-tapper) were analysed using mixed effects models in R package "lme4" (Bates et al., 2011) with a binomial error distribution (logistic regression models). Models were used to compare the differences between interventions and years in terms of the livelihood strategies practiced by households, and to determine which variables were characteristics of households that chose to sign up for PES programmes. Education was expressed as whether a child was attending high school or not.

Table 6.1. Change in household wellbeing and livelihood strategies for a panel of 769 households bordering, within and controls outside Protected Areas in northern of Cambodia during 2008-2011. Tests of difference are mixed effects regression models for continuous variables (poverty, rice harvest, food security, cattle), and generalized mixed effects models with a binomial link function for categorical variables. Households could have more than one livelihood strategy.

	Border Protected Area		Within Protected Area		Controls		Tests of difference (Within PA vs Controls)		
	2008	2011	2008	2011	2008	2011	2008	Change	2011
Households	141	141	443	443	185	185			
Wellbeing Variables									
Poverty	10.5	12.5	9.6	11.8	8.0	11.4	*	ns	ns
Rice Harvest (kg)	2181	3015	1851	2506	1293	2329	ns	ns	ns
Food Security (kg)	219	1942	-230	1337	-633	1109	ns	ns	ns
Livelihood strategies									
Resin-tapper (%)	32	30	55	59	28	37	***	ns	***
Rice Farmer (%)	94	96	91	96	94	95	ns	ns	ns
>1 hectare of paddyfields (%)	90	90	73	85	63	79	*	ns	ns
Mini-tractor (%)	36	54	30	60	26	37	ns	**	***
Rice Shifting Cultivation (%)	38	27	37	26	45	39	*	*	**
Cash Crops	n/a	5	n/a	2	n/a	10			**
Employed (%)	11	10	6	9	3	4	ns	ns	*
Service or Shop (%)	23	24	14	26	14	29	ns	ns	ns

Tests of difference significance values: ns = Not-significant; * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$

Table 6.2. Change in household wellbeing and livelihood strategies for a panel of 769 households in the northern Cambodia during 2008-2011, showing changes for the poorest and richest quintiles in 2008. Poverty was measured using the Basic Necessities Survey score.

	All		Bottom Quintile		Top Quintile	
	2008	2011	2008	2011	2008	2011
Households	769	769	153	153	151	151
Household Size	5.8	5.9	5.5	5.7	6.1	6.0
# Working Adults	3.1	3.2	2.8	2.9	3.5	3.6
Dependency Ratio	1.0	1.0	1.2	1.2	0.9	0.8
Female-headed households (%)	8	7	12	13	3	7
Household head Education (years)	3.0	3.1	1.8	1.7	4.7	4.8
Household Head Age (years)	41.1	43.4	38.2	40.9	42.5	45.0
Wellbeing Variables						
Poverty	9.4	11.8	5.0	9.0	14.0	14.4
Rice total harvest (kg)	1777	2557	816	1355	2893	3820
Food security (kg)	-244	1393	-972	264	646	2625
Resin yield (litres)		619		417		605
Livelihood Strategies						
Resin-tapper (%)	44	49	31	41	50	42
Rice Farmer (%)	92	96	83	94	97	97
Have >1 hectare of paddyfields (%)	74	85	37	66	97	97
Rice Shifting Cultivation	39	29	52	48	20	17
Cash Crops (%)		5		2		7
Employed (%)	6	8	1	3	14	16
- Public Sector (%)	5	5	1	1	11	13
Service or Shop (%)	16	26	5	9	38	49
Assets						
# Cattle (heads)	4.1	3.5	1.7	1.8	7.3	6.2
Mini-tractor (%)	30	54	3	17	64	84
Cattle Draft (%)	37	25	12	22	62	30

6.4 Results

How are livelihoods changing over time?

National economic growth in Cambodia averaged nearly 10% during 1998-2008, declined to 0.1% in 2009 during the global financial crisis, and recovered to approximately 6-7% during 2010-2012 (World Bank, 2012). In the context of this rapid economic development, it is unsurprising that the wellbeing of households in the panel increased significantly between 2008 and 2011 (Table 6.1), both for the poorest and richest quintile in the sample (Table 6.2). On average, household poverty decreased, agricultural productivity and food security improved, land holdings increased, more households operated family businesses, and adopted mechanized agriculture. Households switched from shifting cultivation – which requires lower inputs but is less productive – to paddy rice, and adopted cash crops. Increased mechanization was funded by sales of assets, particularly resin and livestock, leading to declines in the number of cattle and the use of animals for agriculture (Tables 6.1 and 6.2). Households that were poorer in 2008 were less likely to make these switches, and tended to be less educated, with fewer working adults, younger household heads, fewer assets, and they were unlikely to operate family businesses or have jobs (Table 6.2).

What were the additional impacts of Protected Areas on wellbeing?

In both 2008 and 2011 households bordering PAs were less poor, had greater rice harvests and were more food secure than households within PAs or controls in remote forest areas outside PAs (Table 6.1). Mixed effects models indicate that these absolute differences were highly significant in both 2008 (see Chapter 3, Table 3.5 and Figure 3.3a; Clements et al., 2012c) and 2011 (Appendix 6, Table S6.1), and households bordering PAs also increased their rice harvests at a greater rate than the other treatment groups (Appendix, Table S6.2). These results can be explained because households bordering PAs are closer to roads, markets and services than households both within PAs and controls (see Chapter 3; Clements et al., 2012c).

Households within PAs typically were less poor than controls in 2008 (Table 6.1; see Chapter 3, Table 3.5 and Figure 3.3a), however the rate of change in household poverty status was not significantly different between the two treatment groups, and there were no differences for either rice harvests or food security in either the absolute values or the rate of change (Appendix, Tables S6.1-S6.2). The percentage of residual variance explained by the village term was low in all models, implying that unobserved factors at the village level did not bias the results (Appendix, Tables S6.1-6.2). The overall impact of PAs on households was therefore quite limited, suggesting that rates of change were mainly due to larger economic factors at the landscape or national level, such as Cambodia's rate of economic growth during the study period.

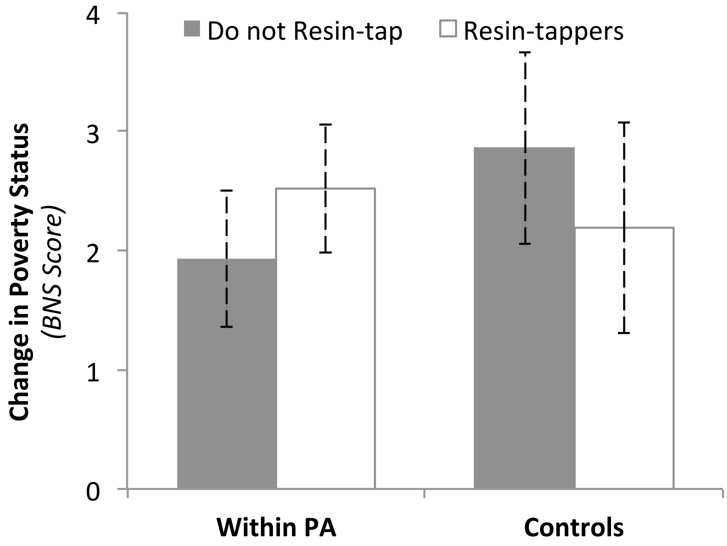
Did Protected Area impacts differ between livelihood strategies?

There was considerable convergence in some livelihood strategies between within-PA and control households, such as owning >1 hectare of paddyfields, mechanization of agriculture, and operating a shop or household business (Table 6.1). However households within PAs were more reliant upon NTFPs, particularly resin, than households bordering PAs or controls (Table 6.1). Control households were more likely to practice shifting cultivation for rice, and this difference between the controls and within-PA households significantly increased from 2008 to 2011. Control households were also far more likely to be growing cash crops in 2011. The PA authorities strictly restricted expansion of shifting cultivation and cash crops, which probably explains these differences. Employment rates increased within PAs during 2008-2011, in comparison with controls, principally due to hiring of local villagers by the PAs. The rate of increase of agricultural mechanization was greater for households within PAs (who had access to more income from natural resources) than controls.

Although the average rates of change were similar between households within PAs and controls, significant differences were observed for different livelihood strategies. Within PAs, resin-tappers improved their poverty status at a greater rate than households that did not resin-tap, whereas the reverse was observed for the controls (Figure 6.1a and Appendix, Table S6.2; significance $P < 0.01$). Households that did not resin-tap were therefore reducing in poverty significantly slower within PAs than those outside, perhaps because control households could practice other forms of agriculture (shifting cultivation and cash crops) that were restricted by the PA authorities. Tenure security over resin trees was greater within PAs, explaining why resin-tappers within PAs, which made up 59% of within-PA households in 2011 (Table 6.1), did significantly better.

Figure 6.1. Effect of Protected Areas on (a) the change in poverty status of resin-tappers, and (b) the change in rice harvests of households with >1 hectares of paddy. The graphs show the predicted effects and 95% confidence intervals from the mixed effects model (within-PA n = 443; controls n = 185).

(a) Change in poverty status for resin-tappers and non-resin-tappers. Within PAs, resin-tappers did significantly better households that did not resin-tap, whereas the reverse was observed for the controls (P < 0.01).



(b) Change in Rice Harvest for households that owned >1 hectare of paddyfields, and households that did not. Households with little land outside PAs did significantly worse than similar households within-PAs (P < 0.05).

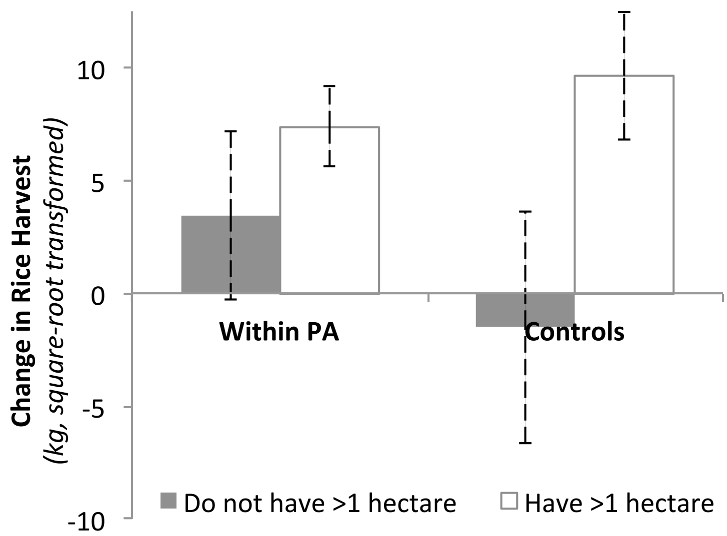


Table 6.3. Differences between household status and livelihood strategies in 2008 before the commencement of payments, for households that decided to participate in the PES programmes and households that did not, from the same villages within the Protected Areas. Data for the Ecotourism and Ibis Rice programme are based upon 4 villages (174 households) where the programmes were implemented; data for the Bird Nests programme are based upon 6 villages (247 households). Tests of difference are mixed effects models with a binomial link function. Average payments and standard errors under the three programmes are also given.

	Bird Nests participants			Ecotourism participants			Ibis Rice participants		
	Yes	No	Sig	Yes	No	Sig	Yes	No	Sig
Households	28	219		27	147		50	124	
Female-headed households (%)	7%	5%		11%	5%		0%	8%	
Wellbeing Variables									
Poverty	9.4	9.4	ns	10.8	10.2	ns	11.1	9.9	*
Rice Harvest (kg)	2154	1935	ns	2811	1926	**	2707	1804	***
Food Security (kg)	-194	-154	ns	304	-191	(*)	486	-357	**
Livelihood strategies									
Resin-tappers (%)	64	55	ns	56	56	ns	54	57	ns
Rice Farmer (%)	89	90	ns	93	88	ns	96	85	(*)
>1 hectare of paddyfields (%)	68	72	ns	93	76	*	94	72	**
Mini-tractor (%)	29	26	ns	33	25	ns	44	19	**
Rice Shifting Cultivation (%)	43	31	ns	11	18	ns	10	19	ns
Employed (%)	0	9	ns	19	7	(*)	14	6	ns
Service or Shop (%)	14	12	ns	19	13	ns	14	14	ns
Average Annual Payments per household (Standard Error)		US$132	($18)		US$225	($14)		US$413	($41)
% of Households in the villages engaged in each payment programme (total households)		7%	(616)		12%	(499)		24%	(616)
% of Households engaged in the payment programme for > 1 year		10%			62%			54%	

Tests of difference significance values: ns = Not-significant; (*) = $P < 0.1$; * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$.

Table 6.4. Effects of the payment programmes on the change in (a) Poverty; (b) Rice Harvest; (c) Food Security; and (d) if a child was attending a high school (in a district or provincial town) between 2008 (before payments) and 2011 (after payments). Mixed effects models (a-c) are based upon a panel of 174 households from four villages where the PES programmes were in operation, with 50 households involved in the Ibis Rice programme, 27 involved in Ecotourism, and 16 receiving direct payments for protection of bird nests. Model (d) is a binomial model using a panel 36 children from 28 households (from the 174 households) that had already completed primary school education by 2008.

	(a) Poverty	(b) Rice Harvest	(c) Food Security	(d) Education
(Intercept)	5.627 ***	26.020 ***	60.507 ***	-3.122 ns
Base variable	-0.386 ***	-0.503 ***	-0.746 ***	
Ibis Rice programme, payment	0.058 **	0.381 ***	0.297 ***	0.110 *
Ecotourism programme, payment	0.053 *	0.003 ns	-0.029 ns	0.074 (*)
Bird Nests programme, payment	-0.022 ns	-0.053 ns	0.078 ns	-2.048 ns
Household head education level				-0.739 ns
Change in poverty				0.311 ns
Random Effect of Households: % residual variation				0.0 %
Random Effect of Village: % residual variation	9.3 %	6.4 %	5.6 %	29.5 %

Significance values: ns = Not-significant; (*) = $P < 0.1$; * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$

Land tenure security was also higher within PAs, whereas some control households lost access to land for farming due to development pressures. Consequently rice harvests of control households that did not own >1 hectare of paddyfields slightly declined, whereas similar households within PAs showed continued improvements in rice harvests (Figure 6.1b and Appendix, Table S2; significance $P < 0.05$). Households that did not own >1 hectare made up a minority of households under each treatment type: 21% of households outside PAs and 15% of households within PAs (Table 6.1).

Who participated in PES programmes?

Entry into two of the three payment programmes was not random. Households that decided to sign up for the Ibis Rice and Ecotourism programmes had greater rice harvests, were more food secure, and were more likely to have >1 hectare of paddyfields than households that chose not to participate in 2008 (Table 6.3). Ibis Rice households were also less poor and more likely to be rice farmers using machinery. In the case of Ibis Rice this is understandable because only net rice producers could afford to sell excess rice to the programme. The Ecotourism programme required households to divert labour from agriculture to invest in tourism activities, which again suggested that households that were more food secure were more likely to sign up. As a consequence participants in the Ecotourism programme were also more likely to participate in the Ibis Rice programme (Fisher's Exact Test of independence, $P = 0.06$). For the Ecotourism some elite capture may have occurred, since participants in 2008 were more likely to be employed, particularly in the public sector. Several public employees (teachers, village chiefs and commune councillors) were initially selected to manage the Ecotourism at the village level due to their positions, although by 2011 they had mostly been replaced by elected villagers. The Ecotourism programme also positively targeted poor female-headed households through participation in a women's group that sold supplies to tourists; all Ibis Rice households were male-headed. Households engaged in the Bird Nests programme were similar to other households in the village, probably because households received immediate payments to cover the costs of nest protection (and then a bonus if the nests were successful), so there were fewer barriers to entry. Engagement in the Bird Nests programme was independent of the other two programmes (Fisher's Exact Test, P = 1 and $P = 1$).

What were the additional impacts of PES programmes on wellbeing?

The average annual payments from the PES programmes ($100-$400) were significant in comparison with the estimate of household consumption in rural forested regions from the 2007 Cambodia

Socio-Economic-Survey of \$329±16 (World Bank, 2009). Households that signed up to the Ibis Rice and Ecotourism programmes improved their poverty status at a greater rate than non-PES households from the same villages, even when the fact that some started at a higher baseline level in 2008 was accounted for (Table 6.4a, $P < 0.05$ in both cases). Ibis Rice households also increased their rice harvests and improved their food security at a faster rate than other comparable households (Table 6.4b and 6.4c, $P < 0.01$ in both cases). Households that received high payments from the Ecotourism and Ibis Rice programmes were able to afford to keep their children in school for longer, and to pay for them to attend secondary and high schools away from their home villages (Table 6.4d, $P < 0.01$). The Bird Nests programme had no additional impact on household wellbeing, perhaps because the payments were significantly lower than the other programmes (Table 6.4). The models for the PA impact evaluation also found that the Ibis Rice and Ecotourism programmes improved household wellbeing (Appendix, Table S2).

6.5 Discussion

The importance of ecosystem services to overall human wellbeing is well recognized (MEA, 2005a), but interventions to manage and conserve ecosystem services may impose costs (Coad et al., 2008; Brockington & Igoe, 2006; Cernea & Schmidt-Soltau, 2006) as well as benefits (Coad et al., 2008, Wunder, 2001; Scherl et al., 2004) on local people. This is the first study to use rigorous impact evaluation methods to analyse the social impacts of interventions to conserve and maintain ecosystem services, through PAs and PES.

Since their initiation, PAs and PES have delivered additional environmental conservation outcomes in northern Cambodia, in terms of protection of tropical forests from deforestation and, for at least one of the PES programmes, protection of globally threatened wildlife species (Clements et al., 2012b; Chapters 4 and 5). These results are robust even accounting for the nonrandom placement of PAs. The principal effect of the PAs since the start of active management was to mitigate external drivers of ecosystem loss (especially deforestation), particularly in-migration to existing villages, formation of new settlements, and large-scale concessions for agro-industrial development. Implementation of the PES programmes would have been impossible without this protective effect of the PAs; the two conservation strategies are complementary.

As a consequence of the exclusion of outsiders from the PAs, local people have been able to continue to use forest and land resources for their livelihoods based upon their legal rights under

Cambodian law, including use of forest resources and farming within agreed land-use plans. No resettlement occurred. The Cambodian PAs are therefore very different from strictly enforced PAs, but share many characteristics with the estimated 56-85% of PAs in developing countries that contain local people (Brockington & Igoe, 2006). Our principal finding is that under these conditions PA management had minimal impacts on the livelihoods of local residents on average, which is consistent with the evidence that the PAs were primarily designed to protect ecosystems from external drivers of loss. Instead, the improvements in wellbeing seen across all treatment groups were driven largely by Cambodia's rapid economic growth. PAs did, however, impose costs on local people, especially for households that grew crop types that were restricted by the PA management (principally shifting cultivation and cash crops) or wanted to clear large amounts of new land. Conversely, PAs provided notable benefits for those households that used forest resources (such as resin-tappers) who gained from the restrictions on outsiders. Continued and unsustainable use of natural resources in PAs by local residents can, however, lead to trade-offs from a biodiversity conservation perspective. PAs with resident human populations may be less effective at conserving key species (O'Kelly et al., 2012).

This is one of the first studies to evaluate the social impacts of PES programmes. Our results provide empirical support for theories that the impacts of PES on human wellbeing depend fundamentally upon programme design (Pagiola et al., 2005; Wunder, 2008). PES can support social goals by (a) minimizing constraints on programme entry by the poor; and (b) providing sufficient incentives to offset the opportunity costs of participation and thereby increase overall human wellbeing. PES programme entry constraints might include eligibility requirements or abilities, which the poor would be less likely to fulfil (Wunder, 2008). Of the three PES programmes evaluated in this study, the Ibis Rice programme had the most significant entry constraints, since participants needed to have sufficient land to produce an agricultural surplus to sell to the programme. By contrast, the Bird Nests programme, which provided direct cash payments for protection of biodiversity, required no capital assets to join and provided a portion of the payment up-front, allowing any household to participate. The pro-poor impacts of ecotourism are known to be limited by the additional capabilities required to engage in tourism (Kiss, 2004). The Cambodian Ecotourism programme specifically contained pro-poor provisions, which mitigated these barriers to some extent.

None of the PES programmes led to net negative impacts for local wellbeing. The two market-based programmes (Ibis Rice and the Ecotourism) had significant net positive impacts for their participants. The development benefits of the three programmes are linked to the magnitude of the payments made. Under the Bird Nests programme, payments were low, based upon the minimum daily wage local people were willing to accept (the opportunity cost of a day's work). Despite the lack of constraints on access, suggesting it should be the most pro-poor of the interventions, the additional livelihood benefit of the programme was therefore limited. By contrast, payment levels under the two market-based programmes were based upon the market's willingness-to-pay for the additional environmental outcomes, which was high.

In conclusion, the combination of the PA and PES interventions described here delivered additional environmental outcomes and had three important social impacts: (a) securing forest resources for local residents, which benefited some groups whilst imposing costs on others; (b) providing new significant sources of cash income for households that could afford to engage in the PES programmes; and (c) delivering net positive impacts for at least some of the PES participants. Our results confirm theories that PES, when designed appropriately, can be a powerful new tool for delivering conservation goals whilst benefiting local people, particularly as a complement to more traditional conservation interventions such as PAs.

Chapter 7. More than just money? How Payments for Environmental Services influences human behaviour

Abstract

Conservation interventions often seek to change local behaviours that lead to the loss of biodiversity and ecosystem services. The influence of three different Payments for Environmental Services (PES) programmes on the forest land clearing behaviour of local people within Protected Areas was investigated. The effectiveness of the programmes at changing local behaviours was directly related to the strength of the conditionality of the payments. An agri-environment programme, which directly rewarded farmers if they did not illegally expand their fields, was the most effective at changing individual behaviour, whereas an ecotourism programme, that only weakly linked incentives to behaviour, had limited impact. A third programme, which targeted hunting behaviour rather than land clearance, unsurprisingly had no impact on forest land clearing behaviours, although it was very successful at reducing hunting. Attitudes towards the fairness of the programmes, the benefits they provided, and the institutional arrangements of the programmes influenced how individuals behaved. Perceptions of fairness were related to the ability of individuals to access the programmes and to participate in decision-making, rather than whether or not the payment distribution was equitable. The ecotourism programme was perceived to be the least fair because it provided limited opportunities for participation, allowing elites to capture a much greater share of the payments. The ecotourism did, however, provide funding to empower local village authorities to enforce land-use plan regulations, which caused villagers' to reduce land-clearing behaviours. The comparison of the programmes suggests that both strong individual conditional payments and providing village-level incentives can change local behaviour, the former through monitoring individual behaviour linked to payments, and the latter by supporting collective action.

7.1 Introduction

Conservation interventions ultimately aim to influence human behaviour, for example by encouraging people to change behaviours that cause biodiversity loss or degradation of ecosystems (St John et al., 2010). Common interventions include the enforcement of environmental protection regulations (for example in Protected Areas; Bruner et al., 2001), changing institutions (e.g. tenure reform; Acheson, 2006), or the provision of incentives (for example through Payments for Environmental Services, PES; Wunder, 2007). Critiques of conservation policies have often focused on their perceived failure to change human behaviour (Ferraro, 2001). Integrated Conservation and Development Projects (ICDPs), for example, were conceived and widely promoted in the 1980s and 1990s as a mechanism to conserve biodiversity by providing local people with alternative development assistance, based on the hypothesis that this would encourage them to change behaviours that led to over-harvesting of natural resources (Barrett & Arcese, 1995; Hughes & Flintan, 2001). This hypothesis was later found to be generally false because people chose to supplement rather than substitute their livelihoods with the new opportunities, thereby failing to induce any behaviour change (Barrett & Arcese 1995; Wells et al., 1999).

PES and direct payments for conservation were conceived in response to the perceived failure of ICDPs as an improved policy tool that directly linked the incentives offered to changes in behaviour (Ferraro, 2001; Ferraro & Kiss, 2002; Wunder, 2007). Similarly, conditional cash transfers have become popular in development policy, because they also link incentives to a behaviour change, such as school attendance (Fiszbein & Schady, 2009). PES provides positive incentives that are conditional on the provision of an environmental service, where successful implementation is based on a consideration of additionality and varying institutional contexts (Sommerville et al., 2009). Whilst the enthusiasm for PES has led to an explosion of direct payments for conservation (Milne & Niesten, 2009), PES and PES-like instruments (including reducing emissions from deforestation and forest degradation, REDD; Clements, 2010; Wunder et al., 2008), there has been little analysis of the extent to which PES does actually lead to a change in behaviour. Studies have suggested, for example, that observed behaviour changes may in fact be due to enforcement of regulations (Wunder et al., 2005), or monitoring of PES programme outcomes (Sommerville et al., 2010a), rather than the payments themselves.

How payments are managed and administered might affect how they are perceived, and then whether they bring about a change in behaviour (Sommerville et al., 2010b). Humans do not always behave “rationally” in accordance with economic theory (Persky, 1995); other motivations such as jealousy, social status, fairness or equitability and social norms have been shown to have a strong influence on behaviour both in laboratory experiments (games played with university students) and in real life (Henrich et al., 2001; Fehr & Falk, 2002; Henrich et al., 2005; Bowles, 2008). This body of evidence suggests that any analysis of behaviour has to be based upon an understanding of how payments influence the underlying psychological factors that affect how we behave. Analysing the effect of payments on psychological factors is however complicated because the relationships between participation in a programme, self-reports of perceptions and attitudes, and then expressed behaviours are essentially endogenous. The most established framework, the Theory of Planned Behaviour, suggests that human behaviour can be predicted from three underlying factors: our attitudes, social norms, and perceived ability to control the behaviour determines our behavioural intentions, which then leads to actual behaviour (Ajzen, 1991).

This study investigated the attitudes and behaviour of local people in northern Cambodia following the implementation of three PES programmes (described in Chapter 2). The comparison between programmes is of interest because they were designed and managed in very different ways, and the extent to which they then brought about changes in local behaviour is informative for the design of future PES programmes. The three PES programmes were: (*1*) direct payments to local people conditional upon protection of nests of globally threatened birds (the Bird Nests programme); (*2*) a community-managed ecotourism programme that linked income to bird and habitat protection (the Ecotourism programme); and (*3*) providing farmers with premium prices for agricultural goods if they limited field expansion to within agreed land-use plans (Ibis Rice). Payments from the Ecotourism and Ibis Rice programmes have led to significant improvements in local livelihoods and household wellbeing for participants (see Chapter 6). Previous research has indicated that all three payment programmes did deliver additional environmental outcomes at the village scale: the Bird Nests programme significantly reduced hunting rates for globally threatened bird species (Clements et al., 2012b; see Chapter 4), and the Ecotourism and Ibis Rice programmes significantly reduced rates of local deforestation (see Chapter 5). This chapter focuses on individual behavioural responses to the programmes. The objectives were (*1*) to describe the distribution of payments to households from the three PES programmes; (*2*) to understand whether these household-level payments changed individual behaviour; (*3*) to investigate how the institutional arrangements of the PES programmes influenced attitudes toward the programmes; and (*4*) to analyse how the attitudes towards the payment programmes then influenced behaviour, based partially on the predictive framework established by the Theory of Planned Behaviour.

7.2 Methods

Payment Programmes

The three PES programmes were designed by the Wildlife Conservation Society (WCS) to complement other conservation interventions in the landscape, such as management of two Protected Areas and participatory processes to determine land-use boundaries for local villages

(Clements et al. 2010). Both protected areas contained substantial human populations in 16 villages, which pre-dated the establishment of the parks and were not resettled. The PES programmes were designed to engage these people in conservation activities. The Ecotourism programme was piloted in the village of Tmatboey on a small-scale during 2005-2008 (for a map see Figure 7.1), and was then scaled up in that village and expanded to include Dangphlat (2008 onwards). The Ibis Rice programme was initiated in late 2008 or early 2009 in four villages: Tmatboey, Dangphlat, Prey Veng and Narong. The Bird Nests programme started in 2003, and by 2009 was operating in 24 villages across the landscape, including the four villages where the Ecotourism and the Ibis Rice programmes were operating. Whereas the Bird Nests programme was entirely managed by WCS, the Ibis Rice and Ecotourism programmes were managed locally by elected village committees, who were legally responsible for managing land-use around the village. The process of determing the village land-use boundaries took approximately two years, and ended up with a final product that was agreed by all the village residents with the Protected Area authorities. The Bird Nests and Ibis Rice programmes provided conditional payments to individuals, whilst the Ecotourism provided payments both at the village level and to some individuals.

This study focuses on eight villages: the four core villages where the Bird Nests and Ibis Rice PES programmes operated, including two villages (Tmatboey and Dangphlat) that also had Ecotourism, and four further villages where local people only received payments for Bird Nest protection (see map in Figure 7.1). The four core villages and four additional villages were split evenly between the two protected areas. In each of the four core villages, a database of all households was compiled over a period of two weeks in each village in 2009, using the official village lists, with additions for new families. The number of households was 237 in Tmatboey, 187 in Dangphlat, 117 in Narong, and 75 in Prey Veng (n = 616). Each household was identified using the names of several senior members (household head and at least one other) in Khmer script, in order to facilitate cross-referencing. Households were defined as the group of people who lived in one house and cooked together. All payments made from the three PES programmes from mid-2008 (when the payments started) until mid-2011 (three seasons of payments) were recorded in a database and assigned to the correct household in the village where that could be identified. Payment households were correctly identified in 96% of cases; most unidentified payments were minor (e.g. small payments to service providers from the ecotourism programme). The payment dataset was investigated to determine the level of participation in each programme, who benefited, the distribution of payments made, and the level of overlap between participation in the programmes.

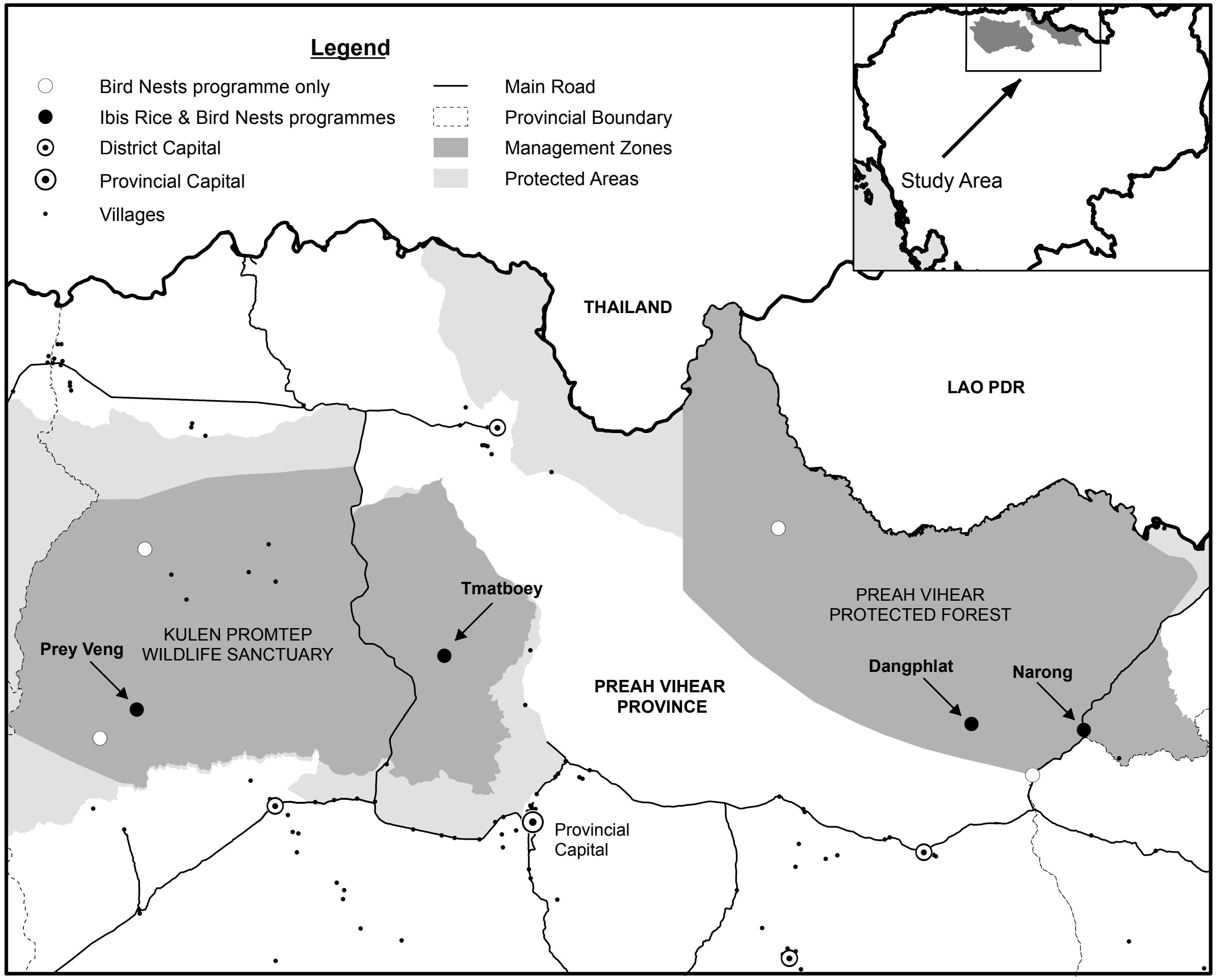


Figure 7.1. Map of Preah Vihear province, Cambodia, showing the two Protected Areas (Kulen Promtep Wildlife Sanctuary and Preah Vihear Protected Forest) and their Management Zones, and the location of the eight villages selected for the study of household attitudes and behaviour. Four of the villages only had the Bird Nests programme in operation, whereas four had both the Bird Nests and the Ibis Rice programme. Tmatboey and Dangphlat (locations shown by arrows on the map) also benefited from the community-based Ecotourism programme.

Understanding individual human behaviour

The Bird Nests and Ibis Rice payment programmes specifically targeted individual hunting and land clearing behaviours respectively, whilst the Ecotourism programme targeted both behaviours at the individual level (for participating households, who agreed to a code of conduct) and village level (Table 7.1). Whereas the Bird Nest and Ibis Rice payments were strictly conditional on participants' behaviour, households participating in the Ecotourism programme were only asked to agree to a code of conduct. The Ecotourism payments then came from the tourists, and hence were only weakly linked to a biodiversity conservation outcome. The Ecotourism and the Ibis Rice programmes were managed at the village level, whereas WCS administered the Bird Nests programme. Previous chapters have analysed the changes in behaviours at the village-level (see Chapters 4 and 5), so only changes in individual behaviour are analysed here. At the individual level, the only behaviour that could be reliably measured was household land clearance, primarily because involvement in other behaviours (such as hunting) was hard to verify.

Table 7.1. The human behaviours targeted by the three payment programmes, and the scale at which changes in behaviour were assessed.

Behaviour targeted	Bird Nests	Ecotourism	Ibis Rice	Measurement of change
Hunting of birds	Yes, monitored	Yes, monitored	No	Village level (Chapter 4)
Land clearance (compliance with land-use plans)	No	Yes, monitored	Yes, monitored	Individual & Village level (Chapter 5)
Acceptance of immigrants	No	Not explicitly	Not explicitly	Village level (Chapter 5)
PES payments target behaviour change at	Individual level	Individual & Village level	Individual level	
PES programme managed by	WCS	Village authorities	Village authorities	

Measuring individual land-clearing or hunting behaviour was challenging because these behaviours were illegal. Various techniques can be used to measure illegal behaviour, including self-reports, arrest records, independent monitoring, and novel methods such as the randomised response technique (Solomon et al., 2007; St John et al., 2011) or choice experiments (Moro et al., 2012). Most of these methods have problems: self-reports may be biased (St John et al., 2011); arrest records fail to identify the level of undetected illegal behaviour; and choice experiments may not capture real behaviour. Consequently this study focused only on land-clearing behaviours because these could be independently verified through field visits.

Land clearing behaviours by individual households were monitored over three years, from 1 January 2008 until 31 December 2010, in the four core villages (Tmatboey, Dangphat, Prey Veng and Narong) that were the first to develop land-use plans and where the PES programmes were implemented. In 2008, the village land-use boundaries were agreed with and approved by the Protected Area authorities and local Government, so the dataset covers the period before the approval of land-use boundaries and the two subsequent years. Land clearance outside the approved boundaries was classified as illegal. Four main data sources were used to monitor illegal land clearing behaviour: (*i*) reports from the Village Committees that were responsible for management of and negotiating the land-use boundaries; (*ii*) WCS-employed community facilitation teams that were responsible for assisting the Village Committees with the Ibis Rice and Ecotourism PES programmes and management of the land-use plans; (*iii*) reports from research teams that were collecting data on bird nests, wildlife, and encounters with illegal activities (see Chapters 4 and 5); and (*iv*) records from protected area rangers based upon enforcement action taken around the study villages. Reports from the research teams and protected area rangers were then verified in discussion with the Village Committees and field visits, and assigned to a particular household in the village.

This dataset was then used to investigate trends in households' illegal land-clearing behaviour, and the impacts of the PES programmes on this behaviour. Households were categorised into participants and non-participants in each of the three payment programmes, in each year (2008, 2009 and 2010) and for each village. The proportion of households clearing land was calculated for each cell of the contingency table. This dataset was analysed using generalised linear models with quasibinomial errors in R 2.14.2 (R Development Core Team, 2012) to investigate the effects of the payment programmes and the year on the proportion of households clearing land. A second contingency table was then prepared that combined data across all three years, recording the proportion of households that cleared land in any of the three years, and analysed in the same way.

The effects of household characteristics and socio-economic status on individual land-clearing behaviour were investigated for a subset of households (n = 340) for which data was available from individual household surveys collected in 2011 (see Chapter 6; Chapter 3 has more details on the different data collected from each of the households). Data were analysed using logistic regression, where the dependent variable was whether or not a household had cleared land, and the explanatory variables included participation in the payment programmes, household characteristics (size, number of working adults, education, etc) and socio-economic status (measured as both poverty status using the Basic Necessities Survey score and the household rice harvest). The significance of individual model terms was assessed by deleting each in turn from the model, and comparing the change in deviance using chi-squared (Crawley, 2007). As with the analysis of proportions, two models were constructed: one with data on the behaviour of each household in each year (2008, 2009 and 2010) using a mixed effects model with household as the random effect, and one with data on the behaviour of each households across all three years.

Local attitudes and perceptions of the PES programmes

Local attitudes towards conservation, the PES programmes, and key issues such as land management and immigration, were assessed through semi-structured interviews in all eight villages between December 2009 and January 2010. Interviews were carried out by the Center for Development Orientated Research into Agriculture and Livelihood Systems (CENTDOR), a Cambodian social research organisation specialising in impact evaluation, in order to limit the potential biases associated with foreign researchers or staff associated with WCS, who were well known in some of the villages. The interview questionnaire was developed based upon the main research questions and was extensively tested by the authors and CENTDOR staff in another village with the Protected Areas, Chaomsre, that had not been visited previously by the author. In total, 464 households were interviewed from the eight villages, or about 42% of the population. Interview subjects were chosen at random based upon a stratified list of households in each of the villages, divided into households participating in one or more of each of the PES programmes and non-participants.

The effect of household attitudes and participation in the PES programmes on land clearance behaviour was investigated through the same questionnaires using a set of standardised questions focusing on underlying social norms, household attitudes towards the PES programmes, and household perceptions of key actors such as the village committee and the protected area authorities. Underlying social norms were assessed using the standard General Social Survey questions, which have been used extensively for this purpose and have been shown to predispose people towards certain behaviours (Glaeser et al., 2002; Karlan, 2005). The three General Social Survey questions are: the trust question: “Generally speaking, would you say that most people in the village can be trusted or that you can't be too careful in dealing with people?”; the fairness question: “Do you think most people in the village would try to take advantage of you if they got a chance, or would they try to be fair?"; and the helpful question: "Would you say that most of the time people in the village try to be helpful, or that they are mostly just looking out for themselves?" Household attitudes and perceptions were assessed using rankings, Yes/No responses, and through qualitative statements which were then coded to group similar responses.

Two of the three PES programmes were managed locally by village groups (Ecotourism and Ibis Rice), whilst one (the Bird Nests) was managed by WCS. All 8 study villages were also subject to the protected area authorities. The attitudes of the villagers towards these internal and external actors was investigated using institutional profiling (Holland, 2007), a participatory rural assessment exercise that was conducted with focus groups in each village. In the exercise, villagers were asked to describe the key internal and external actors in their village, and to rank both the influence of these actors on their lives and whether or not they thought the behaviour of the actor was aligned with their interests or not. Responses were recorded on a map with coloured pieces of paper placed close or far from the village, to reflect the extent to which the actor was aligned with the village's interests, with the size reflecting the relative power of the actor and the colour reflecting whether their influence was positive or negative. The qualitative responses were then transcribed into a table of relative rankings. Two focus groups were conducted in each of the 8 villages, with the exception of Tmatboey where three were completed. Approximately 10-15 people joined each discussion, with a total of 215 participants across the villages, of whom 76 were women.

The influence of local attitudes and perceptions on behaviour was analysed for the subset of households in the four core villages where all the PES programmes were operational (Tmatboey, Dangphlat, Prey Veng and Narong) and for which data on real behaviour were available. The total sample size was 272 households (86 in Tmatboey, 75 in Dangphlat, 46 in Narong and 65 in Prey Veng). Data were analysed using a generalised linear model with a binomial link function, where the response variable was whether or not a household had cleared land illegally at any time over the three years (2008-2010). The explanatory variables included household socio-economic characteristics (household size, agricultural productivity, poverty status, etc), the social norms based upon the General Social Survey responses, household attitudes towards the PES programmes and village rules, rankings of the influence of key actors on household decision-making, and whether or not a household participated in each of the three PES programmes. The initial model fitted included all the main effects, and model simplification proceeded by removing variables based upon chi-squared tests of the change in deviance. Village was included as a main effect in all the models. Three further models were then developed for the subset of households that knew about the Ibis

Rice programme (n = 172 from 4 villages), the Ecotourism programme (n = 108; 2 villages) and the Bird Nests programme (n= 208; 4 villages), to investigate the effect of the perceived benefits and fairness of those programmes on household behaviour. For these models, whether or not a household participated in each of the three PES programmes was excluded because the intention was to investigate perceptions and attitudes towards the programmes, and how these influenced behaviour, rather than analysing participation.

7.3 Results

What were the benefits from the PES programmes, how were these benefits distributed and to whom?

Over the three years 409 payments from the PES programmes, totalling over \$103,000, were recorded to 214 of the 616 households in the four villages. 35% of households benefited from at least one of the programmes in at least one of the years. The average payments were significant in comparison with the estimate of household consumption in rural forested regions from the 2007 Cambodia Socio-Economic-Survey of \$329±16 (World Bank, 2009). Ibis Rice provided the highest annual payments to the most households (Figure 7.2a, Table 7.2), with 17% of households participating in 2010 and 41% participating in more than one year. The Ecotourism programme benefited fewer households (9% in 2010, Table 7.2) at a lower annual payment, but households were very likely to be engaged in more than one year so the total payment participating households received over the three year period was equivalent to Ibis Rice (Figure 7.2b, Table 7.2). The Bird Nests programme benefited households the least; due to the scarcity of nests only a few households participated (4% in 2010, Table 7.2), households were unlikely to be engaged in more than one year, and the payment level was low (Figure 7.2c). The distribution of payments under the Bird Nests programme (Gini coefficient = 0.34) and Ecotourism programme (Gini coefficient = 0.32) was more equitable than for the Ibis Rice programme (Gini coefficient = 0.47; Figure 7.2), since the first two programmes rewarded the level of effort whereas Ibis Rice linked payments to productivity, favouring households with larger fields.

The Ecotourism programme provided both household payments and contributions into a local village development fund based upon the number of tourists that saw key wildlife species. The latter raised nearly \$13,000 over the three year period, mostly in Tmatboey (Table 7.3). As a consequence the total payments to each village from the Ecotourism and Ibis Rice programmes were very similar.

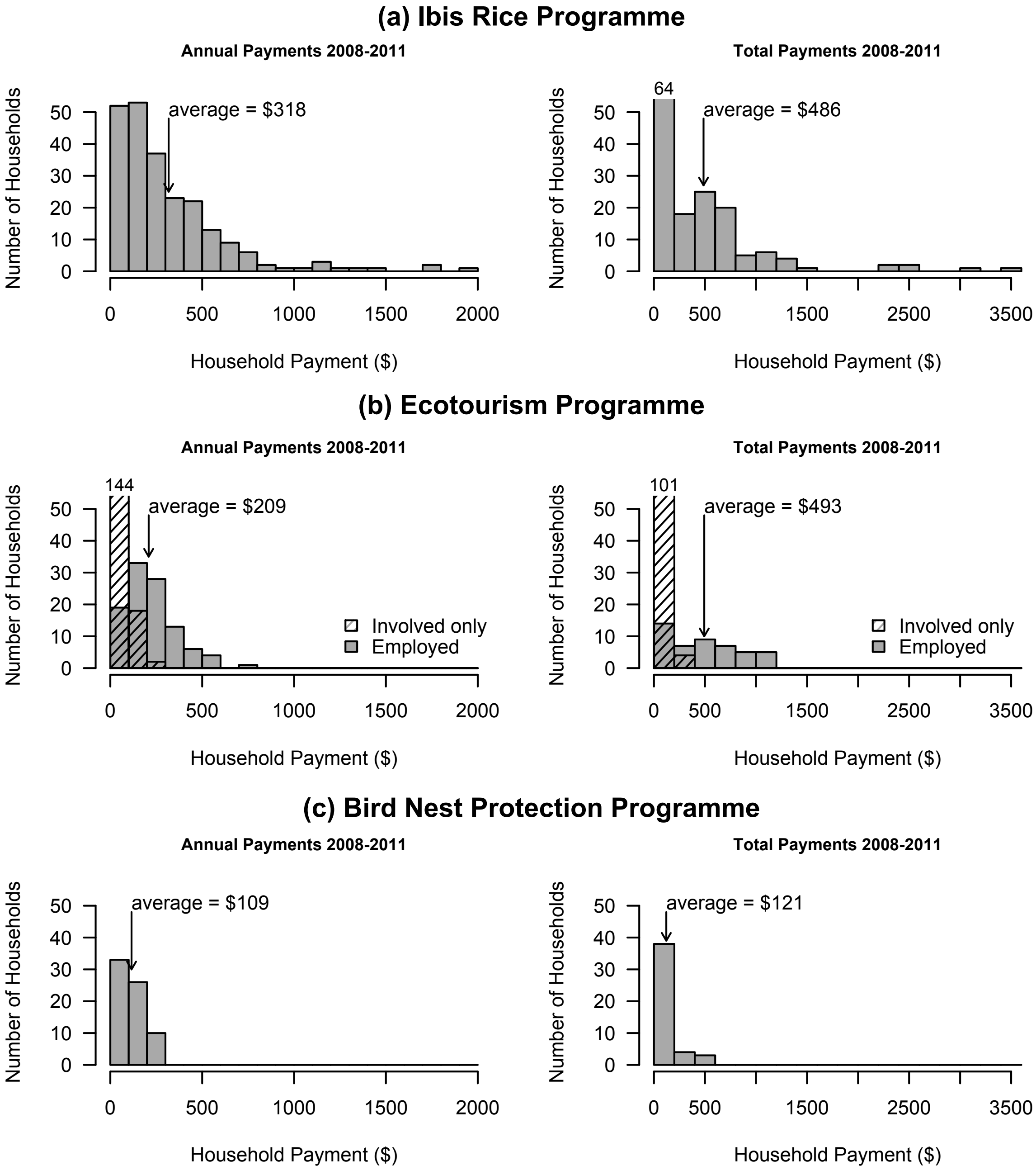


Figure 7.2. Distribution of the annual payments and total payments over three years to households from (a) the Ibis Rice programme (4 villages; n = 228; 17% of households in 2010); (b) the Ecotourism programme (2 villages); and (c) the Bird Nests protection programme (4 villages; n = 69; 4% of households in 2010) over three seasons (2008-2011). For the Ecotourism programme beneficiaries are divided into households that were employed under the programme (n = 104; 9% of households in 2010), and those who benefited indirectly by selling goods and services to the village-managed enterprise (n = 158; 10% of households in 2010).

Table 7.2. Payments made to households in four villages during 2008-2011 (three seasons) under the three payment programmes: Bird Nests protection, Ibis Rice and Ecotourism. The table gives the total number of households that received payments over the three years (noting that not all households benefited in every year), the percentage of households receiving payments during the season 2010-2011, the average payment made in a year and the standard error, and the Gini coefficient as a measure of the inequality. Ecotourism beneficiaries are divided into households that were employed directly, and those who benefited indirectly by selling goods and services to the village-managed enterprise. On average, 7% of households in the villages were female-headed.

		All	Tmatboey	Dangphat	Prey Veng	Narong
Bird Nests	Number of Payments	69	34	18	10	7
	% of Households paid in 2010	4%	5%	3%	4%	2%
	% of Households paid >1 year	13%	13%	8%	n/a	0%
	% of Female-headed households	10%	13%	10%	0%	0%
	Total Payments	$2,503	$874	$677	$410	$543
	Total Household Payments	$121 ($3)	$86 ($2)	$110 ($10)	n/a	$121 ($10)
	Average Annual Household Payment	$109 ($8)	$77 ($6)	$113 ($15)	$123 ($23)	$233 ($9)
	Gini Coefficient of Annual Payments	0.34	0.26	0.32	0.30	0.05
Ibis Rice	Number of Payments	228	97	58	25	48
	% of Households paid in 2010	17%	17%	11%	20%	26%
	% of Households paid >1 year	41%	54%	43%	26%	27%
	% of Female-headed households	5%	9%	3%	0%	4%
	Total Payments	$24,131	$13,991	$3,674	$4,093	$2,374
	Total Household Payments	$486 ($4)	$750 ($13)	$298 ($7)	$192 ($5)	$646 ($38)
	Average Annual Household Payment	$318 ($21)	$433 ($36)	$190 ($16)	$491 ($86)	$148 ($17)
	Gini Coefficient of Annual Payments	0.47	0.39	0.36	0.46	0.39
Ecotourism (Employed)	Number of Payments	104	59	45		
	% of Households paid in 2010	9%	9%	8%		
	% of Households paid >1 year	68%	62%	78%		
	% of Female-headed households	19%	18%	20%		
	Total Payments	$7,725	$4,823	$2,902		
	Total Household Payments	$493 ($7)	$499 ($12)	$484 ($19)		
	Average Annual Household Payment	$223 ($13)	$245 ($18)	$193 ($19)		
	Gini Coefficient of Annual Payments	0.32	0.31	0.33		
Ecotourism (Involved only)	Number of Payments	158	114	44		
	% of Households paid in 2010	10%	15%	5%		
	% of Households paid >1 year	33%	48%	7%		
	% of Female-headed households	5%	7%	3%		
	Total Payments	$1,890	$492	$1,398		
	Total Household Payments	$53 ($1)	$22 ($0)	$102 ($2)		
	Average Annual Household Payment	$36 ($5)	$13 ($1)	$95 ($13)		
	Gini Coefficient of Annual Payments	0.66	0.45	0.47		

Table 7.3. Payments to the Village Development Funds from the Ecotourism programme over three seasons from 2008-2011. The total amount given to the village from the Ecotourism programme is therefore similar to those provided by Ibis Rice, when the household and village-level payments are combined.

	All	Tmatboey	Dangphat
Total Payments to Village Fund	\$12,953	\$10,373	\$2,580
Average Annual Payment to Village Fund	\$4,318	\$3,458	\$860

The Ecotourism also benefited other households in the villages that sold goods and services to the village-managed enterprise but were not fully engaged. This accounted for a further 158 payments over the three years, or an additional 10% of households (Figure 7.2b, Table 7.2). These households were not expected to adhere to the code of conduct developed for the Ecotourism programme.

It has been shown (see Chapter 6) that there were access constraints to participation in two of the three PES programmes, demonstrated by significant differences between the characteristics of participants and non-participants. Ibis Rice participants were less poor and had greater agricultural productivity than other households, and Ecotourism households had greater agricultural productivity. Participants in the Bird Nests programme were similar to non-participants, probably because part of the payment was provided up-front, allowing any household to take part in the programme. The Ecotourism included specific provisions for poor households and female-headed households to access the programme (19% of participants were female-headed households, as opposed to 7% of households in the villages, Table 7.2), whereas there is no evidence for additional participation of female-headed households in the other two programmes.

In addition to the access constraints, there is evidence that village elites were more likely to participate in the Ibis Rice programme (Figure 7.3). Seven of the nine Government officials who lived in the four villages participated in the Ibis Rice programme, whereas only two of the nine would have been expected to participate if participation was random with respect to status. Both the Ibis Rice and particularly the Ecotourism programme were dominated by members of the Village Land-use Committee. Given that these committees were responsible for local management of these programmes the high level of participation by committee members is unsurprising, but is perhaps indicative of committee members valuing individual benefits over their elected mandates.

As a consequence of the access constraints and the preferential participation in some of the programmes by village elites, participation in the Ecotourism and Ibis Rice programmes was not independent. Households participating in the Ecotourism were significantly likely to also be participating in Ibis Rice (χ^2 = 6.59, df = 1, P < 0.05). By contrast, households participating in the Bird Nests programme had the same chance of participating in the Ecotourism or Ibis Rice as any other household (χ^2 = 1.253, df = 1, P = 0.263; and χ^2 = 1.391, df = 1, P = 0.238).

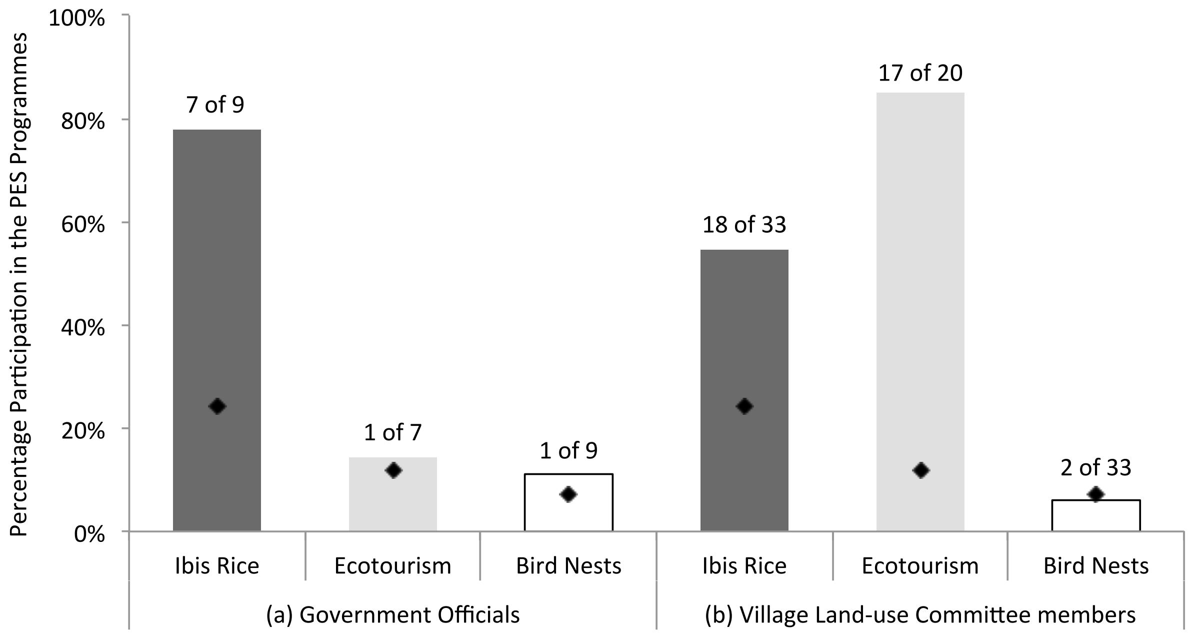
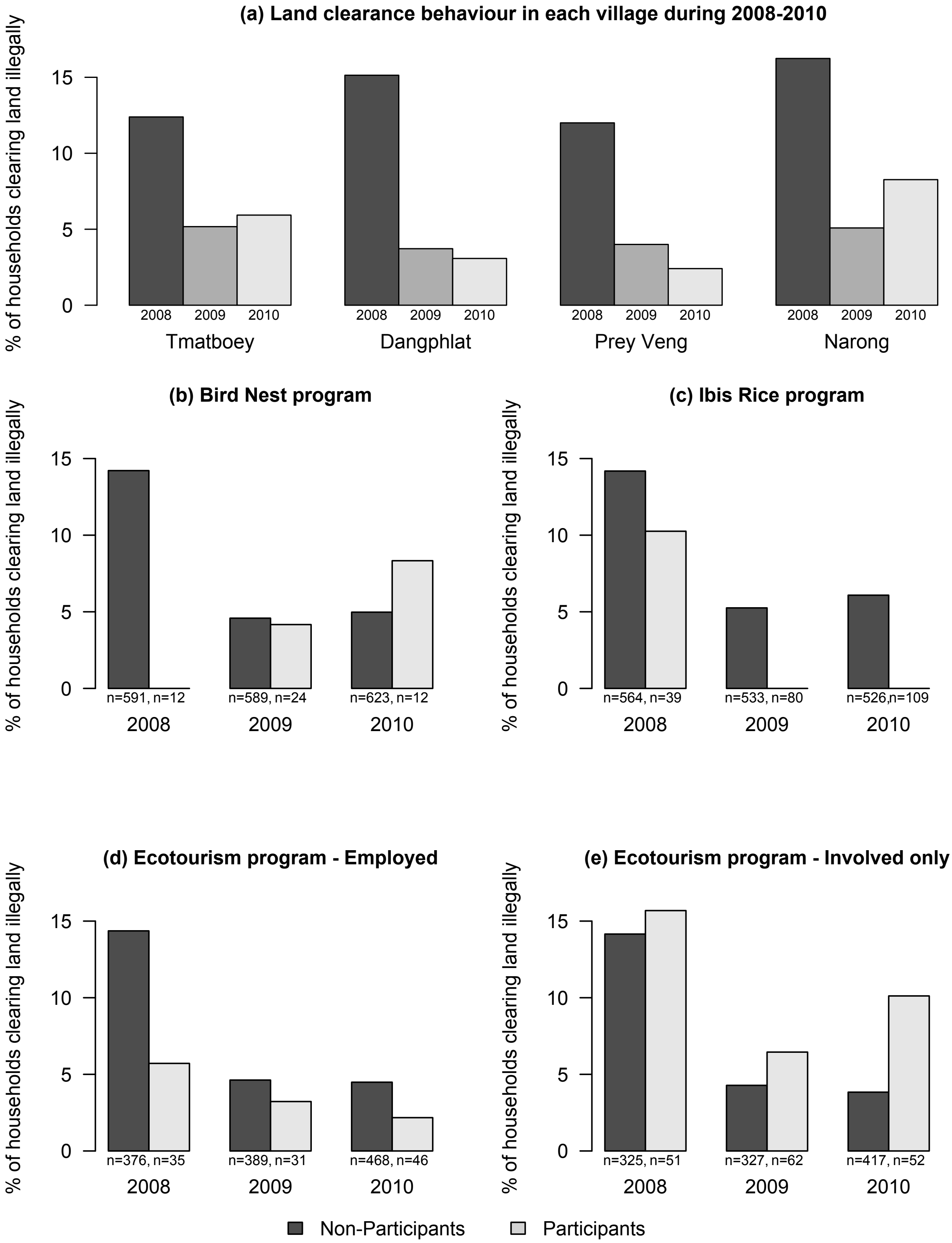


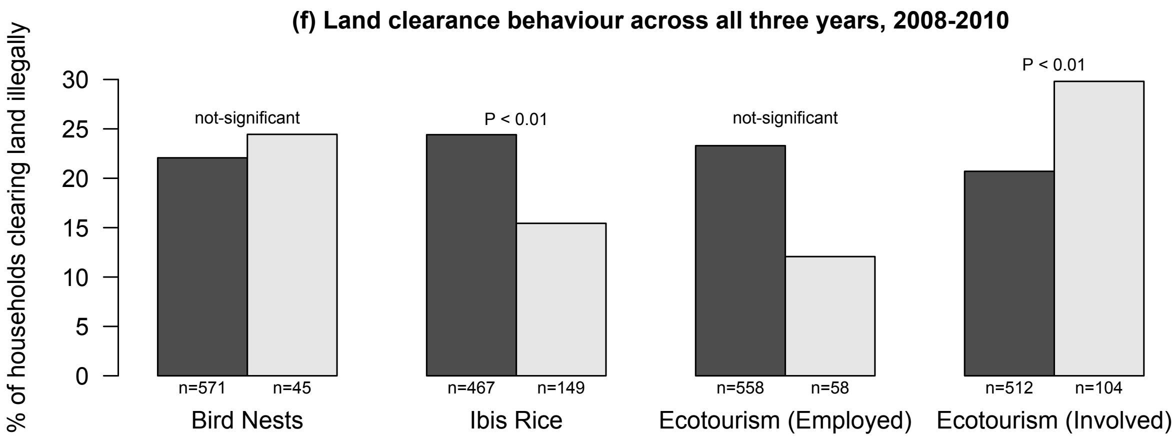
Figure 7.3. Participation by village elites in the PES programmes. The bars show the percentage of (a) Government officials and (b) Village Land-use Committee members in the village who participated in each of the three PES programmes. Data labels give the number participating, out of the total number of elites in the villages. Symbols mark the expected number of participants from village elites, if participation was random with respect to household occupation.

Did payments change individual behaviour?

A large number of households were illegally clearing land in 2008 (14%), which was just after land-use boundaries were approved and when the Ecotourism and Bird Nests programmes were operational but before Ibis Rice was initiated. In 2009 and 2010, the overall percentage of households illegally clearing land was significantly less (5%; Figure 7.4a and Table 7.4). The difference between the number of households illegally clearing land in 2008, and the number clearing land in 2009 and 2010 were similar and highly significant for all four villages (Figure 7.4; Tmatboey: χ^2 = 8.432, df = 1, P < 0.01; Dangphlat: χ^2 = 29.352, df = 1, P < 0.001; Prey Veng: Fisher's Exact Test P < 0.05; Narong: χ^2 = 5.661, df = 1, P < 0.05).

Figure 7.4. Land-clearing behaviour of households for participants and non-participants in the three PES programmes in 2008, 2009 and 2010 in four villages in northern Cambodia. Figure (a) shows the rates of land clearance in each village in each year. Figures (b)-(d) give the results for the Bird Nests, Ibis Rice and households engaged in the Ecotourism programme respectively, and Figure (e) for households that sold goods and services only to the Ecotourism programme. Figure (f) shows the results aggregated across all three years, recording whether a household had engaged in land clearance at any point during that period. The land-use plans were approved in 2008 and the Ecotourism and Ibis Rice programmes became operational at the same time. Illegal land clearance subsequently declined in 2009-2010.





Ibis Rice provided payments at the end of the calendar year if the household had not cleared land illegally. It had the greatest impact on household behaviour (Figure 7.4c and 7.4f), significantly reducing rates of land clearance both for the year a household received a payment (Table 7.4a and 7.4c) and across all three years even if the household was not paid in every year (Table 7.4b and 7.4d). None of the participating households illegally cleared land in 2009 or 2010, a 100% compliance rate (Figure 7.4c). For the Ecotourism programme, which provided continual payments during the tourist season (November-May) the data suggest that participating households were also much less likely to clear land illegally (Figure 7.4d and 7.4f), however this effect was not significant in any year or overall (Table 7.4a and 7.4b). The observed differences are due to the fact that many of the Ecotourism households were also participating in Ibis Rice, and these households were much less likely to clear land; the effect of Ibis Rice is sufficient alone to explain this (Table 7.4a and 7.4b). There is some evidence that households that sold goods and services to the Ecotourism but were not formally engaged were more likely to undertake illegal land clearance (Figure 7.4e and 7.4f; Table 7.4). The Bird Nests programme, which did not target land clearance, had no discernible effect on household land clearance behaviour (Figure 7.4b and 7.4f, Table 7.4).

The socio-economic status of households that were illegally clearing land was very similar to households that did not clear land, for the 340 households for which data was available (Table 7.5). Any differences were not significant either in terms of the average difference (measured using a *t*-test; Table 7.5) or the distribution of values (Kolmogorov-Smirnov test; Table 7.5). For this reason matching statistics were not calculated (because the samples were already well matched). Unsurprisingly, therefore, the same characteristics of the households and their socio-economic status had very limited effect on a household's decision to clear land when the effects were modelled (Table 7.4c and 7.4d). Those households that did not farm were significantly less likely to

Table 7.4. The effects of year and the payment programmes on whether or not a household chose to illegally clear land during 2008-2011, (a) and (b) for the entire dataset (*n* = 616 households), using generalised linear models; and (c) and (d) for a reduced dataset of households for which socio-economic characteristics were also known (*n* = 340), using logistic regression.

(a) and (b) analysis of the effects of the three different payment programmes on the proportion of participating and non-participating households illegally clearing land in three years (2008-2011), using a generalised linear model with quasibinomial errors.

	(a) In each year		(b) Across all years	
	Estimate	Significance	Estimate	Significance
Village : Dangphlat	-1.876		-1.401	
Village : Narong	-1.501		-0.804	
Village : Prey Veng	-2.042		-0.972	
Village : Tmatboey	-1.845		-1.383	
Year : change from 2008 to 2009	-1.197	***		
Year : change from 2008 to 2010	-0.963	***		
Ecotourism (Employed)	-0.591		-0.468	
Ecotourism (Involved only)	0.506	(*)	0.642	**
Bird Nests, participant	-0.555		0.147	
Ibis Rice, participant	-1.451	**	-0.620	**

(c) and (d) analysis of the effects of household characteristics and socio-economic status on whether or not a household chose to illegally clear land, using a logistic regression model based upon the reduced dataset for which household-level data was available (from the 2011 surveys).

	(c) In each year		(d) Across all years	
	Coefficient	Significance	Coefficient	Significance
Village : Dangphlat	0.420		0.933	
Village : Narong	0.429		1.314	
Village : Prey Veng	0.102		1.561	
Village : Tmatboey	0.903		1.485	
Year : change from 2008 to 2009	-1.307	***		
Year : change from 2008 to 2010	-1.244	***		
Ecotourism (Employed)	-0.969		-1.304	**
Ecotourism (Involved only)	-0.040		0.183	
Bird Nests, participant	-0.710		-0.055	
Ibis Rice, participant	-1.802	**	-0.915	**
Household has > 1 hectare of land	-0.869	(*)	-0.895	(*)
Household Poverty Status	0.036		0.053	
Household Rice Harvest (square-root)	-0.018	(*)	-0.017	
Rice Farmer Type [None]	-3.335	*	-3.024	**
Rice Farmer Type [Shifting Cultivation only]	-1.183		-0.242	
Rice Farmer Type [Paddy only]	-0.865		-0.930	(*)
Random effect of households: % residual variation	33.5%			

Significance values: ns = Not-significant; * = $P < 0.05$; ** = $P < 0.01$; *** = $P < 0.001$

Table 7.5. Comparison of the socio-economic characteristics of households that chose to illegally clear land and those that did not, using covariate balancing tests. The two samples are statistically very similar.

Variable	Poverty status in 2011 (BNS Score)	Rice Harvest in 2011 (kg, square-root)	Food Security in 2011 (kg, square-root)
Mean Households clearing land illegally	12.2	47.0	79.2
Mean Households not clearing land illegally	12.6	50.3	80.8
T-test p-value (difference of means)	0.385	0.109	0.166
Kolmogorov-Smirnov test p-value (difference in the distribution)	0.511	0.284	0.212

clear land illegally (Table 7.4c and 7.4d). There was also some evidence that households which already had more than one hectare of land were less likely to clear land illegally (Table 7.4c and 7.4d). Once the socio-economic characteristics of households are taken into account, participants in the Ecotourism programme were much less likely to clear land illegally across the three years (Table 7.4d), although payments from the Ecotourism programme itself had no discernable impact in the year that they were made (Table 7.4c).

The models of household socio-economic characteristics explained only some of the variation in household behaviour, with a significant proportion of the variance (33.5%) left unexplained due to differences between households. This suggests that other factors, which were not related to socio-economic status, had a strong influence on household behaviour.

How were the PES programmes understood and perceived?

There was a high level of awareness and knowledge about the three payment programmes in those villages where the programmes were providing significant benefits (67-78%; Table 7.6). Of households that knew about the PES programmes, the majority could accurately describe the programme and how it operated. There were significant differences between how the programmes were understood and perceived. The conditionality attached to the payments was easiest to understand for the Bird Nests programme, the simplest of the three, and was most weakly perceived for the Ecotourism. The Bird Nests was known to target wildlife protection, whilst the Ibis Rice was known to target land clearance behaviour, and the Ecotourism was perceived to focus on both behaviours in Tmatboey (but only forest protection in Dangphlat). The programme

Table 7.6. Understanding and perceptions of local people towards the three PES programmes, based on a sample of 464 households interviewed across 8 villages, with varying degrees of involvement in the programmes. The level of involvement was defined based upon the total payments provided to each village from the programme. Cells shaded in grey indicate the correct answers for the questions about the programme conditions and management.

			Bird Nests		Ibis Rice		Ecotourism	
Variable	Question	Response	Lower	Higher	Lower	Higher	Dangphlat	Tmatboey
Number of Villages			3	5	2	2	1	1
Payment levels in US$/village/year			<$100	>$400	<$4000	>$4000	$3,700	$7,500
Interviews (% of households in the villages)			160 (47%)	304 (41%)	118 (61%)	162 (39%)	76 (41%)	86 (38%)
Knowledge of the programme			74 (46%)	238 (78%)	55 (47%)	112 (69%)	51 (67%)	58 (67%)
For the households that know about the programme:								
Programme Conditions	Can describe correctly?	Yes	91%	91%	49%	66%	63%	65%
	Protect Forest?	Yes	0%	0%	49%	64%	51%	54%
	Protect Birds?	Yes	91%	91%	29%	29%	14%	59%
Perceptions?	Who manages the programme?	Village Committee	28%	36%	89%	81%	96%	97%
		Villagers	20%	23%	13%	5%	6%	0%
		NGO	62%	71%	20%	48%	37%	35%
	Who can participate?	Anyone	100%	100%	93%	70%	33%	54%
	Is the programme fair?	Yes	96%	94%	85%	79%	59%	62%
Who benefits?	Village Authority?	Yes	0%	0%	0%	0%	0%	0%
	Village Committee?	Yes	0%	5%	11%	7%	37%	35%
	NGO?	Yes	0%	6%	2%	0%	0%	0%
	Individual households?	Yes	93%	93%	87%	97%	80%	86%
	Village?	Benefit a lot	28%	67%	51%	72%	84%	81%
		No benefit	71%	30%	49%	22%	14%	18%
		Lose out	1%	3%	0%	5%	3%	1%

management was clearly understood: the Bird Nests was administered by WCS, an external non-government organisation, whereas both the Ibis Rice and the Ecotourism were managed locally by the Village Committee. There were significant differences in perceptions of the three programmes regarding who benefited, and how those benefits were distributed (Table 7.6). The Bird Nests programme was perceived to be open to participation by anyone, with a fair distribution of benefits, and an absence of elite capture.

Similar perceptions were associated with the Ibis Rice programme, although it was understood that there were some restrictions on participation (namely that households needed to have sufficient produce to sell to the programme). The Ecotourism was widely viewed to be the least fair and to have the most restrictions on participation, and with a significant share of the benefits being captured by the Village Committees. Nevertheless, the Ecotourism was also thought to benefit the village the most, due to the payment structure that provided significant income to the village fund as well as individual payments to a small number of households (Table 7.6). In summary, the knowledge and perceptions of the villagers of the programmes were broadly accurate and consistent with the way the programmes were administered and the actual distribution of the payments (Figures 7.2 and 7.3, Tables 7.2 and 7.3). The ranking of the individual and collective (village-level) benefits of the programmes by local people in the two villages most familiar with the programmes (Tmatboey and Dangphlat) confirmed these results. The Ibis Rice programme was ranked as providing the greatest individual benefits, whilst Ecotourism provided the greatest village-level benefits, and the Bird Nests programme scored low against both criteria (Figure 7.5).

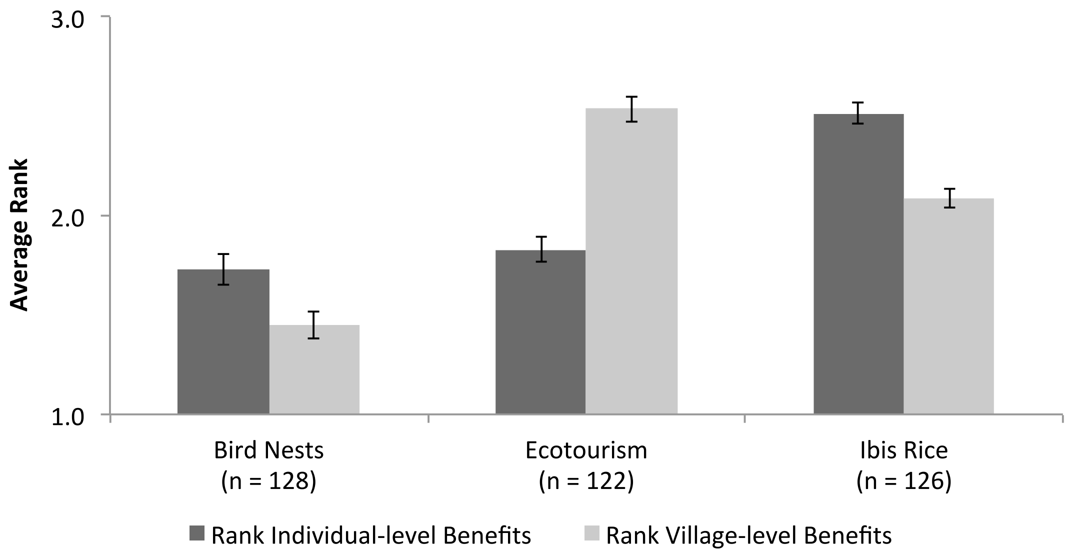


Figure 7.5. Perceived ranking of the benefits on a scale of 0 (no benefits) to 3 (significant benefits) of the three PES programmes, from an individual and a collective (village) level. Results are based upon respondents in the two villages (Tmatboey and Dangphat) where all three programmes had been operational for three years or more.

How did payments influence attitudes towards institutions?

The attitudes of local people towards the Protected Area institutions varied depending upon the extent to which the payment programmes had been implemented in their village (Table 7.7). People in the four villages with only the Bird Nests programme mostly disliked the village land-use planning committee and the protected area authorities, and saw these actors as unaligned with the village's interests. By contrast, local people in the two villages where all three payment programmes were in operation (Dangphlat and Tmatboey), had a more neutral attitude towards the village land-use planning committee and the protected area authorities (Table 7.7). The Ecotourism and Ibis Rice payment programmes also empowered the village land-use planning committee, through its management of the payment programmes, and the committee was therefore seen as having a much stronger influence over the villagers' lives. The village committees were known to enforce land-use plan regulations if households had cleared land without permission (Table 7.8). The Ecotourism and Ibis Rice payment programmes were generally liked, seen as being in alignment to some extent with the villages' interests, and also had a strong influence over local people's livelihoods. The Bird Nests programme was not mentioned by any of the participants in the focus groups.

Why did households decide not to clear forest land?

Both the focus groups and the household interviews indicated that two key actors affected household decisions to clear forest land: the village land-use committee and the protected area authorities. Households that viewed these actors as effective or were afraid of them were significantly less likely to clear land (Table 7.9). Household decision-making was also influenced by the perceived advantages of the land-use planning process, in terms of achieving recognition of their land holdings through land titling, and the perceived disadvantages, such as not being able to clear further land. Underlying social norms had a strong influence on behaviour; households that were more inclined to trust their neighbours and less inclined to be selfish were more likely not to clear land. Participants in both the Ibis Rice and Ecotourism programmes were much less likely to clear land, particularly in the case of the Ibis Rice programme, confirming previous results. The Bird Nests programme had no discernable effect on behaviour. Households that knew about the Ibis Rice programme were motivated not to clear land because they thought the programme was fair and offered significant benefits. No such motivations were detected for the Ecotourism programme, and there was some evidence that households that viewed the Bird Nests programme as providing positive benefits were less likely to clear land.

Table 7.7. Attitudes of local people towards the different external (Protected Area) and internal (Village land-use committee and payment programmes) in operation in their village recorded from the focus group discussions.

Village and Actors	**Influence**	**Alignment with Villages' interests**	**Liked/Disliked**
Antil, Kunapheap, Robunh and Sambour (inside Protected Areas, only Bird Nest payments)			
- Land use planning Committee	Weak (2), Moderate (2)	Low (3), Moderate (1)	Disliked (3), Neutral/Disliked (1)
- Protected Area Authorities	Moderate (3), Strong (1)	Low (4)	Disliked (4)
Narong and Prey Veng (inside Protected Areas, Bird Nest and Ibis Rice payments)			
- Land use planning Committee	Strong (1), Moderate (1)	Moderate (2)	Neutral (1), Disliked (1)
- Protected Area Authorities	Moderate (2)	Low (2)	Disliked (2)
- Ibis Rice	Moderate (2)	Moderate (2)	Neutral (2)
Dangphlat and Tmatboey (inside Protected Areas, Bird Nest, Ibis Rice and Ecotourism payments)			
- Land use planning Committee	Strong (2)	Moderate (2)	Neutral (1), Disliked (1)
- Protected Area Authorities	Moderate (2)	Moderate (1), Low (1)	Neutral (1), Disliked (1)
- Ibis Rice	Strong (2)	Moderate (2)	Liked (2)
- Ecotourism	Strong (1), Moderate (1)	Moderate (2)	Liked (1), Neutral (1)

Table 7.8. Cases of land clearance in the four study villages in 2008, the year land-use plans were approved and PES programmes became operational, and in 2009, after the land-use plans were approved and when payments were being made. The Village Committees in Tmatboey and Prey Veng villages were particularly active at following up on cases of illegal land clearance.

Responses to Land Clearance cases		Number of Land clearance cases	Permission[a]	No permission	Action taken by committee?[b]
Tmatboey	2008	28	7	21	24
	2009	12	0	12	12
Prey Veng	2008	9	5	4	9
	2009	3	0	3	3
Dangphlet	2008	31	0	31	3
	2009	7	0	7	1
Narong	2008	19	0	19	1
	2009	7	0	7	2
Total		116	12	94	55

[a] *Permission:* Households given official permission by the village authorities and committee to clear land,
[b] *Action taken by committee:* if the committee followed up and took action to limit the land clearance.

Table 7.9. Logistic regression model for the effect on household land clearance behaviour of household participation in PES programmes, social norms, and attitudes and perceptions of households towards key actors, land-use planning and the PES programmes. Models are provided for the entire dataset, and then for the subset of households that understood each of the PES programmes. Social norms were assessed using the General Social Survey questions (see text). Negative coefficents mean that the variable was associated with the not to clear land illegally. Village was included as a main effect in all models.

	Entire Dataset		Understand Ibis Rice		Understand Ecotourism		Understand Bird Nests	
	Estimate	*P*	Estimate	*P*	Estimate	*P*	Estimate	*P*
(intercept)	-1.164	ns	-0.181	ns	0.135	ns	-1.364	(*)
Household Size (square-root transformed)	0.657	(*)						
Social norm questions:								
- Trust?	-0.609	(*)						
- Helpful?	-1.033	*	-0.784	(*)			-0.834	*
Payment Programme participation:								
- Ibis Rice	-1.187	***						
- Ecotourism	-1.113	*						
- Bird Nests	(not significant)							
Attitudes towards effectiveness of key actors:								
- Village Land-use Committee	-0.615	**	-0.594	*	-0.907	***	-0.618	**
- Protected Area rangers	-0.337	*	-0.286	(*)				
Attitudes towards advantages or disadvantages of land-use planning:								
- Do not agree with land-use planning	0.371	**	0.286	*	0.378	*	0.334	*
- Want land titles	-0.223	(*)						
Attitudes towards PES programmes:								
- Fair?			-0.861	*	-0.430	ns	0.122	ns
- Provide net positive benefits?			-0.810	*	-0.221	ns	-0.621	(*)
Villages, difference from intercept (Dangphat)								
- Tmatboey	0.775	(*)	0.421	ns	0.118	ns	0.104	ns
- Narong	0.217	ns	0.019	ns			-0.185	ns
- Prey Veng	0.518	ns	0.158	ns			0.059	ns
Households (Villages)	272	(4)	178	(4)	108	(2)	208	(4)

7.4 Discussion

Did household payments change individual behaviour?

Rates of illegal forest land clearance fell significantly between 2008 and 2009-2010 in all the villages. This was probably because the village land-use plans were only agreed in 2008, so some households might have started to clear outside the boundaries before these were fully approved and demarcated, or might have initially not understood the process. Ibis Rice also became operational only from 2009 onwards, The reduction in clearance rate suggests that management of land-use by village committees can be an important tool to regulate illegal forest land clearance, at least over the short time frame of this study.

The effectiveness of household payments at changing individual behaviour was directly related to the strength of the conditionality attached to the payments, as others have hypothesised (Ferraro 2001; Ferraro and Kiss, 2002). The Ibis Rice household payments were directly conditional upon individual behaviour, and, as a consequence, had the greatest impact on that behaviour. This strict conditionality was well understood in the villages. The fact that land-clearing households were similar in socio-economic status to non land clearing households supports the conclusion that payments did have an impact, and that results were not due to household wealth or other factors. Household payments from the Ecotourism were only weakly conditional, and were seen as such, and had negligible impact on individual behaviour. Ecotourism also had a substantial ‘trickledown’ effect, with large number of households selling goods and services to the programme even if they were not directly engaged in it and had not agreed to the code of conduct. The evidence suggests that although these households benefited from the programme, it had no impact on their behaviour and they were possibly even more likely to illegally clear forest. These results support Kiss’ (2004) assertion that community-based ecotourism is an ineffective tool to achieve biodiversity conservation because there is a weak link between the incentives provided and conservation of biodiversity. Unsurprisingly, the Bird Nests programme, which rewarded households for protecting nesting birds rather than forest, had no discernible impact on land clearance behaviour. Previous work has shown that this programme was successful in meeting its aim of improving the conservation status of threatened bird species (see Chapter 4; Clements et al., 2012b).

Do fairness and equity matter?

The extent to which payments are socially appropriate, equitable, fair, or designed to build local support for conservation is often not an explicit consideration in the design of PES programmes (Jack et al., 2008; Pascual et al., 2010), even thought fairness is known to impact individuals' behaviour (Fehr & Falk, 2002). Brown and Corbera (2003) distinguish between three elements of equity in PES programmes: equity in access, equity in decision-making and equity in benefits. Table 7.10 compares the programmes across the three criteria, splitting the third criterion (equity in benefits) into participation and payment distribution. The Bird Nests programme was perceived as being very fair (Table 7.6), but only because anyone could access the programme, although in practice few households benefited and payments were quite inequitable. The Ibis Rice programme was also perceived as being very fair (Table 7.6), probably because it was easy for households to access the programme and participate in decision-making, even though the payments themselves were inequitable and benefited the wealthier households. By contrast, the Ecotourism programme was seen as the least fair (Table 7.6), despite benefiting as many households as the Ibis Rice programme, because the Ecotourism was perceived to be controlled by the village committee for their benefit. This comparison of the three PES programmes suggests that equity in access and equity in decision-making (i.e. ability to participate in the PES programme and the decision-making process) are more important in shaping attitudes towards than the equity in benefits (i.e. the distribution of payments). Differential payments can be seen as fair so long as the payment level is commensurate with effort (Konow, 2003), as was the case in both the Bird Nests and Ibis Rice programmes.

Table 7.10. Assessment of the three PES programmes against the three equity criteria of Brown and Corbera (2003) based upon the results of this study (see Figure 7.2 and Table 7.6).

Criterion	Bird Nests programme	Ibis Rice programme	Ecotourism programme
Equity in access	High; open to anyone	High; open to anyone who farmed	Low; benefits a few households linked to the village committee
Equity in decision-making	Low; managed by WCS	Moderate; managed by the village committee	Low; managed by the village committee but perceived to only benefit the committee
Equity in benefits (Participation)	Low: only 4% of households participate	Moderate: 19% of households participate	Moderate: 19% of households participate or receive benefits
Equity in benefits (Payments)	Moderate: payments are relatively inequitably distributed	Low: more productive households (= more wealthy) benefit more	Moderate: payments are relatively inequitably distributed

Sommerville (2010a) also found that a community-managed PES programme was perceived as fair even when there was some element of elite capture.

Although local people clearly had strong opinions on the fairness of the programmes, it is unclear the extent to which these perceptions influenced behaviour. Untangling the attitudes of local people towards the programmes from their decisions to participate in them and then their behaviours is complex given that these factors are not independent. Nevertheless, the behaviour models do suggest that the perceived fairness of the Ibis Rice programme was associated with individual decisions not to clear forest land, whereas the relative unfairness of the Ecotourism programme had no effect.

Why did households decide not to clear forest land?

This study has shown that a complex set of factors influenced the decisions by households in northern Cambodia to clear forest land. Of these, the household's socio-economic status was the least important, and there was no evidence that differences in wealth influenced decision-making. Households were definitely motivated by economic gain, particularly the desire to clear more land and to receive payments from PES. Perceptions of the benefits relative to the costs of land-titling and the payment programmes were important. Nevertheless, these factors alone were not sufficient to explain household decision-making. Other factors, such as a household's underlying attitudes, social norms, and the institutional framework significantly affected behaviour. Attitudes towards the fairness of the PES programmes were a factor. Underlying perceptions of the prevailing social norms, in particular the selfishness and trustworthiness of others, also had a strong influence on household behaviour.

Institutions are defined by North (1990) as: "the rules of the game in a society or, more formally, ... the humanly derived constraints that shape human interaction". Institutions in northern Cambodia influenced household decision-making in both positive and negative ways. External institutions included the protected areas; seeing protected area rangers as effective (suggesting that the household felt there was a risk of enforcement) was an important factor causing households to decide not to clear land. Desire for land titles (improved property rights) was a key positive motivation for households to abide by the land-use plans. Within the village, although the village land-use committees were often disliked, they were responsible for management of a land-use plan that had been agreed by the majority of households over a two year participatory process, suggesting that the rules had some local validity. The plans themselves had a significant impact on rates of forest land clearance, with incidents falling by fifty percent or more in all the villages after the plans were approved. Local rule enforcement has been shown to be a significant predictor of forest condition (Gibson et al., 2005; Chhatre & Agrawal, 2008; Persha et al., 2011), and locally-developed rules are far more likely to be respected and understood by local people (Berkes, 2003; Ostrom, 1990). The results presented here are consistent with this body of theory, suggesting that the village land-use committees were perceived by local people to be an effective complement to Protected Area authorities at regulating land-use. The two village-managed programmes (Ibis Rice and the Ecotourism) helped to empower the committees to fulfil these tasks, and, in the case of the Ecotourism in particular, provided the necessary incentive and funding to enable this to happen.

Discussion

> *"Our third innovation, a real achievement of Nagoya in 2010, was to put measurement at the centre of all our work…. As biodiversity has disappeared, we have been able to measure its decline more and more precisely. And in the last 2 decades we have started to learn the true value of measurement: that the more we measured, the more important our work was seen to be. The more accurate our measures—of biodiversity change, of conservation effort, of targets hit and missed—the more citizens, governments and corporations have understood and supported our work."*
>
> — fictional satirical quote from an imagined CBD conference in 2030 (Adams, 2010)

Conservation has been characterized as a crisis discipline that is running out of time (Chapin III et al., 2000; Pullin, 2002; Wilson, 2002). Amidst the crisis, the temptation for technocratic fixes is overwhelming (e.g. Adams, 2010). In conservation policy, this manifests itself as strong narratives arguing for particular blueprint approaches (Berkes, 2007; for an example see the Brooks et al.'s (2001) "blueprint for Africa"), which enjoy a brief period of popularity before being superseded by the next 'great idea'. The current enthusiasm for REDD+, for example, misses the fact that prior initiatives to "save" tropical forests have faded into history or irrelevance (e.g. Tropical Forestry Action Plans, FAO 1985; the United Nations Forum on Forests founded in 2000). Against the plethora of competing initiatives, each promising a new vision of the future, the loss of biodiversity and ecosystem services continues (MEA, 2005a; Hansen et al., 2008; Butchart et al., 2010; Laurence et al., 2012). The pessimistic conclusion is that global conservation initiatives have failed, are failing, and will continue to fail.

This failure – in both developed and developing countries – is a failure of human institutions to evolve to incentivize sustainable management. What might avert this decline? The first step is re-setting perspectives, accepting that human dominance of the earth's ecosystems is the norm (Vitousek et al., 1997), has been the norm for much longer than recent history (Adams, 2004), and that any solution must place humans at the centre and be grounded in a solid understanding of both human behaviour and human institutions. The opportunity is that conservation can succeed in the longer-term, but only if based on institutions that are socially and politically legitimate to local people and governments. This cannot be achieved by the export of western institutional models to developing countries (which may have failed in developed countries also; Acheson, 2006), but for local people and governments to decide appropriate institutions to manage their own natural resources, an evolving process that may not be based upon the externally-decided priorities.

Protected Areas and Payments for Environmental Services have both been proposed as institutional fixes to address the loss of biodiversity and ecosystem services, and both are flawed: Protected Areas because they assume it is socially and politically acceptable to restrict access to areas rich in biodiversity and natural resources; PES because it is based upon the simplistic assumption that conservationists can pay for what they want. The findings of this thesis support these two conclusions, whilst also demonstrating that both interventions can still deliver useful results on the ground.

8.1 Environmental and Social change in Cambodia

Cambodia is undergoing a period of rapid environmental change, on a scale not seen since landscapes were re-molded by the Angkorian Empire between 1100 and 700 years ago (Evans et al., 2007). Conflict followed by economic development has caused severe declines in wildlife populations since the 1970s due to over-hunting (Loucks et al., 2009), with many documented local extinctions (Duckworth & Hedges, 1998; Timmins & Ou Rattanak, 2001). Populations of tigers, the charismatic umbrella species that are highly valuable to hunters, have all but disappeared as peace is restored (Walston et al., 2010) and are predicted to soon become locally extirpated (Lynam, 2010). If over-hunting continues, the 'empty forests syndrome' (Redford, 1992) will become an increasingly likely reality (Corlett, 2007). At the same time, Cambodia now has one of the highest rates of land-use change globally (1.3% per year during 2000-2010; FAO, 2011), despite the fact that since 2002 most forest clearance has been illegal under the Forestry Law. Large-scale deforestation is primarily caused by economic land concessions for agri-industrial crops (Cambodia R-PP, 2011), mostly declared since 2005, which currently cover approximately 2 million hectares of the 18 million hectares of land area in the country (Vierze & Naren, 2012). Immigration to the forest frontier, driven partly by increasing landlessness as well as the opportunity for significant economic gain, is another significant driver of forest loss (McMahon, 2008; Chan, 2008; Biddulph, 2011). If these processes continue Cambodia will have lost most of its wildlife and over half of its forests in a period of only 20 years. In the context of such extreme rates of change, a critical question for environmental policy is how to invest conservation funding to achieve maximum impact.

In the face of rapid environmental change, Cambodians' security of access to and tenure of land and natural resources is extremely tenuous (Grimsditch & Henderson, 2009; Biddulph, 2011). The process of declaring economic land concessions barely recognizes the rights of existing residents (Grimsditch & Henderson, 2009) who may not benefit from their implementation (CEA, 2010). Approximately 21% of households are estimated to be landless and another 45% are land-poor, owning less than one hectare (Chan, 2008; Cambodia Human Development Report, 2011). Landlessness is caused by increasing population growth in the high density rice belt around the Tonle Sap lake (families having insufficient to land to share amongst their children) and evictions associated with designation of land for economic development (Grimsditch & Henderson, 2009). Land titling programmes, which might increase local land security, have been characterized as "geographies of evasion" because they prioritise areas of the country where land tenure is already secure (Biddulph, 2011).

To focus on these recent social changes is to ignore Cambodian history (Gottesman, 2004): the initiation of civil war in the early 1970s as a consequence of the on-going war in Vietnam; the Khmer Rouge-induced genocide that decimated and displaced rural populations as a consequence of collectivization (1975-1979); the installation of first a Vietnamese-installed communist government (1979-1989); then a United Nations-run transitional authority (1990-1993); and finally a western-induced model of democratic governance (1993-), all against the background of on-going conflict and civil war that lasted until the defection of the final Khmer Rouge brigades in 1998. For rural people in Cambodia tenure insecurity, oppression by external forces, and elite capture of power are the norm (Öjendal & Sedara, 2006), and the recent expropriation of forest land by elites for economic development is simply another manifestation of this. Going further into history, Southeast Asian peoples in forested upland areas have been subject to periodic oppression by different lowland dynasties, who have come and gone, and have typically responded through passive resistance (Adas, 1981; Scott, 2009). Sociologists and anthropologists emphasize a culture dominated by the exercise of power, social hierarchies, relational rigidity, patriarchal dominance, peasant docility, distance between the state and the people, a lack of general trust and social fragmentation (Öjendal & Sedara, 2006). At the same time, since the overthrow of the Khmer Rouge, Cambodia has been ruled by the same, relatively fixed, elite, who originated from within the Khmer Rouge and have survived to dominate all subsequent forms of government (Cock, 2010). The elite's success in positioning itself so as to benefit from the interplay of external and internal forces has been crucial to its ability to endure (Cock, 2010). These forces have included the competing factions during the two decades of civil war, itself a spillover of the same geo-political conflicts that caused the Vietnam War, donor-funded initiates since the early 1990s, and more recent foreign direct investment. At the same time the elite operate as the major extractor of resources from society through patronage relationships (Cock, 2010). Laws, which are developed by donor-funded projects and often written by foreign consultants, are supported sufficiently enthusiastically to attract donor investment, but then rarely adhered to or enforced (e.g. UN OHCHR, 2007; Cock, 2010). In this context, any attempt to establish new institutions for conservation, such as Protected Areas or Payments for Environmental Services, is fraught with difficulty.

8.2 Effectiveness of Protected Areas

Protected areas are viewed as the traditional cornerstone of conservation (Bruner et al., 2001), and despite the controversies regarding their implementation in many parts of the world (Adams & Hulme, 2001; Hutton et al., 2005; Brockington & Igoe, 2006; Adams & Hutton, 2007) they have proved remarkably durable as an institution. In 2010, the Convention on Biological Diversity at the 10th Conference of the Parties in Nagoya, Japan, adopted as one of the Aichi Targets for countries to have 17% of their terrestrial area under some form of protection by 2020 (CBD, 2010), an increase upon the current coverage of >12% of the terrestrial land surface (UNEP-WCMC, 2012). Large numbers of these protected areas have, however, little management ('paper parks'; Wilkie et al., 2001; Balmford et al., 2003; Chape et al., 2005; Joppa et al., 2008). Cambodia has approximately 4.5 million hectares of land under some form of protection (approximately 25% of the country), all declared within the last 20 years. The enthusiasm to create new protected areas has been driven by the desire of a new nation state for legitimacy and to attract donor investment, but with minimal capacity for implementation. The majority of these protected areas are not managed and all contain resident human populations.

Were Protected Areas effective at achieving conservation in northern Cambodia?

In northern Cambodia, this study has shown that managed Protected Areas were fairly effective at securing habitat, even with relatively small amounts of funding ($2-3/hectare/year). This contributes to an emerging body of evidence that protected areas are an effective policy instrument for tropical forest conservation (Albers & Ferraro 2006; Andam et al., 2008; Angelsen, 2010; Joppa & Pfaff, 2011; Ferraro et al., 2012; Geldmann et al., 2012), even for multiple-use protected areas that are similar to the Cambodia examples (Nelson & Chomitz, 2011). The extent to which Protected Areas in northern Cambodia were effective at achieving other conservation goals, particularly the protection of biodiversity, is unclear. Although wildlife populations were not monitored in the protected areas in this study, a long-term monitoring programme by the author and colleagues in another protected area, under similar management but with greater funding and equally low densities of wildlife, has failed to detect any significant increases since the start of active management in even highly fecund species (e.g. wild pig *Sus scrofa* and red muntjac *Munticaus muntjak*), suggesting that hunting rates remain high (O'Kelly et al., 2012). These findings are consistent with Geldmann et al. (2012) who found that protected areas were more likely to be effective at protecting forests than wildlife species. It is also consistent with concerns about the increasing number of 'empty' tropical forests (Corlett, 2007; Wilkie et al., 2011).

Measuring the impacts of protected areas on wildlife populations is complicated by the difficulty of selecting appropriate controls. Protected areas are much more likely to contain areas of importance for biodiversity conservation, although the global protected area network is far from representative (Brooks et al., 2004). Evidence suggests that densities of large wildlife species are greater inside protected areas and are declining slower (e.g. Caro & Scholte, 2007; Stoner et al., 2007), however the same is not necessarily true for other taxa (e.g. Gardner et al., 2007; Caro et al., 2009). At least one study has attempted to compare trends in wildlife populations inside protected areas with matched areas (Western et al., 2009), but frustratingly they do not report how the matching was conducted. In northern Cambodia, the bird nest monitoring found that large waterbird populations were significantly greater inside protected areas than around the matched control villages (see Chapter 4). Whether this effect was due to the protected area, or the Bird Nest payment programme, or underlying ecological differences between the controls and the protected areas is unclear. Even if appropriate control areas could be found, the high cost and technical expertise required for robust biological monitoring of species trends would make such an exercise beyond the means of most conservation projects (though by no means impossible; Nichols & Williams, 2006). An alternative and more cost-effective approach would be to assess hunting pressure, either through independent surveys for evidence of hunting (such as traps) or through interview methods such as the randomised response technique (St John et al., 2011) or choice experiments (Moro et al., 2012). The randomised response technique is unique in allowing respondents to disclose sensitive information because the interviewer cannot ascertain an individual's true response to the incriminating question (Solomon et al., 2007).

Protected Areas were effective at reducing external drivers of deforestation in comparison with the counterfactual case, principally immigration and large-scale economic development, at least for the period covered by this study. It is quite possible that migrants and large-scale developments were simply displaced to areas adjacent to protected areas, which experienced much higher rates of deforestation. Protected Areas were much less effective at reducing internal drivers of deforestation, such as intrinsic population growth and agricultural expansion by local residents. An interesting area for future research would be to understand why protected area enforcement in Cambodia is so weak. Previous studies have suggested that enforcement against wildlife hunting activities, logging or small-scale encroachment in Cambodia is complicated by the difficulty of detecting offences and the low penalties levied if detected (Claridge et al., 2005). Both of these factors are critical if the law is to act as a sufficient deterrent (Becker 1968; Keane et al., 2008). In the study area, detection probabilities are low due to the relatively small number of protected area rangers (no more than 50 per protected area), spread over a large area (c.1,500km^2) that contains several existing villages with thousands of local residents. Under Cambodian law, the potential penalties for offences are high (up to 10 years in jail), but such penalties are rarely levied by the courts (Claridge et al., 2005). In practice, this means that the options open to protected area authorities are to issue a written warning (which carries no financial penalty), seize any illegal goods and equipment (e.g. chainsaws) or to impose a direct fine (Claridge et al., 2005). However, these problems do not explain why even amongst protected area staff there is considerable reluctance to impose harsh fines, suggesting that such penalties are not seen as socially or politically legitimate (as, indeed, they are not by the courts). The perceived legitimacy of rules is known to affect their acceptance (Hønneland, 1999; Sutinen & Kuperan, 1999).

Protected Area enforcement could be improved by taking steps to increase the detectability of offences, for example using law enforcement monitoring tools (Jachmann, 2008), or through greater investment in patrolling (Hilborn et al*.*, 2006). Previous studies in Cambodia have suggested that protected area authorities should work with the courts to improve the likelihood of successful prosecutions (Claridge et al., 2005). Addressing drivers, for example by closing down wildlife markets and penalizing wildlife traders and sellers, may be more effective than targeting individual hunters or loggers directly (Damania et al*.,* 2005). An alternative, and more innovative, approach would be to attempt to improve the social and political legitimacy of the protected areas, which are currently viewed as an externally-imposed construct. This would require local decision-making and management of rules and regulations to be devolved to local village authorities, who then might take action to promote whatever rules are agreed, increasing social sanctions for compliance (such as peer pressure; Kaplow, 1990). Models suggest that supporting monitoring by village committees does enhance compliance (Mesterton-Gibbons & Milner-Gulland, 1998), and local rule enforcement is known to be effective at conserving forests (Gibson et al., 2005; Chhatre & Agrawal, 2008; Persha et al., 2011). Local sanctioning is already occurring in the villages with functioning village-managed PES programmes: there, local people were more concerned about the village committees (who cannot levy any financial penalties) than the protected area rangers (see Chapter 7).

Based on this evidence, Cambodian protected areas are best viewed as a land-use designation preventing the larger-scale drivers of deforestation (at least during the period of this study), but relatively impotent against small-scale activities (such as hunting by local people).

Did Protected Areas exacerbate local poverty?

By limiting the impact of external drivers of deforestation, the protected areas helped to secure natural resources for local residents, even in the absence of a formal forest tenure reform programme (see Chapters 5 and 6). Local uses of the natural resources within protected areas were permitted under the law. Arguably therefore, in the context of general land insecurity in Cambodia (Grimsditch & Henderson, 2009), being within a protected area (as a more powerful institution than village authorities) afforded local people with greater security of access natural resources than villages outside. At the same time, the protected area was relatively ineffective at detecting and enforcing against the activities of local residents.

Consequently, the protected areas had minimal impacts on the wellbeing of local people over the timeframe of this study, based upon the three main measures of wellbeing used (poverty, agricultural productivity and food security). This finding is consistent with the results of the three other studies of the social impacts of protected areas that have been completed to date – in Costa Rica (Andam et al., 2010), Thailand (Sims, 2010) and Uganda (Naughton-Treves et al., 2011). However, all four studies used very different methodologies. The Costa Rica and Thailand analyses used only low-resolution national census data, and is so doing may well have missed important local heterogeneities. For example, local villages were relocated from the Thai protected areas (Roth, 2004; Sims, 2010), but the scale of the analysis was insufficient to investigate impacts for these people. Similarly, evaluations of the environmental impacts of Costa Rica's PES programme have conclusively shown that national-scale evaluations may miss important local impacts (Arriagada et al., 2012). Naughton-Treves et al. (2011) compared people living within 1 km of the boundary of Kibale National Park with those living >4 km from the park boundary. This is a very different comparison from that undertaken in this study, where local people living inside (not adjacent to) the protected area were compared with matched controls >60 km away. As this study has shown, the characteristics of villages within, adjacent to and further away from a protected area may be qualitatively different, and it is unclear the extent to which Naughton-Treves et al. compared like with like (since villages were not matched). There is a strong need to greatly expand the evidence base, by analysing the impacts of protected areas on local people from a large number of countries and in different contexts over much longer time-scales, before drawing generalisations from four initial studies conducted at very different scales in four very different social contexts.

In the Cambodian villages all study groups (those living within the protected areas, the controls and the border villages) were experiencing significant improvements in living standards, which could be detected even over the short timeframe of this study. Such results can be explained by Cambodia's rapid rate of development: national economic growth averaged nearly 10% during 1998-2008, declined to 0.1% in 2009 during the global financial crisis, and recovered to approximately 6-7% during 2010-2012 (World Bank, 2012). This finding supports the hypothesis that macro-level factors are just as important, if not more important, than micro-level factors at determining changes in local wellbeing. As remote villages become increasingly linked into national, regional and global markets this trend will only continue. A useful area for future research would be to compare the local impacts of conservation policies with the macro-level impacts of national (or regional) economic trends, be they positive or negative (e.g. consequences of the Global Financial Crisis) and to assess the extent to which local level interventions do actually influence livelihoods in different settings.

Disaggregating impacts, protected areas did have important effects on the livelihoods of local people. By restricting some forms of agriculture, particularly shifting cultivation and cash crops (which were relatively easy for protected area staff to detect) the protected areas imposed costs on some households. Poor families are more likely to conduct shifting cultivation if they don't have access to suitable paddy land or cannot afford the necessary draft animals or machinery to farm paddyfields (McKenny et al., 2004). Cash crops are one of the most important pathways out of poverty in the rural uplands of Cambodia. Protected areas did benefit the significant proportion of households that practiced resin-tapping (>55% of households) or had access to only small amounts of land (15% of households). Through safeguarding traditional rights to harvest and use forest resources the protected areas ensured that local poverty was not exacerbated. By contrast, villages outside protected areas were far more likely to lose access to land and particularly forest resources as land was expropriated by concessions and resin-trees were logged (Grimsditch & Henderson, 2009).

The protected areas are therefore constraining the livelihood strategies of local people, in particular the ability to 'step-out' of subsistence rice farming and non-timber forest product collection livelihood strategies by investing or switching into new activities and assets (Dorward et al., 2009). Protected area residents were therefore constrained to a choice between 'hanging-in', which aims to maintain and protect current livelihood activities, or 'stepping-up', which involves investments to expand the scale or productivity of existing assets and activities (Dorward et al., 2009). Whilst safeguarding current livelihoods is probably a viable argument for protected areas in the medium term, especially given the wider tenure insecurity in Cambodia, it is unlikely to be socially acceptable in the longer term. A study of 61 cases of livelihood strategies based on non-timber forest products in Asia, Africa and Latin America found that these strategies did not reduce poverty in most cases (Belcher et al., 2005). The same factors that tend to make non-timber forest products important in the livelihoods of the poor, also limit the scope of non-timber forest products to lift people out of poverty (Sunderlin et al., 2005). It was precisely for this reason that WCS instituted a programme of Payments for Environmental Services in northern Cambodia, in order to provide economic opportunities to local people that were directly linked to conservation – i.e. the ability to 'step-out'.

Panel studies, such as this one, have great power to detect changes in livelihood strategies and the social impacts of interventions, if continued for a long period of time. Only one other panel study of protected area impacts is known to be ongoing (see Wilkie et al., 2006; Foerster et al., 2011), although another study has used longitudinal data over 10 years from the same villages (but not the same households; Naughton-Treves et al., 2011). Other published studies have collected data from intervention and control households from only a single period in time (e.g. for marine protected areas, Leisher et al., 2007). The problem with one-off samples is that they fail to capture livelihood dynamics. For example, if the current study had only assessed impacts in 2008 it might have concluded that protected areas were beneficial for local people (in 2008 households inside protected areas were better off than controls; Chapter 3), whereas the rate of change between 2008 and 2011 suggested that all households were changing in poverty at a similar rate (see Chapter 6). Only by continuing the current study for a much longer period of time will it be possible to determine the impacts of protected areas on different development pathways and the extent to which protected areas may limit local residents to a forest-dependent lifestyle.

The same survey design also has the potential to answer another question that is of huge policy relevance in Cambodia currently: whether or not economic land concessions exacerbate or help to alleviate local poverty (CEA, 2010). Within the past year, rubber plantations have been announced on land that is currently occupied by some of the control villages. Future re-assessments would reveal the extent to which these people benefited or lost out as a consequence of the concessions. The gradual divergence in the development pathways of the control villages from those within protected areas calls into question what is an appropriate comparison to make in impact evaluation studies: ultimately every village is different, and at some stage these village-level differences in development opportunities may be more important than landscape-level factors (such as being within a protected area).

8.3 Design of Payments for Environmental Services in the context of weak institutions[9]

Wunder (2007) suggested that effective implementation of PES may be considerably more difficult where institutions are weak. The majority of PES programmes have been implemented in countries where the institutional framework and property rights are strong and target the behaviours of private landowners (e.g. USA, Europe, Costa Rica, etc). By contrast, this thesis has compared three PES programmes in a forest landscape in Cambodia, where land and resource rights are poorly defined, governance is poor, species populations are low and threats are high. The programmes varied in the extent to which payments were made directly to individuals or to villages, the degree of management by the village committees, and whether payments were strictly or weakly conditional upon performance. Figure 8.1 maps the three programmes onto the spectrum of conservation incentive approaches described by Ferraro and Kiss (2002). In this final section the programmes are evaluated against four criteria: the institutional arrangements, the operational costs, the social impacts and the conservation results observed (Table 8.1), and conclusions are drawn regarding the design of PES programmes in developing countries.

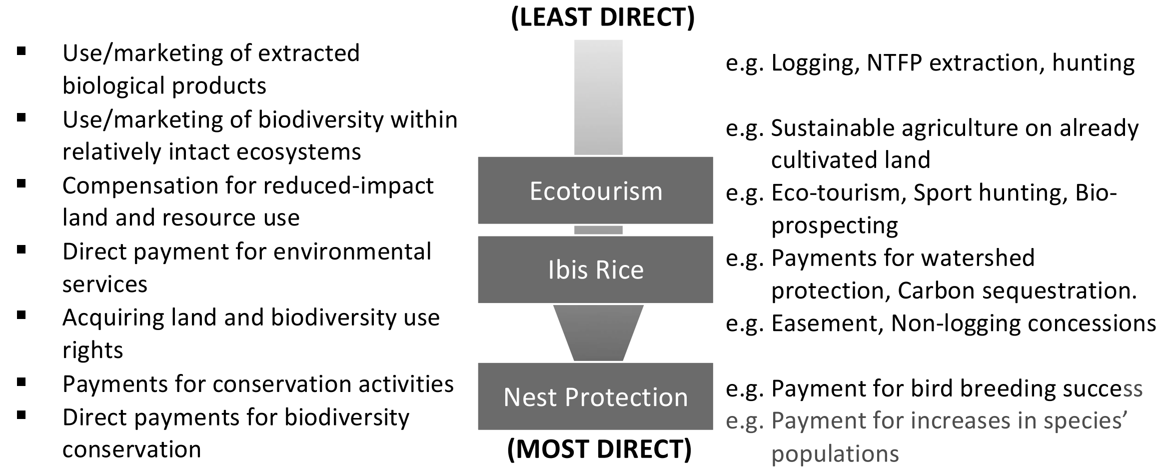


Figure 8.1. Spectrum of investments for biodiversity conservation (from Ferraro and Kiss 2002).

[9] This section and Table 8.1 is adapted from: Clements et al. (2010) *Ecological Economics*, *69*, 1283-1291.

Table 8.1. Summary comparison of the three PES programmes and the Protected Area intervention

	Community-based Ecotourism	Ibis Rice	Bird Nest Protection	Protected Area
Description				
– Conservation Objectives	Wildlife: Yes Habitat: Some	Wildlife: Some Habitat: Yes	Wildlife: Yes Habitat: No	Wildlife: Yes Habitat: Yes
– Targets behaviour of	Individuals & Village	Individuals	Individuals	Individuals, Villages and External actors
– Community Income	$1000-$3500/village/year	>$300/village/year	None	None
– Individual Income				
• Total/village	$4000-$5000/village/year	$4000-$14000/village/year	<$1000/village/year	Limited (tourism and employment, c.25-50 people)
• % participation	9% of households employed,	>17% of households,	4% of households,	
• Average payment	$220/year average payment	$320/year average payment	$120/year	
• Extra payments	Another 10% of households receive $35/year	Potentially all farmers could benefit	1 local ranger/village to monitor	
1) Institutionality				
– Organisational arrangements	Three actors: Village: Management External agency: Certification & Marketing Private Sector: Buyer	Four actors: Farmers: Keep to conservation agreements Village: Management External agency: Certification & Marketing Private Sector: Buyer	Two actors: Individuals: Protection WCS: Monitoring and making payments	One actor: the Protected Area
– Property Rights	Forest: common property co-managed by Village and the PA	Forest: common property co-managed by Village and PA; individually owned fields	Nests: *de facto* individual control	State Public Land
– Contracts	Tourists →Village Committee	Buyers →Village Committee →Villagers	NGO →Villagers	None
– Local Governance	Yes (Local Management)	Yes (Local Management)	No (NGO Management)	None
– Monitoring	External agency (certification)	External agency (certification)	WCS	Protected Area

2) Operational Costs				
– Initial investment	High ($50,000/village)	High ($50,000/village)	Low	High
– % of programme costs paid to locals	24%	55-60%	71-78%	<5% (local salaries and supplies)
– Cost-Efficiency				
• Forest protection	$25-50/hectare	$25-70/hectare	None	$2-3/hectare
• Species protection	$500/nest	$500/nest	$60-$120/nest	
– Financial Sustainability?	Yes (both for business & certification and marketing costs)	Yes (both for business & certification and marketing costs)	No (WCS pays $30,000/year)	No (Minimum of $300,000/year)
3) Social impacts				
– Participation	Limited number of households	Potentially all households	Limited by number of nests	Limited employment
– Equity				
• In access	No	Yes	Yes	No
• In decision-making	No (elite capture)	Some	No	No
• In participation	Some	Yes	No	No
• In payments	Some	No	Some	n/a
– Perceived as fair	No	Yes	Yes	No
– Wellbeing				
• Poverty	Positive impacts	Positive impacts	No impact	No impact
• Agriculture	No impact	Positive impact	No impact	No impact
– Collective Action	Yes	Maybe	No	No
4) Conservation results				
– Address drivers of deforestation				
• Internal	Yes (through village committee)	Yes (changes farmer behaviour)	No	Yes (enforcement)
• External	Some (rejection of immigrants)	No	No	Yes (designation)
– Conservation of				
• Key Wildlife	Some	Unclear	Significant	Unclear
• Habitat	Some	Significant	None	Significant

(1) Institutional Arrangements

The institutional framework includes property rights, monitoring, enforcement, governance and contracting arrangements (Table 8.1). Of the three PES programmes described, the Bird Nests programme has the simplest institutional arrangements, since it relies on a direct contract between the individual and WCS to protect biodiversity. It is assumed that individuals can temporarily control a breeding site even if they do not own it. Simple contracting can fail however if not adequately supported by the institutional framework. For example, the Monarch Butterfly project in Mexico purchased logging rights from forest-dwellers to protect butterfly habitat; however most illegal logging was performed by powerful outsiders, which local people were incapable of preventing (Missrie & Nelson, 2005). Similarly, Cambodian bird nest protectors were unable to stop others from clearing breeding sites.

Both the Ecotourism and Ibis Rice programmes have more complex institutional arrangements. The Ecotourism contract is made directly with a village organisation, which has been approved by the Government to develop local land-use regulations, whilst Ibis Rice is a hybrid programme; the village organisation then sub-contracts to individual farmers. The village institutions – the local rules governing natural resource management – are nested in a multi-layered framework that includes:

- An external agency that provides rewards by connecting the villages to national and international markets, certifies compliance, and helps to mediate conflicts.
- External organisations, including private sector companies and NGOs, that reinforce rules and can assist with resolving conflicts or other problems.
- The Protected Areas, who were particularly effective at preventing external drivers of forest loss, thereby securing the forest resources for local management by the village organisation. It is unlikely that the village authorities alone would have been powerful enough to achieve this.

In summary, institutional arrangements under the most direct contracts programme are considerably simpler than the other two examples, but this is not necessarily an advantage. The more complex institutional arrangements are multi-layered, with redundancy and reinforcement provided by different organisations (for example external oversight by WCS, the Protected Area Authorities and marketing or tourism agencies). These arrangements build resilience and checks in the system that ultimately may make the programmes more effective and sustainable (Berkes, 2007).

(2) Operational Costs and Cost-Efficiency

The trade-off between direct payment and more complex payment models continues when considering the costs and the cost-efficiency of the programmes. The simplified institutional arrangements of the Bird Nest programme meant that it was inexpensive to establish and disbursed a higher percentage of its cost at the local level (consistent with the predictions of Ferraro & Kiss, 2002). It was also remarkably cost-efficient, protecting a very large number of birds for only a modest annual payment (Table 8.1). By contrast, the more complex Ecotourism and Ibis Rice programmes were much less efficient at disbursing revenue locally, mainly due to marketing and monitoring costs incurred by the external agencies, and were expensive to establish, requiring substantial investments over approximately two years to build the capacity of the village organisations. They were also far less cost-efficient than Protected Area management in terms of the cost per hectare of forest protected (Table 8.1). However, the Bird Nest programme was also entirely dependent upon the funding raised by WCS, whereas both the Ecotourism and Ibis Rice programmes, once established, have the potential to be sustained through market sales. This highlights the trade-off between direct payment and more indirect approaches: locally managed market-linked programmes are expensive to establish and maintain, but ultimately will be sustainably financed, whereas direct payment approaches, however cost efficient, are dependent upon external support. Direct payment programmes for biodiversity conservation have been criticised previously for this very reason (Swart, 2003).

(3) Social Impacts

Payments for Environmental Services have been advocated for as a potential 'win-win' policy that will protect the environment and help to alleviate local poverty (Ferraro & Kiss 2002; Pagiola et al., 2005). The results of the current study support this hypothesis (see Chapter 6). The development impact of the different PES programmes was directly related to the magnitude of the payments provided. The lowest paying programme (the Bird Nests), which compensated people based upon the average cost of a day's labour, had no discernible impact on household wellbeing. By contrast the two higher-paying programmes (Ecotourism and Ibis Rice) caused significant improvements in participants' poverty status, agriculture and food security, and allowed them to pay for high school education for their children. Both the Ecotourism and Ibis Rice were explicitly designed by WCS to provide development benefits as well as supporting conservation efforts, which is why the payments were higher, whereas the Bird Nests programme purely targeted species protection. This contributed to the greater cost of the Ecotourism and Ibis Rice programmes. These results are consistent with Wunder et al.'s (2008) hypothesis that payment programmes which are designed to achieve two outcomes (conservation and poverty reduction) will be economically more inefficient than programmes that are designed to only deliver conservation.

Wunder (2008) hypothesised that the poverty impacts of PES would depend upon the extent to which a PES programme had pro-poor or anti-poor selection biases. The Bird Nests programme did not necessarily require a change of livelihood strategy because households could chose to send an older son to protect nests (for example), and since some payments were provided up front there were limited barriers to entry into the programme. By contrast, the Ecotourism and Ibis Rice offered households an opportunity to change their livelihoods by diversifying (into ecotourism; 'stepping-out'; Dorward et al., 2009) or intensifying (agriculture production; 'stepping-up'; Dorward et al., 2009). Better-off households were more inclined to take this new opportunity, which might explain why participation in the two programmes was non-independent; this could be investigated further. There was also some element of elite capture, particularly for the Ecotourism where only a limited number of jobs were available. Kiss (2004) has criticized community-based ecotourism as a development tool due to potential for the benefits to be captured by a small portion of the community. Although Ibis Rice was also managed by the village committee, elite capture was of less overall significance because unlike with the Ecotourism there was no limit on the number of households that could participate. Ibis Rice therefore has the greatest potential to benefit the poor.

All the programmes were perceived by local people as inequitable to some extent, particularly the Ecotourism due to the high proportion of benefits captured by village elites. To local people, the extent to which they could access a programme and, to some extent, be involved in decision-making, were more important than whether the benefits were distributed fairly. Being fair made a programme more popular, and there was some evidence that it had an effect on household behaviour. However, there is a high degree of endogeneity between the attitudes, participation and behaviour of households, which makes drawing inferences from the current analysis difficult. A more comprehensive set of social surveys based upon the theory of planned behaviour might separate out these effects and provide a clearer indication of the importance fairness plays in determining household responses to payments (for an example of such an analysis see Williams et al., 2012). The evidence from the Bird Nests programme suggests that fairness is important, and even if a small subset of people feel that a programme is unfair this can lead to unintended consequences. Replication of the Bird Nests programme by other organisations in Cambodia (WWF and BirdLife International) has found similar results.

(4) Conservation Results

Very little is known about the effectiveness of PES as a conservation intervention in developing countries and no prior impact evaluations of user-financed PES have been undertaken (Ferraro et al., 2012). Theory suggests that the effectiveness of the PES programmes would be directly related to the conditionality attached to the payments (Ferraro & Kiss, 2002). The results of this study confirm the hypothesis that conditionality does matter. The two PES programmes that linked payments directly to behaviour –Bird Nests and Ibis Rice – were the most effective at changing individual behaviour and producing additional environmental outcomes (species protection and reductions in forest clearance respectively). This provides the first evidence that user-financed PES can be effective as a conservation intervention. The Ecotourism programme, which was only weakly conditional, had much less impact on the behaviour of individual participants. These results demonstrate the importance of ensuring payments are strictly conditional when designing PES programmes.

There are trade-offs in PES design associated with paying for actions versus paying for conservation results (e.g. paying for bird nest protection rather than paying for the number of birds around a village; Gibbons et al., 2011; Hanley et al., 2012). Paying for actions certainly has considerable advantages, because the monitoring of compliance is made considerably easier and cheaper (Gibbons et al., 2011; Hanley et al., 2012), as in the Bird Nests and Ibis Rice example. Payments for results are also riskier for the participant than payments for actions, because conservation outcomes (e.g. bird population sizes) are determined by a range of factors, many of which they cannot control (Whitten et al., 2007). However, the implicit assumption when paying for actions is that the underlying dynamics of the system are well understood, including the threats to biodiversity and habitats, the likely response of local people to the conservation payments, and the resulting impacts on the conservation targets the programme was designed to achieve (Tomich et al., 2004). Targeting a single action using a strictly conditional payment programme is risky when there are multiple changing threats to species' populations, and it is unclear whether the payment programme targets the most important threat. Large birds in Cambodia are threatened by poisoning (Clements et al., 2012a) and habitat loss in addition to nest collection, but the Bird Nests programme only addresses the last of these threats. Ibis Rice might be effective at reducing forest land clearing behaviours, but may not incentivise local people to prevent poisoning or collection of bird nests. These examples demonstrate the importance of ensuring payments-for-actions PES programmes correctly target the principal conservation threats, or of having some built-in redundancy, for example by operating multiple payment programmes in the same villages each targeting different behaviours (which is what WCS elected to do in northern Cambodia). Having multiple payment programmes operating in the same place may introduce other economic inefficiencies.

The alternative approach would be to pay for a conservation outcome (e.g. the number of birds around a village), and let the local people decide for themselves how to achieve this goal (Gibbons et al., 2011; Hanley et al., 2012). This may be a more effective approach where there are information asymmetries between the local people and the PES programme administrator, where local people better understand the local context and the actions necessary to conserve biodiversity (Gibbons et al., 2011; Hanley et al., 2012). For example, local villagers in Cambodia probably better understand the occurrences of different threats to bird populations than WCS (e.g. who is poisoning birds and where, which villagers hunt birds, etc.), and, if appropriately compensated, might deliver the same conservation outcome more efficiently than paying for actions. To some extent this is what the Ecotourism programme aimed to achieve, although as discussed the payments were only weakly conditional on performance. There is evidence that the village committee did undertake measures based upon their own local knowledge to protect bird populations. These included paying local villagers from the tourism funds to protect the nests of birds tourists wanted to see (so the Bird Nests programme only needed to operate for a reduced set of species), promoting local conservation rules and regulations, and setting aside local wetlands for tourists. Wildlife populations were not monitored at a sufficiently detailed scale in this study to assess whether or not the Ecotourism programme was effective at protecting bird species, but anecdotal evidence from tour group leaders (who visit the villages every year) suggests that species are recovering. At Tmatboey, populations of one species, the White-shouldered Ibis, increased from a single breeding pair in 2004 to flocks of 34 individuals by 2010 (Clements et al., 2010; Wright et al., 2012). In the future, the Ecotourism programme could perhaps increase the conditionality associated with the payments, for example by linking them to local censuses of bird populations or protection of particular forest areas.

Although there is only limited evidence that the Ecotourism programme changed participants' behaviour, it did provide funding to empower the village committees to enforce local conservation rules and regulations. In so doing, the Ecotourism programme helped to stimulate collective action. Villagers' perceptions of enforcement by the village committee were a strong factor determining their behaviour, suggesting that the village committees were relatively effective. In Madagascar, a different community-based PES programme that was also managed by a local elite similarly had a strong influence on the behavioural intention of individuals (Sommerville et al., 2010a; Sommerville et al., 2010b). This raises the important question of whether PES programmes should provide conditional payments to individuals or communities (however defined), and the balance between these two scales of payments. This is a critical issue currently in REDD+ policy (Peskett, 2011), and evidence from the three Cambodian examples suggests that structuring payments at both individual and communal levels may be more effective, depending upon the context. At least one existing REDD+ demonstration programme (the Bolsa Floresta in the Brazilian Amazon; Viana, 2010) has chosen to structure payments in this way.

Regardless of the way the PES programmes were designed, none of them were able to influence the external drivers, particularly of land-use change, which are so important for the overall integrity of the system. This returns to Wunder's (2007) original concerns about PES in weak institutional contexts. In the Cambodia case, the protected areas provided the security of forest land tenure and a basic rule of law within which implementation of PES was possible (for similar results see Wunder & Albán, 2008). Without this effect the PES programmes would probably collapse. Although the academic discussion of PES and other instruments is often framed in terms of 'either-or', the more policy-relevant question concerns how different instruments should be combined to achieve conservation objectives (Engel et al., 2008). Environmental economic theory tells us that, in a second-best world where several sources of market failure coexist, a combination of instruments is needed, one for each objective (Tinbergen, 1952). As Landell-Mills and Porras (2002) have written: "the key question is, thus, not whether we should promote markets instead of government intervention, but what is the optimal combination of market, hierarchical and cooperative systems for governing forest sector utilization and management?"

Conclusion

Institutional frameworks in tropical forest countries, many of which are undergoing a rapid rate of forest loss or erosion of biodiversity, are often weak and uncertain (Barrett, et al., 2001). Designing PES programmes in the context of weak institutions is challenging, particularly if property rights are not clearly defined. This comparison of three programmes from Cambodia has highlighted two different approaches. The first is direct payments to individuals who can temporarily control a biodiversity resource, modelled on the approach proposed by Ferraro and Kiss (Ferraro, 2001; Ferraro & Kiss, 2002). The second approach is longer-term and requires investing in clarifying property rights and building local institutions for management of wildlife and habitats in addition to provision of incentives (Cranford & Mourato, 2011). The comparison suggests that the first approach can be very effective initially: the Bird Nest programme rapidly protected several hundred pairs of globally threatened bird species, was inexpensive to implement and had low administrative costs with most money disbursed locally. However, this comparison has also suggested two significant problems with the approach.

Firstly, PES requires strong institutional frameworks, particularly enforcement of property rights (Wunder, 2007; Engel et al., 2008; Muradian, et al., 2010). The Cambodian bird nest protectors had weak ownership rights over breeding sites, and were unable to protect them in the longer term from clearance by others. In the absence of strong existing institutional frameworks, payment programmes need to invest in building appropriate institutions both at the village and higher levels. Increasing the diversity of institutions creates checks, improves resilience and sustainability in the system (Berkes, 2007) but imposes its own costs. In the Cambodian cases, both the village-level organisations and the protected areas were needed in order to create a robust institutional framework to facilitate the PES programmes and to mitigate external drivers of biodiversity and ecosystem loss.

Secondly, payments to some individuals, but not to others, may fail to generate support for conservation, which is very necessary when the institutional framework is weak. Unlike the bird nest example, the evidence suggests that the two Cambodian village-managed programmes started to build local support for and understanding of rules and regulations for protected species and land-use plans. These rules and regulations were developed locally and approved by the entire village. Empowerment is an important step in this process, defined by Chambers (1983) as “the process through which people, and especially poorer people, are enabled to take more control over their own lives, and secure a better livelihood, with ownership of productive assets as one key element”. The importance of intrinsic motivation at determining behaviour has been recognised by psychologists since the 1980s (Deci & Ryan, 1985; DeCaro & Stokes, 2008). Endogenous rules are far more likely to be respected and understood by local people (Ostrom, 1990; Berkes, 2003), in comparison with externally-imposed rules (Cardenas et al., 2000). By contrast, bird nests are valued only because WCS chooses to pay for their protection, not through any particular recognition of the birds' importance, and if payments by WCS stopped, even temporarily, collection of bird nests would probably resume. Payment programmes that are structured to facilitate intrinsic motivations are therefore far more likely to be successful.

PES programmes are best viewed as a tool in a broader process of strengthening institutions for conservation of biodiversity (Agrawal & Gibson, 1999; Barrett, et al., 2001; Pagiola & Platais, 2007). The conditions under which institutions for collective management of common pool resources are likely to be formed have been well articulated through several decades of research (Ostrom, 1990; Agrawal, 2001; National Research Council, 2002). However, few settings in the world are characterised by all these conditions (Dietz et al., 2003). The challenge therefore is to devise institutional arrangements that help to establish such conditions or meet the main challenges of governance in the absence of ideal conditions (Dietz et al., 2003). PES programmes can address two critical constraints, firstly by providing an incentive to reform institutional arrangements (for example clarification of property rights), and secondly by increasing the financial returns from collective management through provision of additional payments under conditions where sustainable extraction alone would not be profitable. At the village level, the combination of a stronger institutional framework and payments leads to a greater local incentive for collective action, i.e. the village moves closer towards fulfilling the design principles articulated by Ostrom and others (Agrawal, 2001). In the Cambodian cases the payments are critical for three main reasons. Firstly, they increase the value of the biodiversity resource to local people, both directly through individual payments and indirectly by providing funds for village development. Secondly, the payments fund the costs of management of common pool resources by village institutions, a system which is itself a public good (Ostrom's 'second-order social dilemma' (1990)). Thirdly, the payments fund monitoring and sanctioning by the village institutions (Ostrom's 'third-order social dilemma'). The structure of the payments – providing revenue at both the individual and village-level scale – ensures that these outcomes are possible.

8.4 Future Directions

How robust are these results?

This thesis is the first study to have used evaluation methods to evaluate the social impacts of PES, and only the fourth such study completed for protected areas. The overall findings, that protected areas had limited impacts on the livelihoods of local people whilst PES has the potential to reduce poverty, are potentially of high significance. There are, however, a number of concerns with the methodological approach.

Use of matching methods, the current method of choice in all impact evaluation studies of conservation programmes (see Ferraro et al., 2012), is problematic. Matching relies upon the researcher having a strong qualitative understanding of the system that is being studied in order to select appropriate matching variables (Ravallion, 2006). Inappropriate choice of matching variables leads to unobserved sources of bias (Woolridge, 2002; Ravallion, 2006) and means that the results are unlikely to be internally valid (Drury et al., 2011). Since the majority of completed impact evaluations are desk-based studies based upon low-resolution datasets (such as remote-sensing data) with limited fieldwork, a strong concern is that matching variables are selected based upon expediency rather than a detailed qualitative understanding of the system being studied. The selection of the matching variables is probably the most important step of any study, and yet the underlying dynamics of the system and how the choice of matching variables affects the results is rarely discussed (for a useful exception see Andam et al., 2008). The development economics literature (which is probably a decade or more years ahead of the conservation science literature on these issues) already contains papers comparing estimators from matching methods with randomized controlled trials, with varying results (for a review see Duflo et al., 2007). Arriagada et al. (2012) re-analysed the environmental impacts of the Costa Rican PES programme at a finer scale and obtained different conclusions from previous studies. It is likely that future research in conservation science will demonstrate significant variation between different evaluation methodologies.

Critical to the logic of PES is that payments change the behaviour of people who otherwise would have contributed to the loss of biodiversity or degradation of ecosystem services. Whilst this thesis has presented strong evidence that PES did deliver additional environmental outcomes (over the counterfactual), the evidence that PES led a sustained change in behaviour is weaker. It is very possible that local people saw an opportunity for a quick profit, and will return to hunting wildlife or clearing forests in the future. Boosting market prices for produce, which is effectively what Ibis Rice does, is known to encourage forest clearance (Chomitz et al., 2007; Angelsen, 2010). Individual Cambodian villages have experienced a plethora of development interventions over the past 10-15 years, mainly led by non-government organisations (Marschke, 2005). Local people have become adept making the best out of each new initiative, equivalent to Adas' (1981) 'passive avoidance'.

Only by continuing to monitor the implementation of the PES programmes and the behaviour of participants will the extent to which payments lead to genuine changes in behaviour become clear.

Success from whose perspective?

Implicit in the design of any social impact evaluation is the consideration of the audience the products of the evaluation are to inform. This study focused on objective, quantifiable, measures of human wellbeing from a large-*n* study that would be understandable to policy makers, funders, and technicians, and are therefore externally valid. Using the latest scientific impact evaluation methods ensures that the results are seen as having the greatest level of accuracy possible.

Scientists distinguish between external and internal validity (Drury et al., 2011). External validity evaluates the contexts in which results can be applied, or the extent to which results are representative of a population and are therefore generalizable. Large-*n* statistical studies that can be generalized are pre-disposed towards external validity. Scientific validity, however, also requires that the data must be representative of the particular phenomena the researcher is trying to investigate, thus demonstrating internal validity. Internal validity refers to whether, through carefully defining concepts, constructing measures and conducting research, a study actually investigates what it claims to (Drury et al., 2011).

Internal validity is essential for any scientific question, but it is particularly problematic for the assessment of social impacts. Policymakers may be interested in the question "has this programme exacerbated or reduced local poverty?". However, for local people the relevant question may be "do I think this programme has had a negative or a positive impact on my wellbeing?". This is conceptually a very different question, and it requires very different data to answer it. To a villager in a protected area in northern Cambodia the appropriate comparator against which to assess changes in their wellbeing is almost certainly not a village >60 km away, which they have never visited, that has been selected by a foreign researcher using a statistical matching process. Instead, the most appropriate comparator from their perspective may well be the village just outside the protected area on the main road who are currently perceived to be enjoying a much better standard of living. A completely different methodological approach to social assessment would then be to work with local people to understand how they perceive their livelihoods are changing and how they attribute causality (Catley et al., 2008). Such an exercise might well generate a completely different result that is just as valid as the statistical methods presented here, but for a very different audience. This dichotomy is particularly relevant for the evaluation of conservation projects, which impose costs on local people in addition to providing benefits.

A Turbulent Past, An Uncertain Future

Over the past four decades, Cambodian lives have been plagued by war, genocide and insecurity, a continuing history of a society dominated by patronage relationships that have existed for 100s of years. To a Cambodian villager insecurity is the norm, discount rates are high, and the normative approach is to take what is available now and hope for the future. The recent rapid sell-off of the forests and land where they live is merely yet another manifestation of the exploitation and manipulation of the rural poor. Adjusting to new externally-imposed demands, foreign or domestically driven, is a basic part of existence, to be exploited where possible and endured otherwise. In this, conservation is just as culpable as other great ideas that have aimed to improve human existence but fallen painfully short (Scott, 1998). Amidst such insecurity and cynicism, attempting to build or facilitate any institutions for sustainable management of natural resources is an incredible challenge.

The basic 'deal' offered by conservation in this context is a mixture of safeguarding rights, providing incentives, and accepting responsibilities. Local people receive more secure rights (tenure over forest resources and land) and benefits (from PES), in exchange for more responsibilities (local village management of land-use, protection of wildlife). The results from this thesis suggest that the elements of this basic system are tenable. But the system is under siege: more than 10% of Cambodia's protected areas network has been sold off to developers over the past four years (Vrieze & Naren, 2012), whilst approval of community forests and the indigenous land titling processes proceed at glacial speed (Biddulph, 2011). If conservation cannot guarantee the rights of local people, then why should they accept additional responsibilities? In this context, the system collapses.

Is there a new 'deal' under these circumstances? Conservation will only succeed if it is grounded in institutions that are socially and politically acceptable. Increasing democratization and civil society action in some places is leading to a new constituency of Cambodians that are able to lobby for their rights, including security of forest tenure. Extensive protests by local civil society groups regarding the proposed sell-off of one of Cambodia's last remaining high-value forest areas led to the Royal Government declaring a moratorium on all concessions in May 2012, and cancellation of the specific proposals for that area. International agencies and non-government organisations have not always been a completely helpful presence for nascent Cambodian civil society, tending both to dominate and avoid contentious political issues (Hughes, 2007). As Cambodia moves forward into the 21st century, perhaps the new conservation consensus is grounded in Cambodian people safeguarding their own rights and aspirations for themselves. In this context, payment mechanisms may have a greater, not a lesser, role to play, helping to fund and empower local actors as they take their own decisions.

Appendix

Chapter 3. Model Selection Tables

Table S3.1. The most conservative model for the effect of household and village-level variables on household BNS score. The table shows (a) the change in AICc caused by removing each term from the model; (b) the change in AICc caused by adding other main effects to the model; (c) the change in AICc due to adding other interactions to the model; and (d) change in AICc for interactions with Village Type.

The final model selected was the most conservative model, plus the main effects of 'Rent out labor' and 'Years of schooling in the village', which were the only main effects that improved the AICc score (Table S3.1b). The final model had 27 parameters, an AICc score of 2.62 units less than the most conservative model, and an R^2 of 0.66. Validation plots confirmed that the model met the assumptions, in terms of homogeneity of the variance and normality of errors.

Most Conservative Model:
Household-level Main Effects: Female-headed [Yes/No] + Education (yrs) + Own >1 hectare [Yes/No] + Number of livelihood strategies + Own mini-tractor or draft animals [Yes/No] + Operate a business [Yes/No] + Resin-tapper [Yes/No] + Rice harvest in 2007/8 (kg) + Number of cattle
Village-level Main Effects: Village Type [Inside/Outside/Border PA] + Village population size + Time to Provincial Capital (hrs)
Interactions with Village Type: Village Type * Number of cattle + Village Type * Resin-tapper [Yes/No] + Village Type * Own mini-tractor/draft animals [Yes/No]
Other Interactions: Number of livelihood strategies * Female-headed [Yes/No] + Operate a business [Yes/No] * Education (yrs) + Time to Provincial Capital (hrs) * Rice harvest in 2007/8 (kg)
Random Effects: Village

Model	**K[a]**	**AICc Δ**
(a) Change in AICc for removing each term from the most conservative model		
- Village Type [Inside/Outside/Border PA] * Resin-Tapper [Yes/No]	23	3.37
- Village Type [Inside/Outside/Border PA] * Own mini-tractor/draft animals [Yes/No]	23	4.85
- Village Type [Inside/Outside/Border PA] * Number of cattle	23	3.88
- Village Type [Inside/Outside/Border PA] and interactions	17	15.08
- Education (yrs) and interactions	23	27.33
- Own >1 hectare [Yes/No]	24	43.09
- Female-headed household [Yes/No], number of livelihood strategies and interactions	22	28.19
- Operate a business [Yes/No] and interactions	23	5.74
- Own mini-tractor/draft animals [Yes/No] and interactions	22	45.33
- Number of cattle and interactions	22	53.64
- Resin-Tapper [Yes/No] and interactions	22	11.71
- Rice harvest in 2007/8 (kg)	24	27.49
- Time to Provincial Capital (hrs) * Rice harvest in 2007/8 (kg)	24	10.27
- Village population size	24	7.91
(b) Change in AICc for adding other main effects to the most conservative model		
+ Number of Working Adults	26	2.02
+ Household head Age ^ 2	27	2.35
+ Rent out labour [Yes/No]	26	-1.15
+ Employed [Yes/No]	26	1.38

+ Rice harvest in 2007/8 (kg) ^ 2	26	1.10
+ Years of schooling in the village	26	-1.77
+ Time to Secondary School (hours)	26	1.22
(c) Change in AICc for adding other possible interactions added to the most conservative model		
+ Education (yrs) * Number of Working Adults	26	1.98
+ Number of Working Adults + Education (yrs) * Number of Working Adults	27	1.55
+ Number of Working Adults * Household head Age ^2	27	1.27
+ Number of livelihood strategies * Resin-Tapper [Yes/No]	26	-0.56
+ Own mini-tractor/draft animals [Yes/No] * Female-headed Household	26	-1.68
+ Own mini-tractor/draft animals [Yes/No] * Rice harvest in 2007/8 (kg)	26	1.07
+ Employed [Yes/No] + Employed [Yes/No] * Education (yrs)	27	0.00
+ Time to Provincial Capital (hrs) * Rice harvest in 2007/8 (kg) ^2	26	1.84
+ Time to Secondary School (hrs) * Education (yrs)	26	-0.64
+ Time to Secondary School (hrs) + Time to Secondary School (hrs) * Education (yrs)	27	1.14
(d) Change in AICc for interactions with Village Type		
Village Type * (Resin-tapper + Own mini-tractor/draft animals + Number of Cattle)	25	0
Village Type * (Resin-tapper + Number of Cattle)	23	3.37
Village Type * (Resin-tapper + Own mini-tractor/draft animals)	23	3.88
Village Type * (Resin-tapper)	21	4.04
Village Type * (Resin-tapper + Number of Cattle)	23	4.85
Village Type * (Own mini-tractor/draft animals + Number of Cattle + Rice harvest)	25	7.31
Village Type * (Own mini-tractor/draft animals)	21	7.34
Village Type * (Resin-tapper + Number of Cattle + Rice harvest)	25	7.84
Village Type * (Resin-tapper + Own mini-tractor/draft animals + Rice harvest)	25	8.08
Minus Village type and all interactions with Village type	17	15.08

[a] **K** = number of parameters in the model

Table S3.2. The most conservative model for the effect of household and village-level variables on household rice harvest in 2007/8. The table shows (a) the change in AICc caused by removing each term from the model; (b) the change in AICc caused by adding other main effects to the model; (c) the change in AICc due to adding other interactions to the model; and (d) change in AICc for interactions with Village Type.

The final model selected was the most conservative model, plus the main effect of 'Village population size', which was the only main effect that improved the AICc score (Table S3.2b). The final model had 25 parameters, an AICc score of 0.32 units less than the most conservative model, and an R^2 of 0.49. Validation plots confirmed that that the model met the assumptions, in terms of homogeneity of the variance and normality of errors.

Most Conservative Model:
Household-level Main Effects: Number of Working Adults + Own >1 hectare [Yes/No] + Number of livelihood strategies + Own mini-tractor or draft animals [Yes/No] + Operate a business [Yes/No] + Resin-tapper [Yes/No] + Employed [Yes/No] + Rent out labour [Yes/No] + Number of cattle
Village-level Main Effects: Village Type [Inside/Outside/Border PA] + Time to Provincial Capital (hrs) + Time to Secondary School (hrs)
Interactions with Village Type: Village Type * Resin-tapper [Yes/No] + Village Type * Own >1 hectare [Yes/No]
Other Interactions: Own >1 hectare [Yes/No] * Own mini-tractor or draft animals [Yes/No] + Own mini-tractor or draft animals [Yes/No] * Number of cattle + Time to Provincial Capital (hrs) * Time to Secondary School (hrs)
Random Effects: Village

Model	**K[a]**	**AICc Δ**
(a) Change in AICc for removing each term from the most conservative model		
- Village Type [Inside/Outside/Border PA] * Resin-Tapper [Yes/No]	21	2.63
- Village Type [Inside/Outside/Border PA] * Own >1 hectare [Yes/No]	21	4.98
- Village Type [Inside/Outside/Border PA] and interactions	17	6.52
- Employed [Yes/No]	22	13.94
- Rent out labour [Yes/No]	22	13.00
- Operate a business [Yes/No]	22	4.89
- Number of livelihood strategies	22	12.74
- Number of working adults	22	32.86
- Own >1 hectare [Yes/No] and interactions	19	87.76
- Own mini-tractor/draft animals [Yes/No] and interactions	20	27.47
- Number of cattle and interactions	21	18.84
- Time to Provincial Capital (hrs) * Time to Secondary School (hrs)	22	4.93
- Time to Provincial Capital (hrs) and interactions	24	5.01
- Time to Secondary School (hrs) and interactions	24	6.33
(b) Change in AICc for adding other main effects to the most conservative model		
+ Female-headed household	24	0.16
+ Education (yrs)	24	1.92
+ Household head Age ^2	25	2.11
+ Village population size	24	-0.31
(c) Change in AICc for adding other possible interactions added to the most conservative model		
+ Resin-tapper [Yes/No] * Number of Working Adults	24	1.05
+ Education (yrs) + Education (yrs) * Own mini-tractor/draft animals [Yes/No]	25	-0.14
+ Education (yrs) + Education (yrs) * Own >1 hectare [Yes/No]	25	0.76
+ Education (yrs) + Education (yrs) * Own mini-tractor/draft animals [Yes/No] +	26	0.58

Education (yrs) * Own >1 hectare [Yes/No]		
+ Own mini-tractor/draft animals [Yes/No] * Own >1 hectare [Yes/No] * Education	27	0.09
+ Own >1 hectare [Yes/No] * Number of cattle	24	2.13
+ Female-headed household + Female-headed household * Number of livelihood strategies	25	2.52
+ Employed [Yes/No] * Number of working adults	24	1.42
+ Education (yrs) * Household head Age ^2 + Household head Age ^2	27	0.28
+ Education (yrs) * Number of working adults	24	2.16
+ Education (yrs) * Household head Age ^2 + Education * Number of working adults + Household head Age ^2	28	1.89
(d) Change in AICc for interactions with Village Type		
Village Type * (Resin-tapper + Own >1 hectare)	23	0
Village Type * (Own >1 hectare)	21	2.63
Village Type * (Resin-tapper + Own >1 hectare + Own mini-tractor/draft animals)	25	2.69
Village Type * (Resin-tapper + Own >1 hectare + Number of Cattle)	25	3.32
Village Type * (Resin-tapper)	21	4.98
Village Type * (Own >1 hectare + Employed)	23	5.37
Village Type * (Own >1 hectare + Own mini-tractor/draft animals)	23	5.63
Village Type * (Own >1 hectare + Number of Cattle)	23	6.05
Minus Village type and all interactions with Village type	17	6.52

[a] **K** = number of parameters in the model

Chapter 4. Population and Ecological Data on Large Waterbirds

Table S4.1. Nests Found and Protected: 2004-2012. Giant ibis nests were monitored only. '-' indicates species that were present, but were not protected in that year. Brackets give number of colonies protected for colonial species only.

Species	*2003-4*		*2004-5*		*2005-6*		*2006-7*		*2007-8*		*2008-9*		*2009-10*		*2010-11*		*2011-2*	
	Nests	*Chicks*	*Nests*	*Chicks*	*Nests*	*Chicks*	*Nests*	*Chicks*	*Nests*	*Chicks*	*Nests*	*Chicks*	*Nests*	*Chicks*	*Nests*	*Chicks*	*Nests*	*Chicks*
Kulen Promtep Wildlife Sanctuary																		
White-shouldered Ibis	1	1	2	4	3	4	4	2	5	7	5	4	4	6	5	5	6	10
Giant Ibis	5	n/a	9	12	7	14	9	16	11	12	10	17	18	18	8	11	11	21
Sarus Crane	6	n/a	3	3	7	11	9	12	19	30	24	36	23	39	24	37	26	35
Red-headed Vulture	-	-	1	1	1	1	-	-	-	-	-	-	-	-	-	-		
Black-necked Stork	-	-	-	-	2	6	3	10	2	5	2	7	1	0	2	5	2	7
Oriental Darter	-	-	-	-	-	-	26 (1)	53	33 (1)	103 *(b)*	9 (1)	*(b)*	38 (2)	51	78 (4)	218	50 (2)	203
Greater Adjutant	*(a)*	n/a	21 (2)	38	17 (2)	32	18 (2)	29	10 (2)	20	6 (3)	10	10 (4)	19	5 (3)	11	3	6
Lesser Adjutant	34 (5)	52	32 (7)	56	38 (7)	68	140 (14)	239	159 (18)	310	146 (16)	304	125 (15)	233	94 (16)	168	112 (17)	201
Total	**46+**	**53+**	**68**	**113**	**75**	**136**	**209**	**361**	**239**	**487**	**202**	**378**	**219**	**366**	**216**	**455**	**210**	**483**
Preah Vihear Protected Forest																		
Giant Ibis	-	-	18	34	21	38	19	36	19	19	7 *(c)*	n/a	23	40	11	21	20	38
Sarus Crane	-	-	16	19	22	30	28	39	35	42	33	54	29	50	20	33	24	38
White-rumped Vulture	-	-	-	-	3	3	4	3	4	2	3	3	4	4	2	1	3	3
Red-headed Vulture	-	-	-	-	-	-	1	1	1	1	2	1	-	-	1	0	-	-
Lesser Adjutant	-	-	65 (9)	66	96 (8)	186	81 (8)	140	118 (9)	166	115 (11)	185	150 (11)	288	64 (10)	108	140 (14)	264
Total	**-**	**-**	**99**	**119**	**142**	**257**	**133**	**219**	**177**	**230**	**160**	**243+**	**206**	**382**	**98**	**163**	**187**	**343**
Totals, both	**46+**	**53+**	**167**	**232**	**217**	**393**	**342**	**580**	**416**	**717**	**362**	**621**	**425**	**748**	**314**	**618**	**397**	**826**

(a) present; *(b)* some or all nests destroyed by crows; *(c)* incomplete surveys.

Table S4.2. Nesting Seasons in the Northern Plains

	Northern Plains		
Species	*Start Nesting*	*Eggs Hatch*	*Fledging*
Wet Season (May-November)			
Giant Ibis	June-Aug		Aug-Oct
Sarus Crane	June-Aug	July-Sept	
Oriental Darter	Sept	Sept	Nov
Wet Season – Dry Season			
Greater Adjutant	Nov	Dec-Jan	Mar-May
Lesser Adjutant	Sep-Oct	Dec	Dec-Feb
Dry Season (November-April)			
Black-necked Stork	Nov-Dec	Jan	Feb-Mar
White-shouldered Ibis	Dec-Jan	Jan	Feb-Mar
Vulture species	Nov	Dec-Jan	Mar

Table S4.3. Characteristics of nests of the different species protected

Species	*Nest Description*	*Habitat*	*Predation*
Sarus Crane	Mound of sticks in grassland	Seasonally flooded grasslands	Asiatic Jackal
White-shouldered Ibis	Small platform of sticks at top of *Dipterocarpus intricatus* trees	Deciduous Dipterocarp Forest	
Giant Ibis	Platform of sticks on the side branches of *D. intricatus*, *D. tuberculatus* or *Hopea odorata*.	Deciduous Dipterocarp Forest	Civets, Eagles?
Oriental Darter	Small nest platforms on trees and shrubs above inundated areas.	Flooded Forest along the Stung Sen river.	Large-billed Crows
Lesser Adjutant	Large nests in high trees, *D. intricatus*, *D. tuberculatus*, *D. alatus*, *D. costatus* or *H. odorata*.	Deciduous Dipterocarp or Evergreen Forest	Large-billed Crows
Greater Adjutant	Large nests in high trees, *D. alatus* or *D. costatus*.	Evergreen Forest	

Chapter 6. Wellbeing Models and Model Selection Tables

Table S6.1. Final mixed effects models of household wellbeing in 2011 only for the entire dataset (*n* = 1053). The models show the effect of household and village-level variables in 2011 on (a) household poverty (measured using the Basic Necessities Survey score), (b) household rice harvest and (c) household food security. The table shows the coefficient values for the final model, all of which have a high level of empirical support based on the AICc Δ values. Part (ii) shows the contrasts tests for significant differences between treatment levels.

(i) Mixed Effects models for effect of interventions on household wellbeing variables in 2011

Coefficient	(a) Poverty	(b) Rice Harvest	(c) Food Security
Impacts of PA and PES Interventions			
Intervention [Border PA]	5.915	21.264	76.502
Intervention [Inside PA]	5.315	20.634	77.348
Intervention [Outside PA]	5.692	18.373	75.756
Ibis Rice program, member [Yes]	0.598 *	4.333 **	2.804 **
Ecotourism program, member [Yes]	0.912 *		
Intervention [Inside PA] : Resin-tapper [Yes]	-0.419 ns	4.896 *	5.168 *
Intervention [Outside PA] : Resin-tapper [Yes]	-1.384 *	-0.479 ns	-0.696 ns
Intervention [Inside PA] : Own >1 hectare [Yes]		-8.581 *	
Intervention [Outside PA] : Own >1 hectare [Yes]		-3.485 ns	
Household Characteristics			
Female-headed household [Yes]	-0.701 *	-2.760 (*)	
Working adults		5.913 ***	
Household size			-2.895 ***
Age of household head (years, square-root)	1.864 ns		
Age of household head, squared (years, square-root)	-5.815 *		
Education of household head (years, square-root)	0.330 ***	1.705 ***	1.316 ***
Household Livelihood Strategies			
Rice Farmer Type [None]	1.116 *	-38.532 ***	-12.393 ***
Rice Farmer Type [Shifting Cultivation only]	-0.657 (*)	-9.397 ***	-3.970 **
Rice Farmer Type [Paddy only]	0.035 ns	-5.676 ***	-3.581 ***
Own >1 hectare [Yes]	1.883 ***	10.651 **	3.982 ***
Resin-tapper [Yes]	0.721 (*)	-2.169 ns	
Employed [Yes]	0.863 **		2.047 *
Service provider or Shop-keeper [Yes]	1.697 ***	2.286 *	1.296 *
Household Assets			
Rice Harvest (kg, square-root)	0.046 ***	n/a	n/a
Cattle total (heads)	0.767 ***	4.668 ***	2.958 ***
Own Mini-tractor [Yes]		8.496 ***	5.665 ***
Interactions			
Rice Farmer Type [None] : Own Mini-tractor [Yes]		-19.762 **	-14.601 ***
Rice Farmer Type [Shifting Cultivation only] : Own Mini-tractor [Yes]		-10.382 *	-7.536 **
Rice Farmer Type [Paddy only] : Own Mini-tractor		-1.040 ns	-1.331 ns

[Yes]			
% residual variation due to the random effect of Village	8.7 %	4.7 %	4.5 %

(ii) Tests of the differences between interventions for household wellbeing variables in 2011

Contrasts	(a) Poverty	(b) Rice Harvest	(c) Food Security
Intervention [Outside PA] > Intervention [Inside PA]	-0.013 ns	-0.609 ns	-0.215 ns
Intervention [Border PA] > Intervention [Inside PA]	0.692 ns	6.207 **	3.748 **
Intervention [Outside PA] > Intervention [Inside PA] : Resin-tapper [Yes]	-0.966 *	-5.036 *	
Intervention [Outside PA] > Intervention [Inside PA] : Own > 1 hectare [Yes]		4.548 (*)	1.731 ns

Table S6.2. Final mixed effects models for change in household wellbeing between 2008 and 2011 for the panel dataset (*n* = 769). The models show the effect of household and village-level variables on (a) change in household poverty (measured using the Basic Necessities Survey score), (b) change in household rice harvest and (c) change in household food security from 2008 to 2011. The table shows the coefficient values for the final model, all of which have a high level of empirical support based on the AICc Δ values. Part (ii) shows the tests of difference tests for significant differences between treatment levels. Tables S6.3-S6.5 give the model selection tables for the three models shown here.

(i) Mixed Effects models for the change in household wellbeing variables between 2008 and 2011

Coefficient	(a) Poverty	(b) Rice Harvest	(c) Food Security
Impacts of PA and PES Interventions			
Intervention [Border PA]	5.836	32.323	66.315
Intervention [Inside PA]	5.303	34.646	62.865
Intervention [Outside PA]	6.236	29.695	63.000
Ibis Rice program, member [Yes]	0.712 *	5.195 **	3.767 **
Ecotourism program, member [Yes]	1.094 *		
Intervention [Inside PA] : Resin-tapper [Yes]	-0.141 ns		
Intervention [Outside PA] : Resin-tapper [Yes]	-1.397 *		
Intervention [Inside PA] : Own >1 hectare [Yes]		-8.264 *	
Intervention [Outside PA] : Own >1 hectare [Yes]		-1.088 ns	
Base Variables			
Basic Necessities Survey score in 2008	-0.608 ***		
Rice Harvest in 2008		-0.799 ***	
Rice Surplus in 2008			-0.846 ***
Household Characteristics			
Female-headed household [Yes]	-0.911 **		
Working adults, Change		0.827 *	
Household size, Change			-0.599 **
Age of household head (years, square-root)	4.092 ns		
Age of household head, squared (years, square-root)	-6.967 **		
Education of household head (years, square-root)	0.209 *	2.325 ***	1.645 ***
Household Livelihood Strategies			
Rice Farmer Type [None]	-1.276 *	-38.917 ***	-12.795 ***
Rice Farmer Type [Shifting Cultivation only]	-0.945 *	-10.728 ***	-4.491 **
Rice Farmer Type [Paddy only]	-0.437 *	-7.123 ***	-4.425 ***
Own >1 hectare [Yes]	2.164 ***	8.338 *	3.257 ***
Resin-tapper [Yes]	0.728 (*)		
Employed [Yes]	0.856 **		
Service provider [Yes]	1.455 ***		
Shop-keeper [Yes]	1.547 ***		
Service provider or Shop-keeper [Yes]		2.421 *	1.376 *
Household Assets			
Rice Harvest, change (kg, square-root)	0.017 ***	n/a	n/a
Cattle total, change (heads)	0.419 ***		0.948 *

Own Mini-tractor [Yes]		4.060 ns	6.916 ***
Own Draft Cattle [Yes]		3.901 **	2.038 **
Interactions			
Own Mini-tractor [Yes] : Own >1 hectare [Yes]		7.230 *	
Rice Farmer Type [None] : Own Mini-tractor [Yes]		21.817 **	-17.581 ***
Rice Farmer Type [Shifting Cultivation only] : Own Mini-tractor [Yes]		-8.748 (*)	-7.240 *
Rice Farmer Type [Paddy only] : Own Mini-tractor [Yes]		-0.449 ns	-1.055 ns
% residual variation due to the random effect of Village	10.4 %	2.3 %	4.3 %

(ii) Tests of the differences between interventions for change in household wellbeing variables between 2008 and 2011

Contrasts	(a) Poverty	(b) Rice Harvest	(c) Food Security
Intervention [Outside PA] > Intervention [Inside PA]	0.417 ns	0.829 ns	0.135 ns
Intervention [Border PA] > Intervention [Inside PA]	0.496 ns	4.945 *	3.450 *
Intervention [Outside PA] > Intervention [Inside PA] : Resin-tapper [Yes]	-1.257 **		
Intervention [Outside PA] > Intervention [Inside PA] : Own > 1 hectare [Yes]		7.176 *	

Table S6.3. The selected model for the effect of household and village-level variables on the change in household poverty (measured using the Basic Necessities Survey score) during 2008-2011. The table shows (a) the change in AICc caused by removing each term from the model; (b) the change in AICc caused by adding other main effects to the model; and (c) change in AICc for adding other interactions with the PA intervention.

Selected model:
Interventions: PA type [Border/Inside/Outside PA], Ibis Rice programme member, Ecotourism programme member
Base Variable: Poverty in 2008, measured using the Basic Necessities Survey score
Household Characteristics: Female-headed [Yes/No], Age of Household Head (squared), Education level of Household Head (in years)
Household Livelihood Strategies: Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither], Own >1 hectare [Yes/No], Resin-tapper [Yes/No], Employed [Yes/No], Provide a Service [Yes/No], Operate a Shop [Yes/No]
Household Assets: Change in Rice Harvest (kg), Change in Number of Cattle (heads)
Interactions with PA intervention: PA Type * Resin-tapper [Yes/No]
Other Interactions: none
Random Effects: Village

Models	**K[a]**	**AICc Δ**
(a) Change in AICc for removing each term from the selected model		
- Ibis Rice program, member	23	1.60
- Ecotourism program, member	23	3.99
- PA Type [Inside/Outside/Border PA] * Resin-tapper [Yes/No]	20	2.56
- BNS Score in 2008	23	268.67
- Female-headed household [Yes/No]	23	5.51
- Education level of household head (in school years completed)	23	3.73
- Age of household head (squared function)	22	7.09
- Resin-tapper [Yes/No] and interactions	21	6.11
- Own >1 hectare [Yes/No]	23	66.43
- Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither]	21	4.49
- Rice Harvest, change from 2008-2011 (kg)	23	10.72
- Provide a Service [Yes/No]	23	40.61
- Employed [Yes/No]	23	4.95
- Operate a Shop [Yes/No]	23	23.62
- Number of Cattle, change from 2008-2011 (heads)	23	10.31
(b) Change in AICc for adding other main effects to the most conservative model		
+ Bird Nest program, member	25	2.12
+ Change in Household size (individuals)	25	-0.85
+ Change in number of Working Adults (individuals)	25	0.14
+ Travel time to Provincial Capital, change (hours)	25	-0.39
+ Travel time to full-day Market, change (hours)	25	2.09
+ Village Population, change (number of households)	25	2.11
+ Education level available in the village, change (number of school years)	25	2.12
(c) Change in AICc for adding interactions between livelihood strategies with PA Type		
+ PA Type * Female-headed household [Yes/No]	26	-1.33
+ PA Type * Own >1 hectare [Yes/No]	26	3.78
+ PA Type * Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither]	30	6.45
+ PA Type * Provide a Service [Yes/No]	26	3.31
+ PA Type * Operate a Shop [Yes/No]	26	2.86

+ PA Type * Employed [Yes/No]	26	1.29
+ PA Type * Rice Harvest, change from 2008-2011 (kg)	26	1.32
+ PA Type * Number of Cattle, change from 2008-2011 (heads)	26	1.04

[a] **K** = number of parameters in the model

Table S6.4. The selected model for the effect of household and village-level variables on the change in household Rice Harvests during 2008-2011. The table shows (a) the change in AICc caused by removing each term from the model; (b) the change in AICc caused by adding other main effects to the model; and (c) change in AICc for adding other interactions with the PA intervention.

Selected model:
Interventions: PA type [Border/Inside/Outside PA], Ibis Rice programme member
Base Variable: Rice Harvest in 2008 (kg)
Household Characteristics: Change in Number of Working Adults, Education level of Household Head (in years)
Household Livelihood Strategies: Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither], Own >1 hectare [Yes/No], Provide a Service or Operate a Shop [Yes/No]
Household Assets: Own a Mini-Tractor [Yes/No], Own Draft Livestock [Yes/No]
Interactions with PA intervention: PA Type * Own >1 hectare [Yes/No]
Other Interactions: Own >1 hectare [Yes/No] * Own a Mini-Tractor [Yes/No], Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither] * Own a Mini-Tractor [Yes/No]
Random Effects: Village

Models	**K[a]**	**AICc Δ**
(a) Change in AICc for removing each term from the selected model		
- Ibis Rice program, member	21	5.34
- PA Type [Inside/Outside/Border PA] * Own > 1 hectare [Yes/No]	18	8.55
- Rice Harvest in 2008	21	548.95
- Education level of household head (in school years completed)	21	25.28
- Change in number of Working Adults (individuals)	21	2.79
- Own >1 hectare [Yes/No] and interactions	18	15.66
- Agriculture Type [Paddy/Shifting Cultivation/Both/Neither] and interactions	16	218.62
- Provide a Service or Operate a Shop [Yes/No]	21	2.84
- Own a Mini-tractor [Yes/No] and interactions	17	81.88
- Own Draft Livestock [Yes/No]	21	9.06
- Own >1 hectare [Yes/No] * Own a Mini-Tractor [Yes/No]	21	3.11
- Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither] * Own a Mini-Tractor [Yes/No]	19	4.60
(b) Change in AICc for adding other main effects to the most conservative model		
+ Bird Nest program, member	23	2.07
+ Ecotourism program, member	23	1.54
+ Change in Poverty (Basic Necessities Survey score)	23	2.11
+ Change in Household size (individuals)	23	-0.27
+ Female-headed household [Yes/No]	23	1.91
+ Age of Household Head (years, squared function)	24	0.52
+ Resin-tapper [Yes/No]	23	1.04
+ Employed [Yes/No]	23	1.62
+ Number of Cattle, change from 2008-2011 (heads)	23	-0.17
+ Travel time to Provincial Capital, change (hours)	23	1.90
+ Travel time to full-day Market, change (hours)	23	1.53
+ Village Population, change (number of households)	23	1.97
+ Education level available in the village, change (number of school years)	23	1.58
(c) Change in AICc for adding interactions between livelihood strategies with PA Type		
+ PA Type * Female-headed household [Yes/No]	25	4.29
+ PA Type * Change in number of Working Adults	24	-1.05
+ PA Type * Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither]	28	4.26

+ PA Type * Resin-tapper [Yes/No]	25	1.76
+ PA Type * Provide a Service or Operate a Shop [Yes/No]	24	-1.09
+ PA Type * Employed [Yes/No]	25	5.61
+ PA Type * BNS Score, change from 2008-2011	25	6.12
+ PA Type * Number of Cattle, change from 2008-2011 (heads)	25	2.54

[a] **K** = number of parameters in the model

Table S6.5. The selected model for the effect of household and village-level variables on the change in household Food Security during 2008-2011. The table shows (a) the change in AICc caused by removing each term from the model; (b) the change in AICc caused by adding other main effects to the model; and (c) change in AICc for adding other interactions with the PA intervention. Food security was defined as the difference between a household's rice harvest (from paddy and shifting cultivation) minus the household's food needs for a year.

<u>Selected model:</u>
Interventions: PA type [Border/Inside/Outside PA], Ibis Rice programme member
Base Variable: Food Security in 2008 (kg)
Household Characteristics: Change in Household Size (individuals), Education level of Household Head (in years)
Household Livelihood Strategies: Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither], Own >1 hectare [Yes/No], Provide a Service or Operate a Shop [Yes/No]
Household Assets: Own a Mini-Tractor [Yes/No], Own Draft Livestock [Yes/No], Change in Number of Cattle (heads)
Interactions with PA intervention: none
Other Interactions: Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither] * Own a Mini-Tractor [Yes/No]
Random Effects: Village

Models	**K[a]**	**AICc Δ**
(a) Change in AICc for removing each term from the selected model		
- Ibis Rice program, member	19	7.04
- PA Type [Inside/Outside/Border PA]	18	3.73
- Food Security in 2008 (kg)	19	531.60
- Change in number of Household Size (individuals)	19	7.58
- Education level of household head (in school years completed)	19	30.26
- Own >1 hectare [Yes/No]	19	10.73
- Agriculture Type [Paddy/Shifting Cultivation/Both/Neither] and interactions	14	93.74
- Provide a Service or Operate a Shop [Yes/No]	19	1.63
- Own a Mini-tractor [Yes/No] and interactions	16	72.05
- Own Draft Livestock [Yes/No]	19	5.47
- Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither] * Own a Mini-Tractor [Yes/No]	19	3.35
(b) Change in AICc for adding other main effects to the most conservative model		
+ Bird Nest program, member	21	1.70
+ Ecotourism program, member	21	1.83
+ Change in Poverty (Basic Necessities Survey score)	21	1.57
+ Female-headed household [Yes/No]	21	2.00
+ Age of Household Head (years, squared function)	22	3.94
+ Resin-tapper [Yes/No]	21	1.86
+ Employed [Yes/No]	21	0.08
+ Travel time to Provincial Capital, change (hours)	21	2.09
+ Travel time to full-day Market, change (hours)	21	0.26
+ Village Population, change (number of households)	21	2.01
+ Education level available in the village, change (number of school years)	21	2.09
+ Own >1 hectare [Yes/No] * Own a Mini-tractor [Yes/No]	21	0.08
(c) Change in AICc for adding interactions between livelihood strategies with PA Type		
+ PA Type * Female-headed household [Yes/No]	23	5.04
+ PA Type * Rice Farmer Type [Paddy/Shifting Cultivation/Both/Neither]	26	4.90

+ PA Type * Resin-tapper [Yes/No]	23	4.85
+ PA Type * Provide a Service or Operate a Shop [Yes/No]	22	2.14
+ PA Type * Employed [Yes/No]	23	4.14
+ PA Type * BNS Score, change from 2008-2011	23	5.51
+ PA Type * Number of Cattle, change from 2008-2011 (heads)	22	1.73

[a] **K** = number of parameters in the model.

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